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RESEARCH

Monitoring Contrasting Land Management in the Savanna Landscapes of Northern Australia

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Abstract We compared measures of ecosystem state across six adjacent land-tenure groups in the intact tropical savanna landscapes of northern Australia. Tenure groups include two managed by Aboriginal owners, two national parks, a cluster of pastoral leases, and a military training area. This information is of relevance to the debate about the role of indigenous lands in the Australian conservation estate. The timing and frequency of fire was determined by satellite imagery; the biomass and composition of the herb-layer and the abundance of large feral herbivores by field surveys; and weediness by analysis of a Herbarium database. European tenures varied greatly in fire frequencies but were consistently burnt earlier in the dry season than the two Aboriginal tenures, the latter having intermediate fire frequencies. Weeds were more frequent in the European tenures, whilst feral animals were most abundant in the Aboriginal tenures. This variation strongly implies a signature of current management and/or recent environmental history. We identify indices suitable for monitoring of management outcomes in an extensive and sparsely populated landscape. Aboriginal land offers a unique opportunity for the conservation of biodiversity through the maintenance of traditional fire regimes. However, without financial support, traditional practices may prove unsustainable both economically and because exotic weeds and feral animals will alter fire regimes. An additional return on investment in Aboriginal land management is likely to be improved livelihoods and health outcomes for these disadvantaged communities.

Keywords Aboriginal lands · Feral herbivores · Fire regimes · National parks · Weeds

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The savanna landscapes of northern Australia provide particular challenges for ecosystem monitoring (Whitehead and others 2000). Considerable loss of biodiversity has occurred even in the absence of land clearing or any other gross disturbance (Franklin 1999; Woinarski and others 2001; Woinarski and Catterall 2004). This has been attributed to often-subtle environmental shifts consequent to land-management practices (Woinarski 1999), particularly those associated with the discontinuation of traditional Aboriginal landscape-burning practices. The decline of granivorous birds, a major biodiversity conservation concern in northern Australia (Franklin 1999), has been least severe in areas of ongoing Aboriginal occupation (Franklin and others 2005). However, developing practical on-ground management to arrest or reverse the loss of biodiversity in northern Australia is not easy (Whitehead and others 2003a). A major uncertainty concerns the relative roles of and interactions among threatening processes, such as weed infestation, feral animal impacts, and changes to fire regimes.

The abundance and distribution of feral animals and weeds are practical and at times pivotal indices of ecosystem health in Australia generally (Tait and others 2000; Stone and others 2001) and northern Australia in particular (Whitehead and others 2000). Northern Australia is home to a variety of feral herbivores (Ridpath 1991). The populations of many of these are not regulated by predators or pathogens (Freeland 1990; Freeland and Choquenot 1990), leading to high densities and serious environmental impacts (*e.g.*, Braithwaite and others 1984; Corbett and others 1996; Skeat and others 1996). The most abundant and widespread are the Asian water buffalo (*Bubalus bubalis*), the feral pig (*Sus scrofa*), and the feral horse (*Equus caballus*) (Bayliss and Yeomans 1989). Numerous nonnative plant species have also become established in the region, some of which have transformed ecosystems in a fundamental and detrimental manner. Of particular concern is the floodplain shrub mimosa (*Mimosa pigra*) (Braithwaite and others 1989) and a range of pasture species, including gamba grass (*Andropogon gayanus*), para grass (*Urochloa mutica*), and mission grass (*Pennisetum polystachion*) (Lonsdale 1994; Houston and Duivenvoorden 2002; Rossiter and others 2003; Ferdinands and others 2005). Feral herbivores may promote weeds by digging, trampling, and eliminating palatable native species (Skeat and others 1996); they may also alter fire regimes by changing fuel loads and composition (Werner 2005).

The development of appropriate fire management in northern Australia has proven to be complex and perplexing. These savannas were fire-prone long before the arrival of humans (Bowman 2002). Fires were frequent in the northern savannas when the first Europeans arrived (Fensham 1997; Preece 2002) and have remained so under more or less continuous Aboriginal stewardship in northern Arnhem Land (Haynes 1985; Bowman and others 2004). Changes in the frequency, timing, and spatial extent of landscape fires under European management (Crowley and Garnett 1990) have been implicated in the loss of biodiversity (Williams and others 2002; Andersen and others 2005). Fire is currently used and/or managed in the region for a variety of objectives, including the maintenance of hunting grounds and "cleaning country" on Aboriginal tenures; promotion of fresh growth for livestock in pastoral areas; conservation (particularly national parks); and the prevention of late dry-season wildfires (Haynes 1985; Andersen and Braithwaite 1992; Ash and others 1997; Murphy and Bowman 2007). One hypothesis for a deleterious impact has been a feed-back loop in which dominance by native annual grasses of the genus *Sarga* (ex *Sorghum*) has promoted more extensive fires (Yibarbuk and others 2001; Miles 2003; Bowman and others 2004; Bowman and others 2007a), although the mechanisms involved remain unclear.

Contrasts amongst tenures may shed light on the impacts of management, providing "natural experiments" that may be employed to identify ecologic processes and differing perceptions of ecologic health. For example, introduced pasture plants are seen as a valuable economic resource by some pastoralists, yet they threaten the integrity of an adjoining national park. Asian water buffalo that spread from nineteenth century British settlements may be seen by indigenous land owners as "belonging" to tribal lands, yet they are considered by conservationists as unwanted pests that warrant extermination (Bowman and Robinson 2002; Robinson and Whitehead 2003; Robinson and others 2005). Press (1988) compared fire regimes across three tenures in northern Australia and found that fires occurred earlier in pastoral areas than in Kakadu National Park and Arnhem Land. Bowman and others (2007a) demonstrated that herb-layer biomass and the abundance of annual *Sarga* was higher under European nonpastoral management than Aboriginal and mixed tenure.

Comparison of the ecologic state of different land tenures is important given growing interest in "off-reserve" conservation. In the Northern Territory, this is a particularly important approach given that the formal conservation estate is $< 5\%$ of the land area, whereas Aboriginal people collectively own (in inalienable, communal title) approximately 80% of the coastline and 50% of the land, almost all of which retains natural vegetation cover. Notwithstanding this ownership, Aboriginal people have exceptionally low income and rates of education and employment, and disease and mortality rates are high (Linacre 2004). In the savanna landscapes, as elsewhere, there is a need to develop sustainable livelihoods based on the natural environment. One sustainable employment opportunity for land-holding Aborigines is the management of biodiversity and maintenance of ecosystem services funded as part of a national conservation or resource management strategy (Altman and Whitehead 2003; Altman and Cochrane 2005). Employment in or engagement with land management also yields substantial health benefits (Burgess and others 2005). There have been differences of opinion amongst conservation biologists as to whether Aboriginal people currently have sufficient skills to manage land for conservation outcomes. Andersen (1999) argued that fire management should be developed de novo to meet the novel challenges of biodiversity conservation. The alternative view is that Aborigines have a rich traditional skill set that should be deployed as part of a broader conservation strategy (Yibarbuk and others 2001; Whitehead and others 2003b). However, the debate has featured more rhetoric than hard data.

