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Fire ecology and Aboriginal land management in central Arnhem Land, northern Australia: a tradition of ecosystem management

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Abstract

Aim To compare fire behaviour and fire management practice at a site managed continuously by traditional Aboriginal owners with other sites in tropical northern Australia, including the nearby Kakadu National Park, and relate those observations to indicators of landscape condition.

Location Dukaladjarranj, a clan estate in north-central Arnhem Land, in the seasonal tropics of northern Australia. The site abuts a vast sandstone plateau that is an internationally recognized centre of plant and animal biodiversity.

Methods Ecological assessments included: (1) mapping of the resource base of the estate from both traditional and ecological perspectives; (2) aerial survey of the extent of burning, distribution of the fire-sensitive native pine *Callitris intratropica*, rock habitats, and a range of macropod and other fauna resources; (3) fauna inventory; (4) detailed ecological assessment of the status of fire-sensitive vegetation; and (5) empirical assessment of the intensities of experimental fires. Ethnographic information concerning traditional fire management practice was documented in interviews with senior custodians.

Results Experimental fires lit during the study were of low intensity compared with late dry season fires reported elsewhere, despite weather conditions favouring rapid combustion. In contrast to other parts of the savanna, fuel loads comprised mostly leaf litter and little grass. We found that (i) a large proportion of the estate had been burned during the year of the study (ii) burned sites attracted important animal food resources such as large macropods (iii) important plant foods remained abundant (iv) well represented in the landscape were fire sensitive vegetation types (e.g. *Callitris intratropica* Baker & Smith woodlands) and slow growing sandstone 'heath' typically dominated by myrtaceaous and proteaceous shrubs (v) diversity of vertebrate fauna was high, including rare or range-restricted species (vi) exotic plants were all but absent. Traditional practice includes regular, smaller fires, lit throughout the year, and cooperation with neighbouring clans in planning and implementing burning regimes.

Main conclusions We attribute the ecological integrity of the site to continued human occupation and maintenance of traditional fire management practice, which suppresses

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otherwise abundant annual grasses (*Sorghum* spp.) and limits accumulation of fuels in perennial grasses (*Triodia* spp.) or other litter. Suppression of fuels and coordination of fire use combine to greatly reduce wildfire risk and to produce and maintain diverse habitats. Aboriginal people derive clear economic benefits from this style of management, as evidenced by abundant and diverse animal and plant foods. However, the motives for the Aboriginal management system are complex and include the fulfillment of social and religious needs, a factor that remains important to Aboriginal people despite the rapid and ongoing transformation of their traditional lifestyles. The implication of this study is that the maintenance of the biodiversity of the Arnhem Land plateau requires intensive, skilled management that can be best achieved by developing co-operative programmes with local indigenous communities.

Keywords

Fire regime, fire intensity, fire-sensitive vegetation, land management, savanna, clan estate, conservation, biological diversity.

INTRODUCTION

Managing landscape fire is a critical issue in many terrestrial ecosystems, particularly in highly flammable grass-dominated vegetation types such as seasonal tropical savanna and mid-latitude prairies. This is certainly the case for the vast tracts of highly fire-prone monsoon tropical savannas of northern Australia. In this biome contemporary fire regimes are characterized by numerous, geographically large wildfires (i.e. 10^2–10^4 km^2) that are repeated from year to year. A sparse, albeit locally concentrated, human population frustrates effective control of these geographically massive fires. This serious northern Australian management issue needs to be reconciled with the increasingly accepted view, formed by ecologists and anthropologists, that prior to European colonization landscape fire in northern Australia was effectively managed by small bands of nomadic hunters and gatherers. In contrast to the current vast tracts burnt by single fires it is thought that the Aborigines started many more spatially localized fires. Further, it is believed that the shift in spatial scale of fires has had profound ecological consequences resulting in the loss of fire sensitive vegetation and the predominance of fire-promoting grass fuels (Braithwaite & Estbergs, 1985). In short, frequent Aboriginal burning is thought to have had a negative feedback whilst European burning practices have had a positive feedback on the spatial scale and intensity of landscape fires.

The fire-management problem

About 50% of the woodland savannas in some regions of northern Australia are burnt during the 7-month (May–November) dry season each year (Braithwaite & Estbergs, 1985; Press, 1988), with some sites being burned every year (Russell-Smith *et al.*, 1997b). Contemporary fire regimes are typified by extensive, intense fires (> 2500 kW m^{-1}), burning uncontrolled late in the season (c. Aug–Oct), under extreme fire-weather conditions (Gill *et al.*, 1996). Of particular concern is the widespread destructive impact of such fire regimes on populations of relatively fire sensitive species (Bowman & Panton, 1993; Bowman, 1994; Lowe, 1995), and habitat/communities (McKenzie & Belbin, 1991; Russell-Smith & Bowman, 1992; Russell-Smith *et al.*, 1998).

Fires across northern Australia are lit mostly by people for a variety of purposes associated with indigenous (Aboriginal) land management/cultural practices, the pastoral industry, and conservation management (e.g. Press, 1988; Lewis, 1989; Haynes, 1991; Head *et al.*, 1992; Russell-Smith *et al.*, 1997a). To simplify, burning is undertaken from early in the dry season by both pastoral and Aboriginal traditional managers to provide 'green pick' for cattle, and macropods and other game, respectively, and by conservation and Aboriginal managers to impose patchy, spatio-temporal mosaics of burned and unburned country in order to enhance habitat diversity. Managers from all sectors also seek to use early 'preventative' burns to control late dry season wildfires. Fires ignited by lightning strike are relatively fewer in number and often more limited in areal extent, being restricted to a period of between 1 and 2 months at the start of the wet season (Stocker & Mott, 1981). Fire management is a contentious regional issue (Rose, 1995).

This paper considers the development of ecologically and economically sustainable approaches for long-term fire management in the remote, rugged, vast, biologically diverse, fire-prone region of north-western Arnhem Land, northern Australia. The area in question includes: (i) a range of fire-sensitive endemic plants, animals and communities/habitats (e.g. *Allosyncarpia* closed forests; sandstone heaths); (ii) is owned entirely by Aboriginal peoples under a range of tenures; (iii) includes two large contiguous protected areas, including the World Heritage Kakadu National Park, managed by separate federal and state government agencies; and (iv) in the present day, is very sparsely inhabited outside urban centres.

Specifically, we address: (1) the regional landscape management context; (2) a regional case study concerning Aboriginal approaches to fire management focusing on one Gunei family's traditional estate; (3) an ecological assessment (e.g. seasonality and extent of fires, fire intensity, habitat relations, distribution of fuels, effects on fire-sensitive components) of the traditional approach to fire management using standard survey methodologies; and (4) the relevance and application of this combined knowledge-base for landscape and fire management at different northern Australian regional scales. As such, the paper describes an application of ecosystem management (e.g. Agee & Johnson, 1988; McDonnell & Pickett, 1993; Grumbine, 1994; Wilcove & Blair, 1995; Christensen *et al*., 1996) for addressing an identified threatening process in a culturally and biologically diverse, internationally significant landscape; it demonstrates the utility of a cooperative, engaging, 'little science' approach (Cooperrider, 1996).

REGIONAL CONTEXT

Physical landscape

The study focuses on a tract of country on the northern rim of the vast sandstone Arnhem Plateau, in the upper catchment of the Cadell River, north-central Arnhem Land (Fig. 1). The plateau, mostly at less than 400 m elevation, comprises resistant, flat-bedded Middle Proterozoic quartzose sandstones, criss-crossed by tensional joints which have been deeply weathered and eroded to form a maze of narrow valleys and gorges (Rix, 1965; Needham, 1988). Soils, where present, are typically skeletal and infertile sands (Aldrick, 1976). Elevation in the upper Cadell River study area lies between 50 and 150 m, with resistant sandstone outliers interspersed amongst broad valleys of sandy colluvium. Localized areas of seasonally inundated floodplains, typically comprising fine-textured soils, are developed along some sections of the Cadell and its seasonal tributaries.

Vegetation cover of sandstone formations is mostly a low open-woodland, describing a complex of shrubby communities (e.g. *Acacia*, *Asteromyrtus*, *Calytrix*, *Hibbertia*, *Hibiscus*, *Pityrodia*, *Tephrosia*) with scattered emergent trees (e.g. *Eucalyptus*, *Gardenia*, *Terminalia*), interspersed with a substantial cover of highly flammable hummock spinifex grasses (*Triodia*). On deeper sands the vegetation takes an open-forest form dominated by *Eucalyptus* over a range of shrubs and slender perennial and annual grasses (e.g. *Aristida*, *Eriachne*, *Schizachyrium*, *Sorghum*). The fire-sensitive conifer, *Callitris intratropica*, occurs widely either singly or in copses on these sandy soils. Seasonally inundated floodplains support mostly *Melaleuca* and *Eucalyptus* over a range of vigorous perennial grasses. Rainforest species provide a narrow evergreen fringe along the Cadell River.

As for northern Australia generally, the regional climate is characterized by marked rainfall seasonality, with over 90% falling in the summer months November–April. Mean annual rainfall in the region is likely to be between 1000 and 1200 mm, based on records from the only regional weather station, at Maningrida, some 70 km to the north. While the amount of rainfall is highly variable from year to year, the rainy or wet season is a highly reliable event (Taylor & Tulloch, 1985). Daily maximum temperatures average above 30 °C over the year. Such a predictable seasonal climate, in combination with rampant growth of grassy fuels, provides highly combustible conditions for the extensive passage of fire across this savanna landscape on an annual basis.

Cultural landscape

Currently, population densities outside major communities established in coastal/near-coastal areas of Arnhem Land are sparse. Maningrida, the major centre to the north of the study area, comprises a resident population of *c*. 1300 people. This town services 30 or so smaller family based

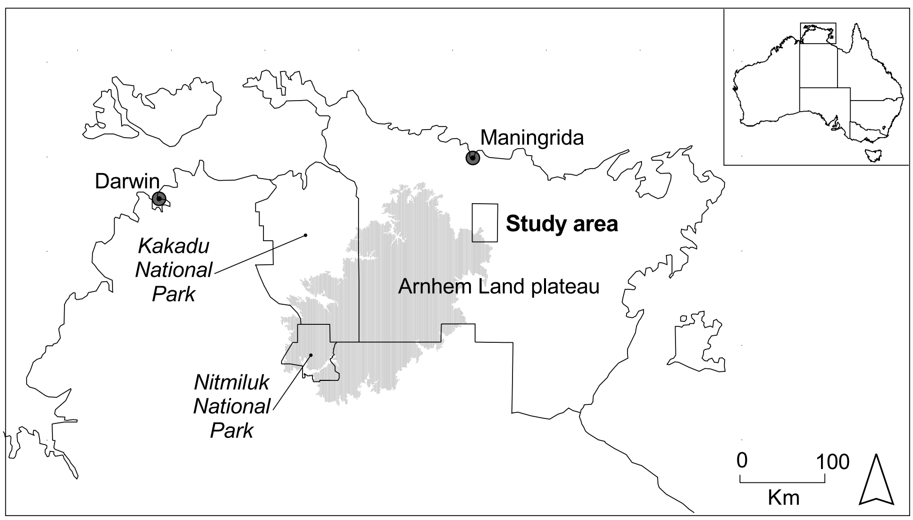

Figure 1 The location of the study site in the Northern Territory of Australia. Arnhem Land is entirely owned by Aboriginal people from a range of language groups, and is largely managed to support a traditional subsistence economy. Large settlements are few, and include the township of Maningrida (population 1300) approximately 70 km north of the Korlorbirrahda outstation.

communities, or outstations, scattered in an arc up to 100 km distant, particularly to the south and south-west.

There is a close correspondence between the average population size resident at outstations (*c*. 25 persons) and the band, an anthropological term describing the major land-using, as opposed to land-owning, group in traditional societies (Altman, 1987; Altman & Taylor, 1987). Although outstation groups may be as large as 100 persons, there is a tendency for such large groups to split and resettle in smaller residential units. The range utilized by the band consists of a number of traditional parcels of land (estates) over which members of the band have usufructuary rights. Estates may be defined loosely as aggregations of associated named sites, or by precise boundaries (e.g. elevation, vegetation boundaries and watercourses).

Interest in land is based on spiritual affiliations with certain sites, and with the ancestral beings who imbued those sites with particular spiritual meaning. People with such spiritual interests in land may be variously described as owners, managers or custodians. In the study area, owners refer usually to a group of people united by patrilineal descent who are associated with a particular estate. This group, known locally as the *kunmukurrkurr*, is referred to as a clan in anthropological literature. Clans are exogamous, as are the two moieties to which they are assigned. Complex relationships that emphasize both complementarity and separation in the nature and roles of moieties drive the social dynamic and define the roles and responsibilities of individuals in respect of land and its management. People with matrifilial links to a particular clan and estate have strong rights and responsibilities as managers, both for land and rituals. The term custodian is used broadly to denote such people with responsibilities for land, whether derived through patri- or matri-filiation. Thus a band, or a modern outstation group, is comprised of people with a variety of custodial responsibilities (including burning) for a number of estates.

This study focuses on one Gunei language-group clan estate, Dukaladjarranj, of *c*. 90 km^2 on the upper reaches of the Cadell River (Figs 1 and 2). Since the 1960s this estate, and a number of others adjacent, have been under the continuous custodianship of one extended family who, unlike many other clans (particularly to the south of the Cadell, in central Arnhem Land), have remained on their traditional lands throughout. Further, apart from a couple of years in the 1950s when most family members were resident elsewhere, there has been an unbroken tradition of fire management undertaken on Dukaladjarranj and other adjacent estates which extends to the present.

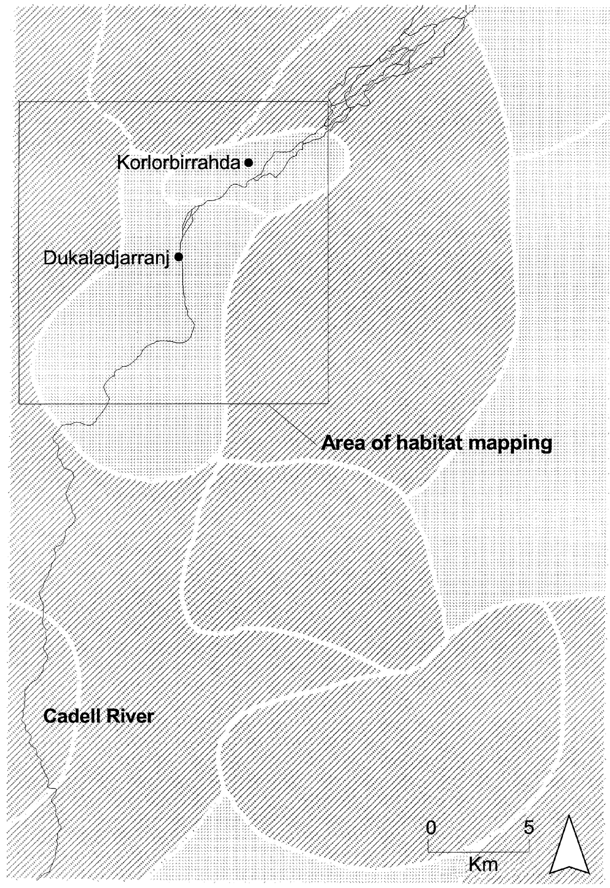

Figure 2 Map of approximate clan boundaries in the region and the area subject to air and ground-based survey. The boundaries are indicative only as some sites are shared by more than one clan.