Table 1 Summary of the ecosystem state indices assessed in this study

Theme	Indices and their units	Tenures	Land systems	Method
Fire	1. Frequency (no. of years burnt / 4) 2. seasonality (month burnt)	All	Lowland savanna & floodplain	Remotely-sensed fire history for the years 2002 to 2005
Herb-layer	1. Biomass (t/ha) 2. Cover of annual *Sarga* (%) 3. Cover of other annual grasses (%)	All except pastoral	Lowland savanna	Quantitative field assessment of 120 sites with dominant vegetation type in the dry season of 2005
Weeds	1. Weed index (*WI* – see Methods section)	All	Lowland savanna & floodplain	Analysis of Herbarium database
Feral herbivores	1. Frequency of animal signs (% of sample points)	All except pastoral & MBTA	Floodplain and adjacent savanna	Quantitative field assessment of animal signs around 31 wetlands

MBTA = Mount Bundy (military) Training Area

Table 2 Summary characteristics of tenure units

Tenure unit	Area (km^2)			Management
	Total	Lowland savanna	Floodplain	
Pastoral	1,921	1,219	698	Long-standing private pastoral holdings
MRNP	1,026	324	677	Managed by the Parks and Wildlife Service (Northern Territory)
MBTA	1,056	955	73	Australian Army, ground-based military training
KNP, Kakadu	10,559	7,457	2,562	Joint management, Department of the Environment and Water Resources (Australian Government) and traditional owners; the north of the Park, which is included in this study, was reserved in sections in 1965, 1979, and 1984
Oenpelli	12,099	7,128	1,315	Aboriginal; traditional owners hold inalienable communal property rights
Maningrida	12,677	7,257	985	Aboriginal; traditional owners hold inalienable communal property rights

MRNP = Mary River National Park; MBTA = Mount Bundy (military) Training Area; KNP = Kakadu National Park

In this social, scientific, and political context, we quantitatively compared six adjacent land-tenure units in the savannas of northern Australia using indices relevant to four environmental themes: fire, the composition and biomass of the herb-layer, and the abundance of weeds and feral herbivores (Table 1). These land-tenure units include two Aboriginal-owned and -managed areas, two national parks (one of which is Aboriginal-owned and jointly managed), military land, and a number of adjacent pastoral leases (Table 2). Our aims were to shed light on the state of ecosystems under differing management regimes, to assess the practicality of doing so, and to briefly consider opportunities to engage Aborigines in land management in northern Australia.

Methods

Study Area

The study area stretches 350 km along the north coast of the Northern Territory, Australia, from the Mary River (13°S, 132°E) to the Blythe River (13°S, 135°E), and approximately 100 km inland (Fig. 1). The regional climate is monsoonal, with mean annual rainfall in the range of 1200 to 1450 mm. More than 95% of the rain falls in the 7 months from October to April inclusive, and the months June to August are typically quite rainless. Temperatures are warm to hot throughout the year. The combination of warm temperatures and a long rainless period following an intense wet season renders the area extremely fire prone (Gill and others 1996). During the dry season, humans are the only source of ignition, but lightning can ignite fires during the dry-wet transition (Bowman 2002).

We considered six tenure-based land units (Fig. 1A and Table 2). In the vicinity of the Mary River, a series of pastoral leases are aggregated into a pastoral unit. On these leases, cattle are raised commercially principally for live export but also for domestic consumption. The proposed Mary River National Park (MRNP) is a conservation zone maintained by the Northern Territory Parks and Wildlife Service, whilst the adjacent Mount Bundy Training Area (MBTA) is a military reservation used to train army personnel. To the east of these tenures lies the World Heritage–Listed Kakadu National Park, the northern (lowland) half of which was considered in this study.

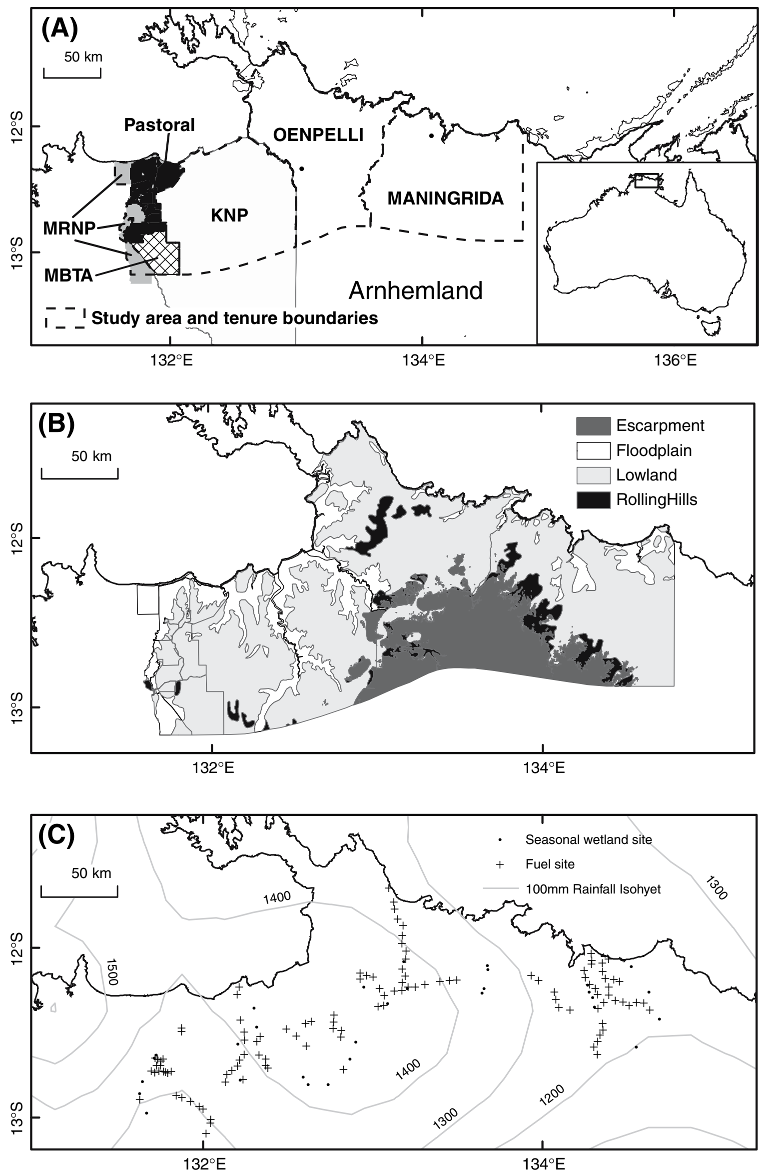

Fig. 1 The study area. (**A**) Tenure units. Towns of the same name in the Oenpelli and Maningrida tenures are indicated with a *dot*. (**B**) Land systems. (**C**) Herb-layer and feral animal sites and 100-mm rainfall isohyets. MRNP = proposed Mary River National Park; MBTA = Mount Bundy (military) Training Area; KNP = Kakadu National Park

Kakadu National Park is managed jointly by traditional owners and the Australian Department of the Environment and Water Resources for its natural and cultural values. The eastern half of our study area is part of an Aboriginal-owned estate known as Arnhem Land. The two areas considered here are owned and managed by people based in or around the communities of Oenpelli and Maningrida, respectively.