Fire management

Traditional approaches to fire management of Dukaladjarranj over the seasonal cycle can be summarized as follows. Towards the start of the dry season family members, including relatives resident on other estates who have social/management responsibilities for Dukaladjarranj, discuss and plan for the year ahead. This includes identifying areas needing special treatment (e.g. spiritual sites, yam beds, preparations for macropod fire drives). From early in the dry season, but particularly in the cooler mid-year, winter season (*wurrgeng*), burning is undertaken purposefully and systematically over the estate as people move around.

Burning focuses initially on cured grasses on higher ground, progressively focusing on moister sites as these dry out with the developing dry season. Such fires are typically of low intensity and small in extent. The net result is a patchy mosaic of burned and unburned country, although certain habitats (e.g. creeklines and other favoured macropod hunting areas) are more targeted than others. Come the very hot, dry time towards the end of the dry season (*gurrung*), broadscale burning ceases and is replaced with more controlled applications (e.g. burning onto previously burnt areas). Such practice was evidently widespread in other sandstone regions of northern and western Arnhem Land (Russell-Smith *et al*., 1997a). However, one notable regional difference is/was that, in western Arnhem Land, burning resumed on floodplains with the onset of the first storms of the wet season (*gunumeleng*), whereas burning of floodplains to the north of the Cadell is not resumed at that time.

Conversely, most of central and western Arnhem Land today is unoccupied and thus unmanaged. As such, fires lit in the late dry season for whatever purpose may burn unchecked for months, over tens of thousands of square kilometres until the first rains of the wet season (e.g. Russell-Smith *et al.*, 1997b; Press, 1988). Such fires, fanned by south-easterly winds, regularly find their way onto the western rim of the Arnhem Plateau, and into Kakadu and Nitmiluk National Parks.

METHODS

Following discussions between the responsible government conservation agency (Parks and Wildlife Commission of the Northern Territory; PWCNT) and a major regional indigenous land management organization (Bawinanga Aboriginal Corporation) concerning current fire management problems associated with the vast Arnhem Plateau, it was agreed that a joint research/educational programmes involving ranger staff from both organizations, senior Aboriginal custodians and PWCNT ecologists, be held on lands which continue to be managed in a traditional manner, in order to make an assessment of the ecological value and effectiveness of such practice. As such a field camp was arranged over 10 days based at Dukaladjarranj on the upper Cadell River, focusing on fire management of the Dukaladjarranj estate, with the permission and support of senior custodians. The camp was held in early September, at the start of the traditional season of *gurrung*, by which time early dry season fire management would have been completed.

Ecological assessments included: (1) mapping of the resource base of the estate from both traditional and ecological perspectives; (2) aerial survey of the extent of burning, distribution of fire-sensitive *Callitris*, rock habitats, and a range of macropod and other fauna resources; (3) fauna inventory; (4) detailed ecological assessment of the status of fire-sensitive vegetation; and (5) empirical assessment of the intensities of observed fires. Specific field and analytic methods for these activities are outlined below. As well, ethnographic information concerning traditional fire management practice was documented in interviews with senior custodians; these data are referred to throughout this paper, where appropriate.

Geographic resource survey

A map of major vegetation formations in the study area was prepared by interpretation of aerial photographs (1 : 50,000 scale) (Fig. 3). This map was employed to design a stratified survey. Floristic variation was sampled within the major habitats with the study area. A total of 94 quadrats was placed during the course of field traverses. At each quadrat the percentage of the ground surface covered by rocks and grass was visually estimated and the slope angle and canopy height was measured with a clinometer. A tree, that for the

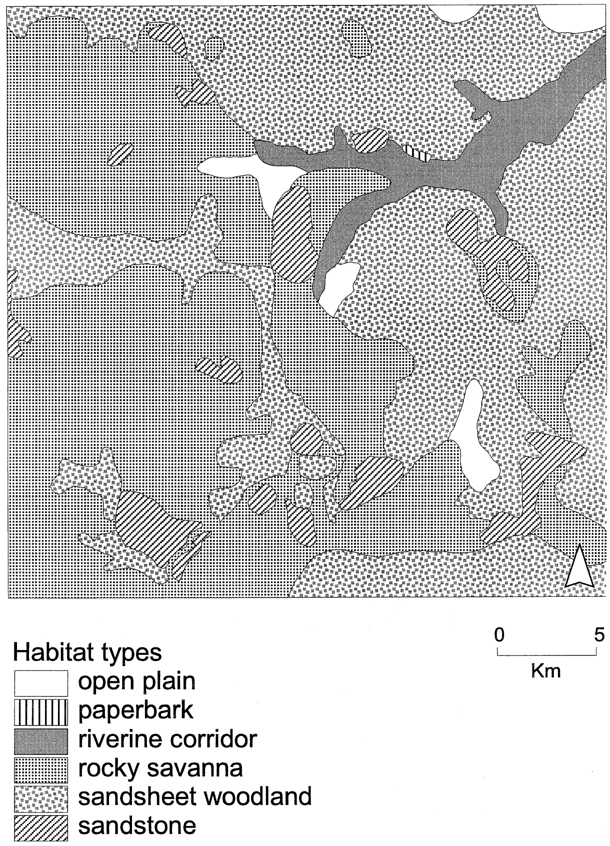

Figure 3 Map of the major habitat types in surveyed region shown in Figure 2. The map was prepared by on-site interpretation of aerial photographs involving both indigenous and non-indigenous participants.

purposes here shall be defined as a woody plant > 2 m tall, was arbitrarily selected to serve as the centre of a circular 'quadrat' of variable radius. The basal trunk diameter of that and the other nine such trees (including dead stems) closest to the central tree were measured with a pair of forestry callipers. The distance from the central tree to the most distant of the nine trees defined the radius of the circular quadrat. For unburned quadrats the same arbitrarily selected tree was used to serve as the centre of a circular 5 m quadrat in which the density of all woody plants less than 2 m tall were counted and the presence of all other plant species was noted. For sites burnt in the same year as sampling (1997) the average height of charred foliage and foliage killed by over-heating was determined with a clinometer for all trees within the quadrat.

The presence and absence of 57 woody species, for which basal area had been measured at least once in the 94 quadrats, was subjected to classification by TWINSPAN (Hill, 1979). This classification of quadrats clearly defined the following three habitats: river-flat, woodlands and sandstone. This habitat classification served as the basis for all subsequent analyses. The percentage frequency and mean basal area of the 57 woody species in the three habitats was determined and a sorted table was prepared. Using the data collected from all 94 quadrats the mean rock cover, slope angle, total basal area of living and dead stems in each quadrat and canopy height was determined for each habitat. For the 37 quadrats placed in recently burnt habitats the mean scorch height and char height was calculated for each habitat. For the remaining 57 unburned quadrats the mean cover of grass, richness of trees, shrubs and grass species was determined. The mean density of woody species < 2 m tall and the proportion of these that are obligate seeders (*sensu* Gill, 1981) were also determined for each of the three habitats. The Pearson correlation coefficient was determined between the proportion of obligate seeding woody plants and the rock cover and slope angle of the unburned quadrats.

Fauna survey

Vertebrate fauna within the study area was sampled using techniques commonly applied elsewhere in northern Australia (e.g. Woinarski & Fisher, 1995a, b). Mammals, birds, reptiles and frogs were censused within 0.25 ha quadrats over a 72-h period using a combination of visual counts, metal box (Elliott) traps, wire cage traps, pit buckets with drift fences, and diurnal and nocturnal searches. At least three quadrats were sampled in each of the three major habitats previously described. Three quadrats were also sampled within a 'riparian' habitat, a narrow zone bordering and including some major drainage lines, with a tall open or closed forest dominated by *Melaleuca* spp. (Fig. 3). Additional information on fauna was provided by: visual counts of birds during brief visits to areas of sandstone outcrop and open forest; observations during aerial counts of large mammals; incidental observations during travel between sites; capture of bats using harp traps; and capture of turtles using hoop-net traps.

For each species recorded in quadrats, counts (transformed as $\log_e(\text{count} + 1)$) within quadrats aggregated over the survey period were compared among different habitat types by ANOVA (SAS Procedure GLM: SAS Institute, 1989). The mean site richness of frogs, reptiles, mammals, birds and all vertebrates was also compared among habitat types by ANOVA.

Fire study

A study was conducted to determine the effect of fires of different intensities on woodland vegetation. A total of 14 plots was selected from two sites that contrasted in levels of grassy fuels. At each plot five 0.4×0.4 m quadrats were used to collect (i) living and dead grasses and (ii) leaf litter and twigs < 0.5 cm. The quadrats were placed every 5 m on a line transect. The grass and leaf litter material from each sample was placed in a uniquely numbered calico bag and subsequently oven dried (>100 °C) to constant weight to determine the dry mass of these two types of fuel.

During the course of 1 day fires were lit up-wind from the plots. The rate of spread was determined by direct measurement of slow moving fires or, in the case of fast moving fires, by triangulation following the methods of Williams *et al.* (1998). Prior to lighting the fires the humidity and air temperature was measured with a sling hygrometer and wind speed was estimated using the Balfour scale. Two days after the fire the height of char leaves and the height of canopy scorch was measured with a tape or clinometer at 10 randomly selected trees.

For each of the 14 plots the average total fuel mass (grass and litter combined) from the five quadrats was determined. The within-plot variation was determined by calculating the coefficient of variation around the sample mean. The percentage of the each combined fuel sample composed of grass was determined and the within plot variation was also determined by the calculation of the coefficient cent of variation. The overall mean and range of these four measures was determined for the 14 plots. The intensity of the fires was calculated by using the Byram (1959) equation assuming the energy content of fuel was 20,000 kJ kg^{-1}, as suggested by Gill & Knight (1991). The equation of Williams *et al.* (1998) between fire intensity char and scorch height were used to predict scorch and char heights. The linear relationships between predicted char and scorch and actual char and scorch heights, and meteorological variables and fuel mass variables, were determined by generating a matrix of correlation coefficients.

Aerial survey

Large animal survey

The total survey area was 15×15 km (225 km^2). The density of large kangaroos (antilopine, wallaroos, agile wallabies and euros), feral animals (buffalo and pigs) and emus was estimated from corrected helicopter counts, conducted at an average altitude of 30.5 m (100 ft), and an average airspeed of 80 km h^{-1}. Experienced observers in the rear left and right-hand seats of a Bell Jetranger helicopter counted groups of animals as they occurred in 150 m wide transects delimited by poles attached at right-angles to either side of the helicopter. Groups of animals counted represented sighting entities rather than biologically meaningful associations of animals. All counts were made onto a continuously running tape recorder and collated at the completion of each survey. For both surveys, east–west transects were placed at 500 m intervals across the site (two transects per 1 km swathe), to achieve a sampling rate of 60%. A Global Positioning System receiver was used for navigation and to monitor airspeed during all surveys, and a stop watch used to divide each transect into 0.5 km segments. This allowed observations to be aggregated into 0.25 km^2 cells for mapping.

Landscape assessment

At each segment break, the presence of large areas of slab rock, stands of *Callitris* pine and the degree of recent burning for the area immediately adjacent to the helicopter was scored, allowing these features to be mapped in the same

0.25 km^2 cells. The degree of burning was scored as nil (no evidence of recent burning), little burned (less than 20% of the non-rock area displaying evidence of having been burnt in the current dry season) and substantially burned (more than 20% of the non-rock area displaying evidence of having been burnt in the current dry season).

Estimates of animal density

Aerial counts of animal groups represent underestimates of the actual density of groups because not all groups present in a transect will be observed. We used published multiplicative correction factors to account for this bias in counts of buffalo (Bayliss & Yeomans, 1989) and pigs (Choquenot, 1995a). Because visibility bias associated with aerial counts of the macropod species encountered in this study has not been estimated, we assumed bias would be similar to that for eastern grey kangaroos and used correction factors derived for that species (Choquenot, 1995b). As such, density estimates for macropods reported in this study should be interpreted cautiously. Counts of emu have not been corrected for the effects of visibility bias, hence density estimates will represent substantial underestimates of true density. The density of each species was estimated as the average of corrected group density for each transect, multiplied by the average size of each group observed across the entire survey. Coefficients of variation around density estimates were calculated from the variance in density between transects. The tendency of macropods to occur on burnt country was evaluated using a chi-square test of association, with macropod presence assessed against the burning index for each 0.25 km^2 cell.

RESULTS

Geographic resource survey

The major habitats in the study region form a simple topographic sequence from major drainage lines up to sandstone ridges (Table 1, Fig. 3). Rock cover and ground surface slope angle increase across this topographic gradient, with a corresponding decrease in the basal area of woody stems (Table 2). Canopy height was similar between the river-flat and woodland habitats but about one-third to half as high for the sandstone habitats. Dead stems comprised less than 10% of the total basal area and showed no pattern across the topographic gradient.

The riverflat community was dominated by *Lophostemon lactifuus* (Muell.) Peterwilson & Waterhouse, *Eucalyptus alba* Reinw. ex Blume and *Erythrophleum chlorostachys* (Muell.) Baillon with substratum dominants including *Grevillea heliosperma* Br. and the screw-palm *Pandanus spiralis* Br. (Table 1). The woodland habitat was dominated by *Eucalyptus tetrodonta* Muell. and codominated, or was locally replaced by, a variety of other woody species. The abundance of codominant woodland plants is known to vary in response to local changes in edaphic factors such as soil rockiness (e.g. *Eucalyptus phoenicea* Muell.), wet season impeded drainage (*Melaleuca nervosa* (*Lindley*) *Cheel*) and sandy soils (e.g. *Eucalyptus arnhemensis* Carr & Carr) (Bowman & Minchin, 1987; Bowman *et al*., 1993).

None of the live basal area of trees in the river-flat community was made up of species whose regeneration strategy was classified as being obligate regeneration from seed following wildfire (Table 2). The greatest proportion of the live basal area comprising obligate seeder species was in the woodland, reflecting the high mean basal area of the native conifer *Callitris intratropica* (Table 1). Within both the woodland and sandstone habitats there was no correlation between slope angle or rock cover and the proportion of the live basal area composed of obligate seeders ($P > 0.05$).