All tenures except Maningrida were managed as low-intensity pastoral leases during the early twentieth century in which the harvest of feral Asian water buffalo for their hides was the major industry. The intensification of pastoralism and its restriction to current tenures was prompted in considerable part by the eradication of a major portion of the buffalo population in the late 1980s (Freeland and Boulton 1990). This eradication program was concentrated in areas around and to the west of Kakadu National Park, with only limited activity in Arnhem Land and none in the Maningrida area (P. Whitehead, personal communication, 2006).

For the purpose of this study, we recognised four land systems: floodplains, lowland savannas, rolling savanna-clad hills, and sandstone uplands (Fig. 1B). Our study was largely concentrated in the floodplains and widespread lowland savannas, because sandstone uplands do not occur, and the rolling savanna-clad hills occur only sparingly in some tenures. Floodplains are characterized by deep estuarine and depositional clays that are seasonally or permanently inundated. Depending on the length of inundation, floodplain vegetation may be a grassland, a woodland, or an open forest, the main tree species being *Melaleuca* spp. (Finlayson and Woodroffe 1996). The lowland savannas typically have lateritic sand or loam soils and are frequently dominated by the trees *Eucalyptus tetrodonta* (Darwin stringybark) and *Eucalyptus miniata* (Darwin woollybutt) (Wilson and others 1990). In this study, land systems were identified digitally from a combination of a digitised vegetation map (Wilson and others 1990) and a digital elevation model.

Fire Frequency

Annual fire scar polygons were provided by the Tropical Savannas Cooperative Research Centre for the period 2002 to 2005 for the entire area of all tenures. They had been derived from Moderate Resolution Imaging Spectroradiometer (MODIS) satellite imagery (resolution 250 m) (Justice and others 2002) captured at 2 to 4 week intervals. Polygons were segmented from the MODIS images captured at 2 to 3 week intervals using ECognition software v4.0 (Definiens AG, Munich). Fire polygons were intentionally overclassified and exported to ArcGIS software v9.1 (Environmental Systems Research Institute Inc., Redlands, California) for manual editing of misclassified polygons. Each fire polygon was attributed with the month of image capture. Additionally, each polygon layer was converted to a 100×100 m raster layer, and each raster was attributed with a value of 1 if a fire had occurred within the raster during the year. The four resulting raster layers were summed to provide an index of fire frequency.

Herb-Layer Biomass and Composition

Field sampling (n = 120 sites; Fig. 1C) was conducted between April and June 2005, *i.e.,* soon after the end of the wet season, in lowland savannas dominated by either or both *E. tetrodonta* and *E. miniata*. Sampling sites were constrained to be at least 5 km apart in the Kakadu NP, Oenpelli, and Maningrida tenures and 2 km apart in the smaller MRNP and MBTA units. Only areas unburnt since the end of the 2004 to 2005 wet season were sampled, and these only within unburnt patches >500 m long to minimize bias toward areas where fire is less likely to carry because fuel loads are too low. In addition, sites had to meet the following criteria: (1) at least 50% of the basal area was made up of a combination of *E. tetrodonta* and *E. miniata*; (2) there were no signs of mechanical disturbance in the canopy or understorey; and (3) exotic grasses were absent or a minor component of the herb layer.

At each site, a plot of 10-m radius was randomly selected. All herbs within either five or nine 0.5 x 0.5 m quadrats were collected, the greater number at sites judged a priori to be heterogeneous at the quadrat scale. One quadrat was placed at the plot centre; four were placed 10m from the centre in the four cardinal directions; and, where needed, the additional quadrats were placed halfway between. Quadrat herb samples were combined, placed in cotton bags, and subsequently oven dried at 80°C for 48 hours and then weighed. Dry mass values were converted to tonnes/ha.

At each site, basal area and tree species composition was recorded from the plot centre using a basal area wedge. Soil colour was noted, and clay content estimated with the ribbon test of McDonald and others (1998). The following additional environmental variables were recorded on a 0 to 5 Braun-Blanquet cover scale: annual *Sarga* spp., other annual grasses, perennial grasses, and surface gravel cover.

The response variables considered in the analyses were: (1) herb biomass; (2) percentage cover of annual *Sarga*; (3) percentage cover of perennial grasses; and (4) percentage cover of other annual grasses. We considered whether site factors (mean annual rainfall, basal area, soil texture, and gravel cover) could account for tenure-based differences using generalised linear models. Herb biomass was treated as a continuous variable with a gamma distribution. Percent cover data featured high frequencies of zero values, so we partitioned the data into low and high cover and modelled these as binomial distributions. For annual *Sarga* and perennial grasses, low and high cover were dichotomised at 25%, whilst the sparser other annual grasses were dichotomised at 10%. Soil texture was treated as a continuous variable. Mean annual rainfall was derived from gridded monthly climate data for 1961 to 1990 at 0.25° of latitude–longitude from the National Climate Centre of the Bureau of Meteorology Research Centre, Melbourne, Australia. For each response variable, we evaluated the following models using Akaike Information Criterion (AIC_c; Burnham and Anderson 2002) and information–theoretic model weighting: tenure, tenure + all site factors, all site factors, all site factors except basal area, and the null model. Our rationale for this subset of all possible models was that we aimed to discriminate between tenure and site factors as contributors to variation.