All river-flat quadrats were burnt at the time of sampling so data on the ground layer could only be collected from the woodland and sandstone habitats. Cover of grass was approximately the same in woodland and sandstone habitats although grass species diversity was greater in the woodland (Table 3). Density of woody plants < 2 m tall was highest in the woodland but the proportion of obligate seeders was nearly three times higher in the sandstone habitats compared with the woodland. In the woodland the proportion of obligate seeders in the ground stratum was significantly correlated to ground slope-angle ($r = 0.38$, $P = 0.021$, $n = 42$), and in the sandstone habitat with rock cover ($r = 0.61$, $P = 0.012$, $n = 15$).

At the time of sampling about one-third of the woodland and sandstone quadrats had been burnt (Table 4). The scorch height of foliage was greatest in the woodland and least in sandstone habitats. The height of charred leaves (used here as a proxy for flame height) was similar in the river-flat and woodland habitats and lowest in the sandstone habitats. The empirical relationships between fire intensity and leaf and scorch height, using the Williams *et al*. (1998) equations for *Eucalyptus tetrodonta* savanna on laterite soils, are < 2500 kW m^{-1} across all communities, with maximum intensities less than 10,000 kW m^{-1}.

Fauna survey

A total of 155 vertebrate species was recorded during the study, comprising 83 birds, 11 frogs, 33 reptiles and 28 mammals (5 feral, 7 bat and 16 non-flying native mammals). Richness at the 14 intensively sampled sites ranged from 6 to 33 species. Mean site richness differed significantly among the four major habitats for a number of taxa (Tables 5 and 6). Riparian sites had higher species richness of frogs than other habitats, while mammal richness was greater in rocky (sandstone) sites (Tukeys HSD, $P < 0.05$). The high bird species richness in riparian sites contributes substantially to the high mean total richness for this habitat.

Assessment of total species richness for each habitat must be made cautiously as the total area of the habitats and hence sampling intensity varied among them. If this measure is restricted to species recorded within the quadrats (Table 6), then the riparian habitat had the highest number of native vertebrate species, made up predominantly of birds (66%).

Table 1 Percentage frequency of woody species having measurable basal area in 94 quadrats in the three habitat types defined by a TWINSPAN classification of the presence or absence of species with basal area. Only species with a frequency of 10% in at least one of the habitat types are listed. The mean basal area (m^2 ha^{-1}) of each species in each habitat type is listed in brackets. Mean basal area of less than 0.01 m^2 ha^{-1} are indicated by a ' +'. M

	River flats		Woodland		Sandstone	
Lophostemon lactifluus (Muell) Peterwilson & Waterhouse	69	(2.60)	–	(0)	–	(0)
Eucalyptus alba Reinw. ex Blume	69	(0.33)	–	(0)	–	(0)
Eucalyptus ptychocarpa Muell.	46	(0.55)	–	(0)	–	(0)
Eucalyptus clavigera Cunn. ex Schaver 38	(1.17)	–	(0)	–	–	(0)
Eucalyptus polycarpa Muell	38	(1.26)	–	(0)	–	(0)
Erythrophleum chlorostachys (Muell) Baillon	69	(0.33)	13	(0.09)	–	(0)
Melaleuca nervosa Cheel	62	(0)	5	(0.01)	–	(0)
Pandanus spiralis Br.	61	(0.45)	3	(+)	–	(0)
Buchanania obovata Engl.	38	(0.11)	35	(+)	–	(0)
Planchonia careya (Muell) Kunth	31	(0.01)	13	(0)	–	(0)
Syzygium suborbiculare (Benth.) Hartley & Parry	23	(0.03)	2	(0.01)	–	(0)
Alphitonia excelsa (Fenzl) Benth	23	(0)	8	(+)	–	(0)
Banksia dentata L.F.	15	(0.14)	–	(0)	–	(0)
Verticordia cunninghamii Schauer	15	(0)	10	(0.01)	–	(0)
Acacia aulacocarpa Cunn. ex. Benth.	8	(0.7)	20	(0.05)	–	(0)
Grevillea heliosperma Br.	62	(0.56)	32	(0.03)	14	(0.01)
Grevillea pteridifolia Knight	15	(0.15)	12	(0.01)	5	(0)
Eucalyptus tetrodonta Muell.	–	(0)	70	(2.03)	–	(0)
Persoonia falcata Br.	–	(0)	37	(0.02)	–	(0)
Callitris intratropica Baker & Smith	–	(0)	35	(2.20)	–	(0)
Acacia torulosa Benth. ex. Muell.	–	(0)	33	(0.05)	–	(0)
Calytrix exstipulata DC.	–	(0)	33	(0.03)	–	(0)
Eucalyptus miniata Cunn. ex Schauer	–	(0)	30	(1.66)	–	(0)
Stenocarpus acacioides Muell.	–	(0)	17	(0.01)	–	(0)
Exocarpos latifolius Br.	–	(0)	15	(+)	–	(0)
Owenia vernicosa Meull.	–	(0)	15	(+)	–	(0)
Petalostigma pubescens Domin	–	(0)	13	(+)	–	(0)
Gardenia megasperma Muell	–	(0)	12	(+)	–	(0)
Planchonella arnhemica (Muell.) Royel	–	(0)	12	(0.01)	–	(0)
Denhamia obscura (Rich.) Walp.	–	(0)	10	(0.01)	–	(0)
Acacia platycarpa Muell.	–	(0)	40	(+)	10	(0.02)
Eucalyptus arnhemensis Carr & Carr	–	(0)	35	(0.38)	33	(0.59)
Acacia latescens Benth.	–	(0)	33	(0.01)	5	(0.02)
Jacksonia dilatata Benth.	–	(0)	28	(0.02)	10	(0)
Eucalyptus phoenicea Muell.	–	(0)	27	(0.86)	24	(0.36)
Xanthostemon paradoxus Muell.	–	(0)	28	(0.20)	10	(0.12)
Terminalia carpentariae White	–	(0)	27	(0.08)	10	(+)
Thryptomene remota Bean	–	(0)	5	(0.03)	95	(2.18)
Gardenia fucata Br. ex Benth.		(0)	3	(+)	52	(0.37)
Acacia multisiliqua (Benth.) Maconochie	–	(0)	2	(0)	14	(+)
Acacia sp	–	(0)	–	(0)	52	(0.30)
Grevillea formosa McGillivray	–	(0)	–	(0)	48	(0.07)
Calytrix decussata Craven	–	(0)	–	(0)	29	(0)

A number of species showed a high degree of fidelity to a particular habitat (Table 5), the majority of which were confined to rocky and riparian habitats. Few species were mostly restricted to woodlands on sandy soils, although this was spatially the dominant habitat type. Species with high fidelity to river flats were mostly those occurring on dense grasses and sedges in run-on areas (Golden Headed Cisticola, Red-chested Button-quail, Tawny Grass-bird) or also occurred in adjacent riparian zones (the rodent *Melomys burtoni* Ramsey, and scincid lizard *Carlia gracilis* Storr).

A number of species recorded during the study could be considered notable because they are at or beyond the previously known distribution for the species; have a geographically restricted distribution; have a threatened status or are known to have undergone population decline; or are infrequently recorded. The majority of 'notable' species occurred only in the rocky habitat and include

Table 2 Mean and range (in brackets) of surface rock cover, slope angle and four characteristics of the tree layer in the 94 plots sampled by the geographical survey

Variable	River flat (n = 13)	Woodland (n = 60)	Sandstone (n = 21)
Rock Cover (%)	0	16 (0–100)	74 (0–100)
Slope angle (°)	0.5 (0–2)	2 (0–5)	6 (1–15)
Canopy height (m)	16 (6–20)	17 (9–21)	6 (0–15)
Basal area of living stems (m^2 ha^{-1})	10.2 (2.3–27.0)	8.2 (1.2–30.4)	4.3 (0.8–10.8)
Basal area of dead stems (m^2 ha^{-1})	0.4 (0–4.2)	0.6 (0–9.2)	0.1 (0–0.5)
Live basal area obligate seeders (%)	0	20 (0–100)	4 (0–56)

Table 3 Mean and range (in brackets) of six vegetation characteristics of unburned quadrats in the woodland and sandstone habitats. No river flat habitats were unburned at the time of sampling

Variable	Woodland	Sandstone
Grass cover (%)	17 (1–50)	20 (5–60)
Tree species richness	7 (1–13)	1 (0–4)
Shrub species richness	8 (3–17)	8 (5–16)
Grass species richness	5 (0–9)	3 (1–5)
Total density of woody plants < 2 m tall (n ha^{-1})	3660 (380–6620)	2070 (630–3800)
Woody plants < 2 m that are obligate seeders (%)	21 (0–87)	59 (0–100)

species restricted to rugged sandstone ranges in north-western Australia (e.g. *Pseudantechinus bilarni* Johnson, *Meliphaga albilineata* White, *Petrophassa rufipennis* Collett) or more specifically to the sandstone plateau of western Arnhem Land (*Morelia oenpelliensis* Gow, *Ctenotus coggeri* Sadlier, *Gehyra pamela* King, *Litoria personata* Tyler, Davies & Marun, *Pseudothecadactylus lindneri* Cogger, *Zyzomys maini* Kitchner, *Macropus bernardus* Rothschild, *Amytornis woodwardi* Hartert). The large number of species that reach the eastern-most limit of their known distribution at this site reflect both the location on the eastern margin of this plateau and the limited biological survey in Arnhem Land to date. The Partridge Pigeon (*Geophaps smithi* Jardine & Selby), which is considered vulnerable (Garnett, 1992) was only observed in open forest. The Northern Quoll (*Dasyurus hallucatus* Gould), which has declined through much of its range (Braithwaite & Griffiths, 1994) occurred here in all major habitats at relatively high abundances.

Fire Study

The woodland had a mean fine fuel load of about 4 t ha^{-1}, of which 10% was made up of grass biomass (Table 7). Relative to the coefficient of variation in total fuel loads within plots, the proportion of grass fuel is highly variable within plots with an average coefficient of variation of 72%. All the fires, which were lit at various times through the day at the end of the dry season, occurred under relatively humid, hot conditions with light winds (Table 8). The intensity of the fires varied by two orders of magnitude ranging from 10 to 1500 kW m^{-1} with an average intensity of about 300 k Wm^{-1}. The correlation analysis suggested that fire intensity was significantly correlated with average char height but not average scorch height (Table 9). Scorch height was significantly correlated with the average proportion of grass fuel on each plot. The relationships between char and scorch height and estimated fire intensities are shown in Figs 4 and 5.

Compared with *Eucalyptus tetrodonta* savanna on laterite the proportion of grass fuels is much lower than the average late dry season value of 41% reported by Williams *et al.* (1998) from the Kapalga fire experiment and the 48.2% for woodland sites reported by Russell-Smith *et al.* (1998). The average total fuel mass of 4 t ha^{-1} is lower than the late dry season average of 5 t ha^{-1} reported by Williams *et al.* (1998) at woodland sites and 5.9 t ha^{-1} (Russell-Smith *et al.*, 1998) in the Kakadu escarpment. It should be noted that the higher grass biomass at Kapalga is not simply a product of the slightly higher soil fertility of laterite soils compared with the siliceous soils at the Cadell River. Rather, it reflects the development of a dense sward of the annual tall grass *Sorghum intrans* Muell. ex Benth in response to a very high frequency of late dry season fires. Recurrent high intensity fires in *Eucalyptus tetrodonta* savannas on siliceous soils can also result in swards of annual *Sorghum* spp. (Russell-Smith *et al.*, 1998).

The lower mass of grass fuel may account for the significantly lower fire intensities reported here compared with the average late dry season fire intensities of 7700 kW m^{-1} measured by Williams *et al.* (1998) at Kapalga. Indeed, the fire intensities reported here are much lower than the average early dry season fires intensities of 2100 kW m^{-1} recorded at Kapalga.

Table 4 Mean and range (in brackets) of scorched and char heights measured in burnt quadrats within each vegetation type. The number of burnt quadrats and the percentage (in brackets) of the total number of quadrats in each vegetation type is also shown

Variable	River flat	Woodland	Sandstone
Scorch height (m)	3.7 (1.0–8.0)	7.9 (1.0–20)	1.4 (0.5–2.7)
Char height (m)	0.7 (0.1–1.2)	0.8 (0.1–3.7)	0.1 (0.1–0.4)
Number of burnt quadrats and percentage of total	13 (100%)	18 (30%)	6 (29%)

Table 5 Fauna identified in survey quadrats in the different habitat types. Observations made in the dense riparian fringe have been separated from other observations on river flats. Numbers under habitat types are the frequency of encounter (number of sites in which the species was recorded at least once during the survey period) and in parentheses the mean abundance (mean counts per site over the survey)