Weeds

A spatial database of plant species records was obtained from the Northern Territory Herbarium. We constrained the database to include only the 43,646 plant records within 5 km of our study area, including 906 records of nonnative species (weeds). Records were attributed to 10×10–km grid cells (Fig. 2), and each cell attributed to a land system, a tenure, and a distance from the nearest road based on the centre of the cell. Distance to roads were calculated from a Northern Territory government spatial database of major and minor roads. Only cells attributed to lowland savanna

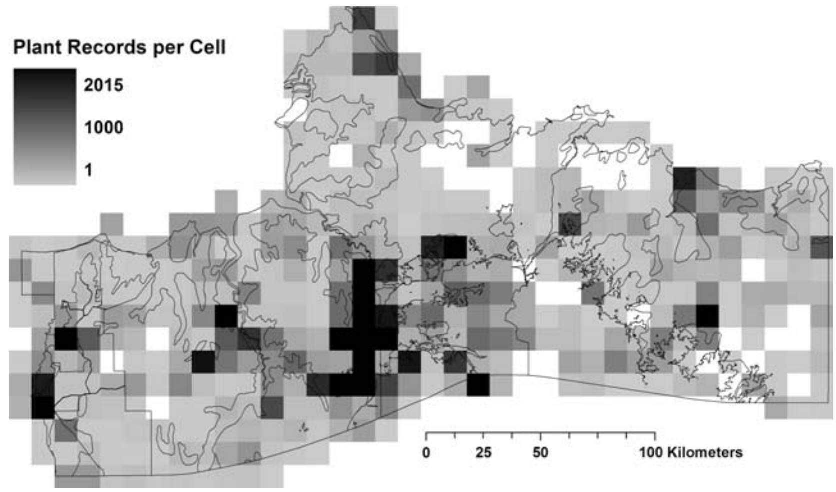

Fig. 2 Location of 43,646 plant records for the weeds analysis in 10 × 10–km grid cells. *White cells* indicate no records

and floodplain were analysed further, and the category of MBTA floodplain was discarded because it was represented by only a single cell. For each cell, we summed the number of records (n_{all}), the number of records of weeds (n_{weed}), the number of species (R_{all}), and the number of species of weed (R_{weed}). A weed index, *WI*, which aimed to describe the richness of the weed flora compared to the native flora, was calculated to include a correction for variation in sampling effort across the study area as

$$WI = (R_{weed}/n_{weed})/(R_{all}/n_{all})$$

Cells with $n_{weed} = 0$ were given an index value of 0.

We used a recursive partitioning algorithm, or "regression tree" (Breiman and others 1984), to model *WI*. A regression tree is a nonparametric, nonadditive, and assumption-free means of interrogating data sets that is particularly useful for sets with large numbers of categoric variables. It successively partitions a data set based on binary splits in the values of explanatory variables that best reduce overall deviance in the data set. We modelled the *WI* for each land system separately as a function of tenure and distance from road. We stopped splitting the data when further splits failed to decrease the unexplained deviance by >1%, and then pruned the floodplain tree to avoid iterative divisions on distance from settlement.

Feral Animal Impacts

Thirty-one seasonal, wooded wetlands corresponding to the shallow, seasonally dry *Melaleuca* communities (usually *M. dealbata, M. viridiflora*, and/or *M. cajuputi*) of Finlayson and Woodroffe (1996) were selected across four tenures (Table 1 and Fig. 1C). At each wetland, 8 × 100–m transects were established in an inner and outer set of four transects each. The inner set were placed radially in cardinal directions from the low centre of the wetland and within the wooded wetland vegetation type. The outer four were placed in line with the inner four but in adjacent, usually upslope vegetation. Vegetation in each of the outer transects was assigned to one of five categories: (1) shallow seasonally dry *Melaleuca* communities defined as above; (2) treeless grass and sedgelands; (3) dry upland *Melaleuca* communities dominated by *M. nervosa* and *Asteromyrtus symphyocarpa*; (4) seasonally inundated open woodlands characterized by a co-occurrence of *Melaleuca* spp. (usually *M. viridiflora* and *M. nervosa*) with one or more of *Corymbia bella*, *C. polycarpa*, *C. latifolia*, and *C. foelscheana*; and (5) dry open woodland, a vegetation type characterized by well-drained sandy soils and little or no wet season inundation and usually dominated by *Eucalyptus tetrodonta* and/or *E. miniata* frequently with *Erythrophleum chlorostachys* and *Lophostemon lactifluus* subdominant.

At 10-m intervals along each transect, a 5-m rope with knots spaced at 50 cm was laid out perpendicular to the transect. At each knot, herbivore surface disturbance was attributed to species by a combination of footprints and the type of disturbance (pig diggings, buffalo wallows), and ground cover was classified as bare, sedge, grass, woody, or litter. Each transect, therefore, had a score out of 100 recorded for animal disturbance and ground cover. At each intersect between the transect and rope, the ground was recorded as burnt or unburnt. At the centre point of each transect, basal area by tree species was recorded using a basal area wedge.

Summary data are presented for each animal species, but for statistical analyses all species were summed, including the "unknown" class, because the data were sparse. The latter could include native species, though this is thought unlikely, but was only a minor component of the total (see "Results"). Extent of animal damage was converted to proportions, arcsine-transformed, and modelled with a Gaussian distribution and an identity link function in the program R v. 2.3.1 (R Foundation for Statistical

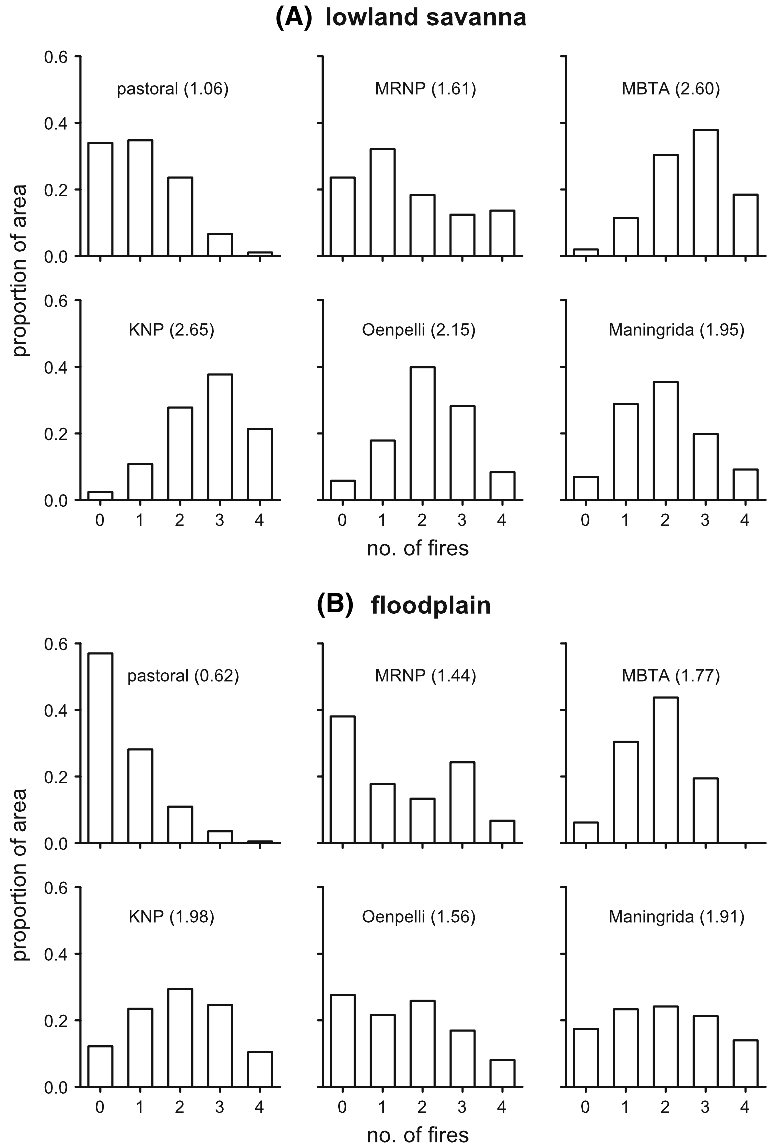

Fig. 3 Distribution of fire frequencies (number of years burnt) across six tenure units and two land systems (**A**) lowland savanna, and (**B**) floodplain, for the 4 years from 2002 to 2005. Numbers in brackets after tenure unit names are the mean number of years burnt (of 4) per pixel. MRNP = proposed Mary River National Park; MBTA = Mount Bundy (military) Training Area; KNP = Kakadu National Park