Species (status code)	Binomial & authority	Habitats: Riparian fringe	River flats	Woodland	Sandstone	All habitats
Birds	**Birds**					
Banded Honeyeater	*Certhionyx pectoralis* Gould	2 (6.3)	2 (37.7)	2 (1.2)	2 (7.3)	8 (11.4)
Bar-breasted Honeyeater	*Ramsayornis fasciatus* Gould	2 (8.0)	0	1 (0.2)	0	3 (1.8)
Bar-shouldered Dove*	*Geopelia humeralis* Temminck	3 (2.0)	0	0	0	3 (0.4)
Barking Owl	*Ninox connivens* Latham	1 (0.7)	0	0	0	1 (0.1)
Black-faced Cuckoo-shrike	*Coracina novaehollandie* Gemlin	0	0	1 (0.4)	1 (0.3)	2 (0.2)
Blue-faced Honeyeater	*Entomyzon cyanotis* Latham	0	0	0	1 (0.3)	1 (0.1)
Blue-winged Kookaburra	*Dacelo leachii* Vigors & Horsfield	1 (0.3)	1 (0.3)	0	0	2 (0.1)
Brown Honeyeater	*Lichmera indistincta* Vigors & Horsfield	3 (17.0)	2 (3.0)	4 (9.2)	3 (6.0)	12 (8.9)
Chestnut-quilled Rock-Pigeon*	*Petrophassa rufipennis* Collett	0	0	0	2 (0.7)	2 (0.1)
Crimson Finch*	*Neochmia phaetom* Hombron & Jacquinot	3 (17.0)	0	0	0	3 (3.6)
Double-barred Finch	*Taenipygia bichenovii* Vigors & Horsfield	3 (4.3)	0	1 (0.4)	1 (3.0)	5 (1.7)
Dusky Honeyeater	*Myzomela obscura* Gould	3 (4.0)	1 (1.0)	2 (1.2)	1 (0.7)	7 (1.4)
Forest Kingfisher*	*Todiramphus macleayii* Jardine & Selby	2 (2.3)	0	0	0	2 (0.5)
Golden-Headed Cisticola	*Cisticola exilis* Vigors & Horsfield	0	1 (4.0)	0	0	1 (0.9)
Great Bowerbird	*Chlamydera cerviniventris* Jardine & Selby	0	0	2 (0.4)	1 (0.3)	3 (0.2)
Grey Butcherbird	*Cracticus torquatus* Latham	0	0	1 (0.2)	0	1 (0.1)
Helmeted Friarbird	*Philemon buceroides* Swainson	0	1 (0.3)	3 (3.2)	2 (1.0)	6 (1.4)
Leaden Flycatcher	*Myiagra rubecula* Latham	1 (0.3)	0	1 (0.2)	0	2 (0.1)
Lemon-bellied Flycatcher	*Microeca flavigaster* Gould	2 (2.3)	1 (0.3)	0	0	3 (0.6)
Little Friarbird	*Philemon citreogularis* Gould	1 (0.3)	0	2 (0.6)	1 (0.3)	4 (0.4)
Mistletoebird	*Dicaeum hirundinaceum* Shaw	2 (4.3)	1 (0.3)	2 (0.6)	1 (0.3)	6 (1.3)
Northern Fantail	*Rhipidura rufiventris* Vieillot	3 (3.3)	1 (2.0)	2 (0.8)	0	6 (1.4)
Olive-backed Oriole	*Oriolus sagittatus* Latham	0	0	1 (0.2)	0	1 (0.1)
Peaceful Dove	*Geopelia striata* Linnaeus	2 (11.3)	1 (2.0)	1 (0.6)	1 (2.7)	5 (3.6)
Pheasant Coucal	*Centropus phasianinus* Latham	0	1 (0.3)	0	0	1 (0.1)
Rainbow Bee-eater	*Merops ornatus* Latham	1 (0.7)	1 (0.3)	2 (0.6)	2 (1.0)	6 (0.6)
Red-backed Wren	*Malurus melanocephalus* Latham	3 (4.7)	2 (4.3)	1 (0.6)	0	6 (2.1)
Rainbow Lorikeet	*Trichoglossus haematodus* Linnaeus	2 (1.3)	0	1 (1.0)	0	3 (0.6)
Red-chested Button-quail	*Turnix pyrrhothorax Gould*	0	1 (0.7)	0	0	1 (0.1)
Red-tailed Black Cockatoo	*Calyptorhynchus banskii* Latham	0	1 (0.7)	0	0	1 (0.1)
Red-winged Parrot	*Aprosmictus erythropteurs* Gmelin	0	0	2 (1.2)	0	2 (0.4)
Nankeen Night-Heron	*Nycticorax caladonicus* Gmelin	1 (0.3)	0	0	0	1 (0.1)
Rufous-throated Honeyeater	*Conopophila rufogularis* Gould	2 (1.7)	0	1 (0.4)	0	3 (0.5)
Rufous Whistler	*Pachycephala rufiventris* Latham	1 (1.0)	0	2 (0.4)	0	3 (0.4)
Sandstone Shrikethrush	*Colluricincla woodwardi* Hartert1 (0.3)	0	0	1 (0.3)	2 (0.1)	
Spangled Drongo	*Dicrurus bracteatus* Gould	1 (0.3)	0	0	0	1 (0.1)
Striated Pardalote	*Pardalotus striatus* Gmelin	2 (0.7)	1 (1.0)	1 (0.6)	1 (0.3)	5 (0.6)
Tawny Grassbird	*Megalurus timoriensis* Wallace	0	1 (0.3)	0	0	1 (0.1)
Varied Lorikeet	*Psitteuteles versicolor* Lear	0	1 (5.0)	0	0	1 (1.1)
Variegated Fairy-Wren	*Malurus lamberti* Vigors & Horsfield	0	0	0	1 (1.7)	1 (0.4)
Weebill*	*Smicrornis brevirostris* Gould	0	0	4 (2.4)	0	4 (0.9)
Whistling Kite	*Haliastur sphenurus* Vieillot	1 (0.3)	2 (0.7)	0	0	3 (0.3)
White-bellied Cuckoo-shrike	*Coracina papuensis* Gmelin	1 (0.3)	1 (0.3)	0	1 (0.3)	3 (0.2)
White-gaped Honeyeater*	*Lichenostomus unicolor* Gould	3 (4.3)	1 (0.3)	1 (0.4)	0	5 (4.7)
White-throated Gerygone	*Gerygone olivacea* Gould	0	0	1 (0.2)	0	1 (0.1)
White-throated Honeyeater	*Melithreptus albogularis* Gould	3 (10.3)	2 (7.3)	2 (2.4)	1 (0.3)	8 (4.7)
White-winged Triller	*Lalage sueurii* Vieillot	1 (0.3)	0	2 (1.0)	0	3 (0.4)
Willie Wagtail	*Rhipidura leucophrys* Latham	1 (1.0)	0	0	1 (0.3)	2 (0.3)
Yellow Oriole	*Oriolus flavocinctus* Vigors	0	1 (0.3)	0	0	1 (0.1)
	Amphibians (frogs)					
	*Limnodynastes convexiusculus** Macleay	0	2 (0.7)	0	0	2 (0.1)
	Limnodynastes ornatus Gray	0	1 (0.3)	0	0	1 (0.1)
	Litoria bicolor Gray	0	1 (0.3)	0	0	1 (0.1)

Table 5 (*Continued*)