Computing, Vienna). The inner and outer transects were analysed separately, the former as generalised linear models and the latter as generalised linear mixed models using the *lme4* package v. 0995-2. For the inner transects analysis, the four transects were combined to avoid pseudoreplication. For the analysis of outer transects, transects were retained as replicates with wetland included as a random effect (additional to other effects listed below) in all models to control for nonindependence of transects within wetlands.

For both analyses, the value of models were evaluated using the AIC_c (Burnham and Anderson 2002) and information–theoretic model weighting. We first identified the best subset of effects other than tenure unit. We refer to this subset as the vegetation model. We then evaluated five models: global, null, tenure, vegetation, and tenure + vegetation. Our rationale for this subset of all possible models was that we aimed to discriminate between tenure and site factors as contributors to variation. Vegetation variables included in the preliminary analyses and the global models were basal area of trees, cover of herbs, percentage burnt, the interaction of cover of herbs and percentage burnt and, in the analysis of the outer sets only, vegetation type.

Results

Fire

In the lowland savannas, fire frequencies in 250×250–m pixels ranged from 1.06 in 4 years in the pastoral zone, to 2.60 in 4 years in the MBTA and 2.65 in 4 years in Kakadu National Park (Fig. 3A), with intermediate values in the Aboriginal lands of Oenpelli and Maningrida and in

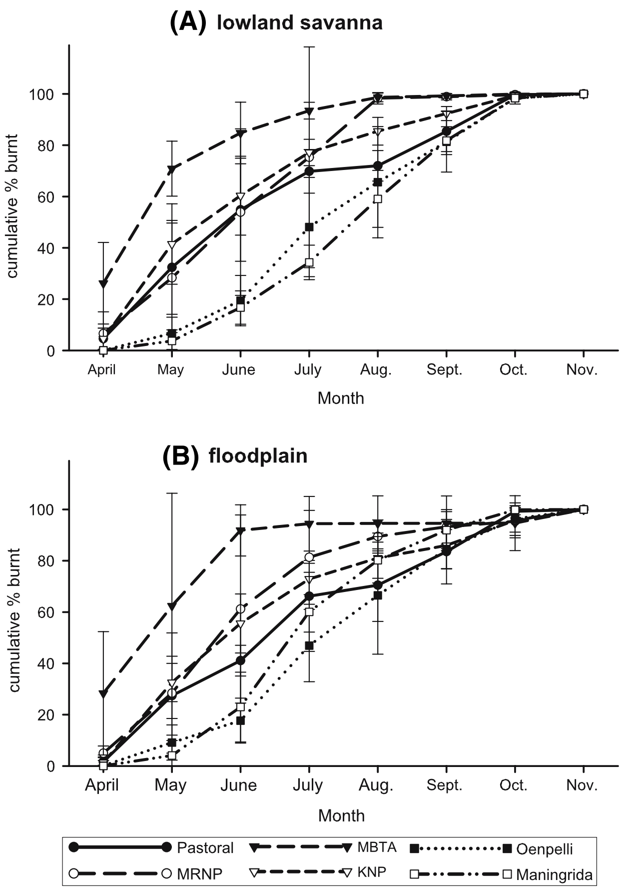

Fig. 4 Season of fire for (**A**) lowland savanna, and (**B**) floodplain, illustrated by the cumulative area burnt as a percentage of all area burnt in a year. Error bars are SD, expressing variation among the 4 years of the study

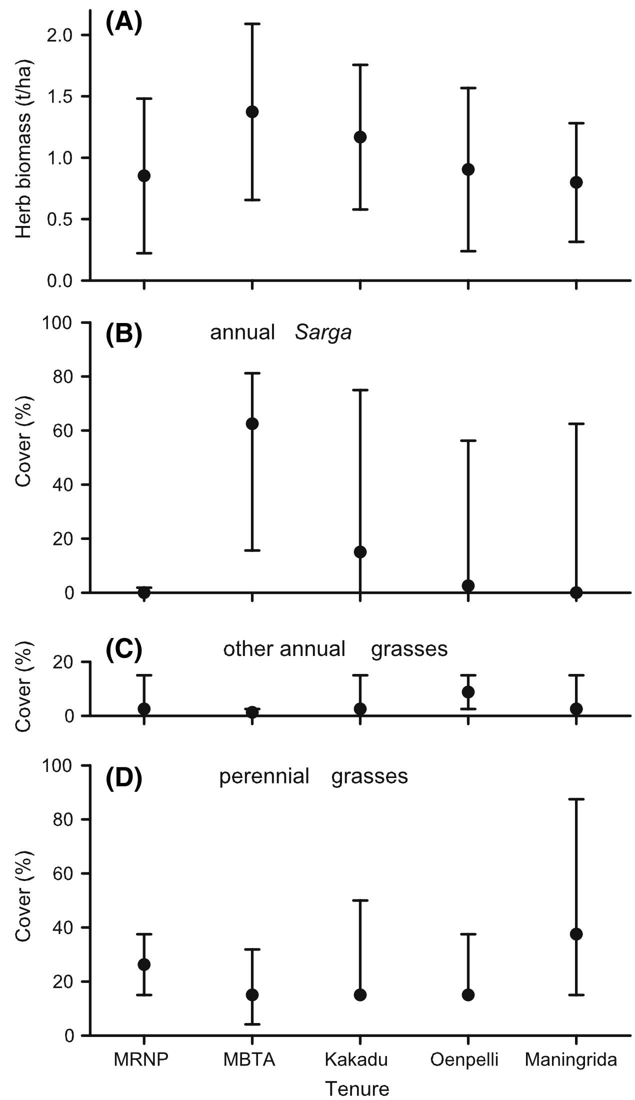

Fig. 5 Biomass (**A**) and composition (**B** – annual *Sarga*, **C** – other annual grasses, and **D** – perennial grasses) of the herb-layer in *Eucalyptus miniata* and *E. tetrodonta* lowland savannas across five tenures. Herb biomass (**A**) data are mean ± SD. Grass cover data (**B**, **C**, and **D**) are median ± 25th to 75th percentile. Sample sizes are MRNP = 16; MBTA = 8; Kakadu = 29; Oenpelli = 28; and Maningrida = 39

MRNP. In Kakadu and MBTA, <2.5% of the lowland savannas remained unburnt in four years, whereas 34% of the pastoral zone remained unburnt (Fig. 3A). The lowland savannas in MBTA were mostly burnt in the first half of the dry season, with approximately 70% burnt before the end of May (Fig. 4A). In contrast, the Aboriginal lands of Oenpelli and Maningrida were mostly burnt in the second half of the dry season, whilst those of Kakadu, MRNP, and the pastoral lands were burnt more uniformly throughout the dry season.

Fires were somewhat less frequent on the floodplains than the lowland savannas in all tenures (range among tenures 0.62 to 1.98 in 4 years), but the order of frequency across tenures was similar (Fig. 3B) as was the timing of the fires (Fig. 4B). The portion of floodplains that remained unburnt was somewhat to markedly greater than in the savannas, with the marked difference in Kakadu and the Aboriginal tenures (12% to 28% unburnt) reflecting a more even distribution of fire frequencies within these tenures (Fig. 3B).

Grass Fuels

Herb biomass was highest in the MBTA and lowest in MRNP and Maningrida (Fig. 5A). However, this variation did not simply reflect land tenure, given that there was 62% support for a model that included only site factors, but 31% support for a model that combined site factors and tenure (Table 3, herb biomass).

Annual *Sarga* was most abundant in MBTA (though the sample size was small), secondarily in Kakadu, and especially low in MRNP. Perennial grasses were notably abundant in Maningrida and secondarily in MRNP. Annual

Table 3 Generalised linear models for herb biomass and the cover of its key constituents

Model	AIC_c	Δ_i	w_i	%DE
Herb biomass				
Basal area + soil texture + gravel cover + MA rainfall	186.9	0.00	0.616	12.9
Basal area + soil texture + gravel cover + MA rainfall + tenure	188.2	1.37	0.310	18.3
Tenure	192.8	5.92	0.032	7.2
Soil texture + gravel cover + MA rainfall	193.4	6.58	0.023	6.7
Null	193.8	6.89	0.020	0.0
Cover of annual *Sarga*				
Tenure	160.5	0.00	0.533	6.3
Null	161.8	1.32	0.276	0.0
Soil texture + gravel cover + MA rainfall	163.5	3.05	0.116	4.4
Basal area + soil texture + gravel cover + MA rainfall	165.1	4.61	0.053	4.8
Basal area + soil texture + gravel cover + MA rainfall + tenure	166.9	6.44	0.021	9.6
Cover of perennial grasses				
Basal area + soil texture + gravel cover + MA rainfall	168.8	0.00	0.390	7.1
Soil texture + gravel cover + MA rainfall	169.5	0.73	0.271	5.3
Null	169.6	0.87	0.253	0.0
Tenure	172.3	3.57	0.065	3.6
Basal area + soil texture + gravel cover + MA rainfall + tenure	174.71	5.96	0.020	9.2
Cover of other annual grasses				
Null	159.5	0.00	0.421	0.0
Soil texture + gravel cover + MA rainfall	160.2	0.72	0.294	5.1
Tenure	161.2	1.71	0.179	4.5
Basal area + soil texture + gravel cover + MA rainfall	162.3	2.80	0.104	5.2
Basal area + soil texture + gravel cover + MA rainfall + tenure	169.6	10.15	0.003	6.6

AIC_c = Akaike Information Criterion corrected for small sample sizes; Δ_i = the difference in AIC_c between the model and the best-supported model; w_i = the model weight; %DE = percent of deviance explained; MA rainfall = mean annual rainfall

grasses other than *Sarga* were a minor component of the herb-layer in all tenures but were most abundant in Oenpelli. Controlling for site factors, there was strong support for a tenure effect alone in the cover of annual *Sarga* (53%; Table 3, cover of annual sarga). However, the cover of perennial grasses was best explained by site factors (Table 3, cover of perrenial grasses), whilst the effect of tenure (18%) on the cover of other annual grasses was less well supported than the null model (42%) and site-factors model (29%). For all categories of grass cover, the percent of deviance explained was low, and there was some support (25% to 42%) for the null model.

Weeds

The lowland savanna regression tree (Fig. 6A) explained 46.3% of the deviance, and the floodplain tree (Fig. 6B) explained 57.9% of the deviance. In both land systems, the primary split was among tenures, with higher weed richness indices in pastoral, MRNP, and Kakadu tenures. The weed index was also higher closer to roads in both land systems, although the critical distance varied amongst analyses.

Feral Animals

In *Melaleuca* wetlands (the inner transects), a model that included an effect of tenure and a positive relationship between animal disturbance and the basal area of trees was well supported with 68% weighting and explained almost 60% of the deviance (Table 4). Animal disturbance was greatest in Maningrida, intermediate in MRNP and Oenpelli and least in Kakadu National Park, trends that are apparent in the raw data (Fig. 7).

Rates of animal disturbance were lower in adjacent vegetation (outer transects) than in *Melaleuca* wetlands in all tenures (Fig. 7). As with the inner transects, the best supported model (85% weighting with 46% of deviance explained; Table 4) contained tenure and a vegetation effect, the latter being an interaction between herb cover and percentage burnt in which burning had a negative effect on rates of animal disturbance. Animal disturbance

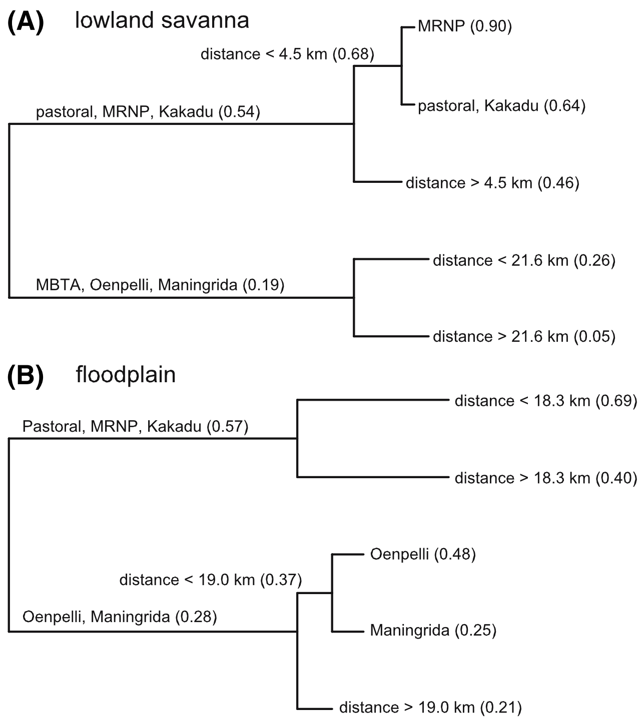

Fig. 6 Regression tree analyses of the weed index with tenure and distance from roads as explanatory variables for (**A**) lowland savanna, and (**B**) floodplain. Numbers in parentheses are mean weed index values for the group. Horizontal lines are proportional to the deviance explained, comparable within but not between trees. Sample sizes (no. of 10 × 10–km cells) are lowland savanna = 271 and floodplain = 88

Table 4. Generalised linear models (inner wetland transects) and generalised linear mixed models (outer mixed transects) for the frequency of animal disturbance in and near wetlands in four tenures in northern Australia

Model	AIC_c	Δ_i	w_i	%DE
Inner (wetland) transects				
Tenure + basal area	–17.91	0	0.680	59.4
Global	–15.16	2.75	0.172	67.2
Tenure	–14.86	3.05	0.148	51.2
Null	0.48	18.39	0.00007	0
Basal area	1.52	19.43	0.00004	3.5
Outer (mixed) transects				
Tenure + herbs*burn	–46.01	0	0.852	46.1
Tenure	–42.26	3.75	0.131	36.4
Global	–38.18	7.83	0.017	47.9
Herbs*burn	–30.62	15.39	0.0004	19.0
Null	–28.51	17.50	0.0001	0

For the vegetation model (see Methods section), preliminary analysis of nontenure effects demonstrated most support for the *basal area* of trees for the inner transects alone and most support for the interaction between herb cover and percentage burnt for the outer transects.

AIC_c = Akaike Information Criterion corrected for small sample sizes; Δ_i = the difference in AIC_c between the model and the best-supported model; w_i = the model weight; %DE = percent of deviance explained

in the outer transects was greatest in the Maningrida tenure and least in Kakadu (Fig. 7).

Trends across tenures in the species of animal, as suggested by the raw data (Fig. 7), were similar across transect groups. Feral pig abundance varied relatively little across tenures. Asian water buffalo were prevalent in MRNP and especially Maningrida but scarce elsewhere. Feral horses were markedly more abundant in Oenpelli than elsewhere.

Discussion

Indices of Ecosystem State

The landscapes of the study area retain their natural vegetation, and yet we detected marked differences among tenures for a variety of basic indicators of ecosystem state. European tenures varied greatly in fire frequencies but were consistently burnt earlier in the dry season, whereas the two Aboriginal tenures exhibited intermediate fire frequencies and were mostly burnt in the mid to late dry season. Fire frequencies were highest in the jointly managed Kakadu National Park, with fire occurring throughout the dry season. Weed richness was higher in the European tenures and Kakadu National Park, whilst feral animals were most abundant in the Maningrida Aboriginal tenure and least abundant in Kakadu National Park. This variation strongly implies a signature of current management and/or recent environmental history.

Indices relating to two of four themes considered in this study—fire and feral herbivores—provide efficient and powerful measures for monitoring management outcomes that we have demonstrated can be applied in large landscapes where human resources for monitoring are sparse. The analysis of fire frequency and seasonality is a desktop Gobal Information System exercise based on images that, for northern Australia, can be downloaded free from the internet. Whilst the frequency of animal signs requires the collection of field data, the protocols identified here worked effectively with moderate sampling effort and permitted statistical control for possible confounding environmental effects, although sample sizes were insufficient for full numeric analysis of individual feral animal species.

Herb-layer biomass and composition is influenced by land management in northern Australia (Bowman and others 2007a), but our data could only provide equivocal support for this notion. The failure of our statistical modelling to meaningfully extract a land-tenure effect most probably reflects the need for a much larger sample given

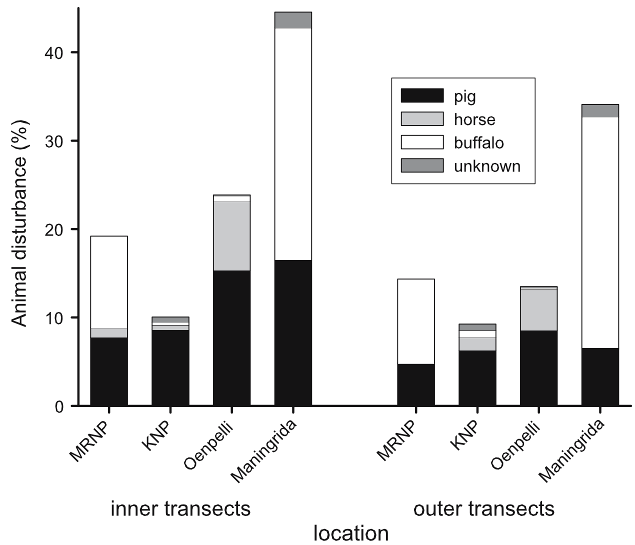

Fig. 7 Frequency of animal disturbance on wetlands (inner transects) and adjacent vegetation (outer transects) summarised by tenure and species

that we evaluated savanna dominated by two widespread tree species (*E. miniata* and *E. tetrodonta*) and that the system probably has high variance relative to the drivers of plant distributions and environmental change. Bowman and others (2007a) showed a tenure effect when considering only savannas dominated by *E. tetrodonta*. Sample sizes required for successful measurement of savanna fine-fuel mass across multiple land tenures is possibly beyond what was feasible to obtain in the available early portion of the dry season. However, relevant indices that could be sampled more rapidly and thus extensively are the cover of annual *Sarga* and the ratio of annual to perennial grasses. This study demonstrated an effect of tenure consistent with previous arguments that annual *Sarga* is an increaser in the face of the breakdown of traditional burning practices (Yibarbuk and others 2001; Miles 2003; Russell-Smith and others 2003; Bowman and others 2007a).

The weed index employed in this study has been informative, but because its source is a long-term Herbarium database, it is not readily replicable as an ongoing measure of land management outcomes. It also suffers the considerable deficiency of rating only presence rather than abundance of weed species and in not rating weed species according to their known or postulated environmental consequences. Some weeds of particular concern, such as mimosa and gamba grass, may be identifiable from fine-scale remote sensing. Indeed, images from Google Earth (http://www.earth.google.com/) have recently been used around Darwin in northern Australia to identify infestations of gamba grass (Peter Jacklyn, personal communication, 2007). Although the index fails to provide information about the abundance of weeds, weed richness is itself an important environmental indicator. Large-scale studies have found a positive association between weed richness and native species richness, a relationship thought to result from greater niche availabilty in more heterogenous habitats (Deutschewitz and others 2003). A high weed richness relative to native plant richness suggests, instead, the occurrence of multiple weed introduction events because of human activity. In a large-scale study in Mexico, based on herbarium records, Espinosa-Garcia and others (2004) found a positive association between human population density and weed richness, particularly for alien weeds. The higher *WI* values in our study in European land tenures, and close to roads, reflects a similar effect.

From a conservation perspective, the presence and abundance of weeds and feral animals (and perhaps also the cover of annual *Sarga*, a native species) are basic indices in that higher values represent poorer conservation outcomes. However, fire frequency and seasonality cannot be interpreted in such a simplistic way. Fire is an inevitably frequent feature of the northern Australian savanna landscape to which a wide variety of flora and fauna are adapted (Bowman 2002), some obligately (*e.g.* Dostine and others 2001). Nevertheless, the higher fire frequencies observed in the north Australian landscape are inimical to the conservation of biodiversity (Andersen and others 2005), and a case may be made for the maintenance of elements of the landscape at very low fire frequency (Woinarski and others 2004).

The seasonality of fire remains controversial. Early dry-season fires are considered necessary in some landscape settings to prevent intense and destructive late dry-season fires. In contrast, Haynes (1985), Yibarbuk and others (2001), and Bowman and others (2004, 2007b) presented a case that traditional burning occurs mostly in the mid and late dry season, is nevertheless of low intensity, and is optimal for the maintenance of biodiversity. Remote sensing may underestimate the frequency of early burning where individual fires are small and of low intensity (Russell-Smith and others 1997). Furthermore, rainfall early in the dry season is slightly higher in northwest Arnhem Land than in the western-most tenures considered here. This trend was evident in 3 of the 4 study years, with the tenure median east–west difference in the date on which the 99th percentile of the wet season's rain fell ranging from 0 to 25 days (A. Petty, unpublished data). However, this difference is insufficient to account for the 2.5- (floodplain) to 3-month (lowland savanna) difference in the dates by which 50% of burning had occurred (Fig. 4). Andersen and others (2005) argued that frequency rather than seasonality is the key issue. At least some of the variation in perspective may be attributable to the history of land management in the region, in which poor burning practice and/or release from intense grazing pressure by

Asian water buffalo (Werner 2005; Petty and others 2007) has increased grass biomass and/or its dominance by annual *Sarga*, prompting a need for preventative early burning that does not exist elsewhere.

Management Implications

We have shown a number of differences between land tenures. Furthermore, our analysis showed there was no simple pattern in which one tenure type is in a better state than another. These complex patterns reflect not only current management practices and differing management priorities but also the consequence of past land use. The latter point is well illustrated by the impacts of Asian water buffalo. Historically, buffalo have been concentrated in the Kakadu region (Petty and others 2007). The population is derived from introductions to two isolated British coastal settlements in the early 1800s and were observed in what is now Kakadu National Park by the first European explorer to visit the region in 1845. Populations were constrained by sustained hunting for hides through the first half of the twentieth century. After the collapse of a buffalo hide industry in the 1950s, populations erupted to approximately 15 animals per km^2. A government-sponsored eradication campaign in the late 1980s decreased populations by 99% in Kakadu and the Mary River catchment (Freeland and Boulton 1990; Skeat and others 1996), the impact of which is still evident today. Buffalo were uncommon in the Maningrida area in the mid 1980s (Bayliss and Yeomans 1989), but the tenure was not subject to the control measures extended to other tenures in the 1ate 1980s. This occurred in part because buffalo were an important component of subsistence for Aboriginal livelihoods in the area (Altman 1982). Our data suggest this tenure is now a population centre for the species. In the vicinity of wetlands, feral pigs had the highest densities in the Oenpelli and Maningrida tenures but were common in all tenures, reflecting the difficulty of controlling the species. Feral horses were most abundant in the Oenpelli tenure, possibly because the juxtaposition of floodplain with rock outcrops in this area provides optimal habitat. The pattern of weeds also seems to reflect historic land use, with the areas that have supported pastoralism and historically high numbers of buffalo having the highest weed indices. Fensham and Cowie (1998) noted that weed species richness was positively related to time since settlement. It is to be anticipated that weediness may increase in the Aboriginal tenures with time, particularly given current levels of disturbance by feral animals. There is need for pre-emptive action to stop this from occurring. The efficacy of pre-emptive weed control is demonstrated by ongoing control of the floodplain weed mimosa by staff of Kakadu National Park.

Of the indices considered here, the frequency and timing of fire is the most easily manipulated. Most fires in northern Australia are lit by humans, with the only natural source of ignition being lightning in October and November (Stocker 1966; Bowman 2002), at which time few fires occurred in this study. Lowland savannas on Aboriginal lands were burnt later in the dry season than other tenures, a pattern consistent with the research of Press (1988) and of Bowman and others (2004). Furthermore, the high fire frequencies associated with Kakadu National Park and the MBTA flag a marked excursion away from traditional Aboriginal fire regimes. In the early 1980s, Press (1988) noted the prevalence of early fires in the pastoral leases and later fires in Kakadu. Since the 1980s, there has been a deliberate policy to increase burning in the first half of the dry season with a measurable decreased in late dry-season fires (Russell-Smith and others 1997). This management approach is most pronounced at MBTA, which was burnt almost entirely early in the dry season, a practice that is driven by the need to eliminate the risk of fires started from using live ordinance on the military firing range later in the dry season. The markedly higher frequency of fire-free refuges in the savannas of pastoral lands and MRNP in this study is noteworthy, since Woinarski and others (2004) argued the need for such refuges.

The diversity of fire management described here provides natural experiments that are useful for resolving the ongoing debate about the merits of various land-management practices. For example, surveys in the Darwin area have suggested that low fire frequencies may be vital for the viability of small-mammal populations. This may explain the decrease in mammal populations in Kakadu National Park and the high diversity in areas under Aboriginal fire management (Woinarski and others 2001; Yibarbuk and others 2001; Pardon and others 2003). These contrasts have prompted recognition of the potential of Aboriginal land management for the conservation of biodiversity (Whitehead and others 2003b). Indeed, it has been argued that Aboriginal land management is a relative low-cost option for biodiversity conservation compared with the management of large national parks (Altman and Whitehead 2003).

The poor state of Aboriginal health and high rates of welfare dependency and poverty are topics of current policy discourse in Australia. Natural-resource management has the potential to provide much-needed employment on Aboriginal lands. Although the economics of indigenous off-reserve conservation remain uncertain, there is evidence that the health benefits for Aboriginal people may be substantial. Indeed, Burgess and others (2005) argued that natural and cultural resource management provide a culturally appropriate vehicle for health promotion and disease prevention through the associated improvements in diet, physical activity, autonomy, and social and spiritual connection to land. Even from a narrow economic perspective, employment in natural-resource management would appear to be an important health intervention strategy (Baker and others 2005).

For the foreseeable future, land in northern Australia will continue to be held in a diversity of tenures, each with its own management imperative. We are not arguing for homogeneity of management; indeed, there are advantages in diversity. However, it is useful to recognise the consequences of current management regimes as a basis for maintaining strengths and changing practices to improve on weaknesses. Aboriginal land may well provide a unique opportunity for the maintenance of biodiversity through the maintenance of current fire regimes, which combine customary practices with modern technology. However, without financial support, these practices may prove unsustainable both economically and because of the impact of weeds and ferals on fire regimes.

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