Species (status code)	Binomial & authority	Habitats: Riparian fringe	River flats	Woodland	Sandstone	All habitats
	Litoria coplandi Tyler	1 (0.7)	0	0	0	1 (0.1)
	Litoria inermis Peters	0	1 (0.3)	0	0	1 (0.1)
	Litoria meiriana Tyler	1 (16.7)	0	0	0	1 (3.6)
	Litoria nasuta Gray	2 (1.0)	2 (1.0)	0	0	4 (0.4)
	Litoria personata Tyler	1 (0.3)	0	0	0	1 (0.1)
	*Litoria tornieri** Nieden	2 (1.0)	0	0	0	2 (0.2)
	*Litoria wotjulmensis** Copeland	3 (4.7)	0	0	0	3 (1.0)
	*Ranidella bilingua** Crinia	2 (1.0)	0	0	0	2 (0.2)
	Reptiles					
	Boiga irregularis Marrem	0	1 (0.3)	0	0	1 (0.1)
	*Carlia amax** Storr	0	0	5 (8.6)	3 (12.3)	8 (5.7)
	*Carlia rufilatus** Storr	2 (1.7)	3 (12.7)	0	0	5 (3.1)
	*Cryptoblepharus plagiocephalus** Cocteau	0	3 (3.0)	2 (0.6)	0	5 (0.9)
	Ctenotus essingtoni Gray	0	0	1 (0.2)	0	1 (0.1)
	Ctenotus inornatus Gray	0	0	0	1 (0.3)	1 (0.1)
	Ctenotus saxatilis Storr	0	0	0	1 (0.3)	1 (0.1)
	Ctenotus vertebralis Rankin & Gillian	1 (0.3)	0	2 (0.6)	0	3 (0.3)
	Diporiphora bilineata Gray	0	1 (0.3)	1 (0.2)	0	2 (0.1)
	*Gehyra australis** Gray	1 (0.3)	0	0	3 (2.7)	4 (0.6)
	*Gehyra nana** Gray	0	0	0	2 (0.7)	2 (0.1)
	Heteronotia binoei Gray	0	0	2 (0.4)	0	2 (0.1)
	Lophognathus gilberti Gray	2 (1.0)	1 (0.3)	2 (0.6)	0	5 (0.5)
	Menetia greyi Gray	0	1 (0.7)	0	0	1 (0.1)
	Morethia ruficauda Lucas & Frost	0	0	0	1 (0.3)	1 (0.1)
	Notoscincus ornatus Broom	0	0	3 (0.8)	0	3 (0.3)
	Oedura marmorata Gray	0	0	0	1 (0.3)	1 (0.1)
	Proablephorus tenuis Broom	0	0	1 (0.3)	0	1 (0.1)
	*Pseudothecadactylus lindneri** Cogger	0	0	0	2 (1.0)	2 (0.2)
	Tiliqua scincoides White	0	0	1 (2.0)	0	1 (0.7)
	Tropidonophis mairii Gray	1 (0.3)	1 (0.3)	0	0	2 (0.1)
	Varanus panoptes Storr	0	1 (0.3)	0	0	1 (0.1)
	Mammals					
	Dasyurus hallucatus Gould	2 (2,1)	1 (1)	3 (4, 2, 8)	2 (1, 1)	8
	Macropus agilis Gould	0	1 (2)	0	0	1
	Macropus bernadus Rothschild	0	0	0	2 (S)	2
	Macropus robustus Gould	0	0	0	1 (S)	1
	*Melomys burtoni** Ramsay	1 (3)	3 (9, 3, 1)	0	0	4
	Petaurus breviceps Waterhouse	0	0	1 (2)	0	1
	Petrogale brachyotis Gould	0	0	0	3 (S)	3
	Pseudantechinus bilarni Johnson	0	0	0	1 (2)	1
	Pseudocheirus dahli Collett	0	0	0	1 (S)	1
	Pseudomys delicatulus Gould	1 (2)	0	1 (1)	0	2
	Pteropus scapulatus Peters	1 (2)	0	1 (1)	1 (4)	3
	Tachyglossus aculeatus Shaw	0	0	0	2 (S)	2
	Exotic Mammals					
	Bubalus bubalis Linnaeus	3 (S)	2 (S)	2 (S)	0	7
	Felis catus Linnaeus	1 (S)	0	0	0	1

An S in parentheses indicates that the presence of the species was determined by sign or spoor rather than capture. Species for which recording rates varied among habitat types are indicated by *($P < 0.05$). A number of additional species were recorded outside quadrats and are referred to at various places in the text.

Aerial survey

Of the 900 survey cells, 575 (63.9%) had been at least partly burned during the dry season of survey. Population estimates for large mammals and emus are summarized in Table 10. Macropod groups were sighted in 84 cells, 71 (84%) of which had been burned during the current year. The association of macropods with the degree of burning in cells

Table 6 Mean and (total) native vertebrate species richness recorded in the comprehensively sampled quadrats within each habitat type. Additional species were recorded outside the quadrats (see text)

	Habitats			
Taxon	Riparian fringe (n = 3)	River flats (n = 3)	Woodland (n = 5)	Sandstone (n = 3)
Frogs	4.0* (7)	2.3* (5)	0 (0)	0 (0)
Reptiles	2.3 (5)	4.0 (8)	4.0 (10)	4.7 (8)
Mammals	1.7 (4)	1.7 (3)	1.2 (4)	4.3* (8)
Birds	19.3 (31)	9.3 (23)	9.2 (27)	8.3 (19)
All vertebrates	27.3 (47)	17.3 (39)	14.4 (41)	17.3 (35)

Asterisks indicate that the mean richness significant exceeded ($P < 0.05$) observations for at least one other habitat. Despite markedly greater mean diversity of birds in the riparian fringe, variance among quadrats was large and the difference was not statistically significant ($P = 0.18$).

Table 7 Mean and range (in brackets) of fuel characteristics on 14 plots that were burnt in the late dry season at Maningrida

Variable	Mean	Range
Total fuel load (mg ha^{-1})	4.3	2.9–6.2
Coefficient of variation of total fuel load (%)	27	12–56
Moisture levels in fuels (%)	8.0	3.2–15.8
Coefficient of variation of moisture levels	39	Missing
Grass fuel (%)	10.2	1.1–42.3
Coefficient of variation of proportion of grass fuel (%)	72	11–166

Table 8 Mean and range of weather and fire characteristics for 11 of the 14 plots that were burnt in the late dry season at Maningrida

Variable	Mean	Range
Relative humidity (%)	56	40–78
Air temperature (°C)	31.5	26–37
Wind speed (knots)	3	0–5
Fire intensity (kW m^{-1})	300	10–1490
Char height (m)	0.4	0.2–0.8
Scorch height (m)	4.2	1.4–12.8

was highly significant ($\chi^2 = 16.22$, d.f. $= 2$, $P = 0.0003$), suggesting macropods were aggregating on recently burned areas. Variation in mean counts of groups present in cells with different recent fire histories is shown in Fig. 6.

Stands of *Callitris intratropica* were observed in 27.2% of the survey cells, and of these stands, 86.1% were found in cells which were scored as non-rocky. Dense *Callitris* stands are common in this landscape and, in contrast with the situation in many other areas, persist in areas with limited topographic protection (as evidenced by the observation that 72.2% of the stands were in cells that had been at least partially burned during the current year). There was no evidence of high densities of dead *Callitris* stems that characterize many other landscapes in the region (Bowman & Panton, 1993).

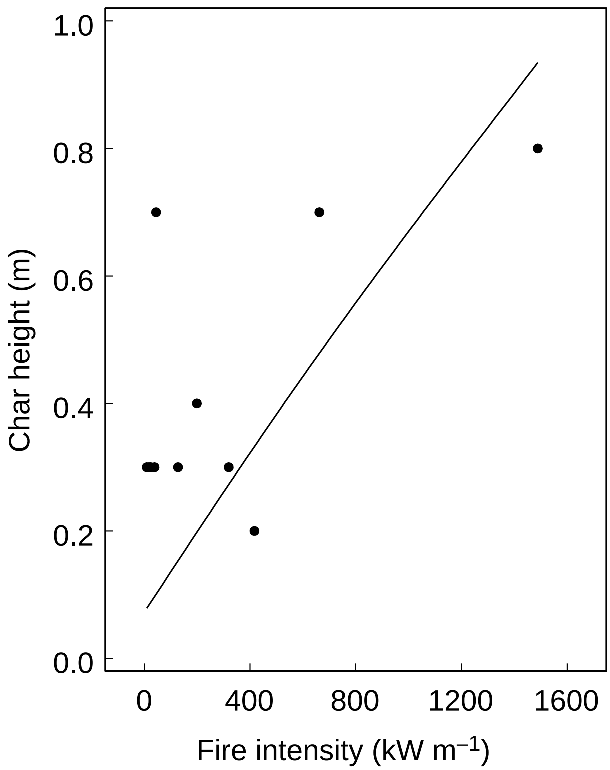

Figure 4 Relationship between char height and estimated fire intensity. Measures are shown for each fire plot (dots) and the line shows predictions from the equation of Williams *et al.* (1988).

DISCUSSION

Fire management in Australia's Top End

The vast sandstone plateau of Arnhem Land supports an internationally significant centre of plant diversity (Ingwersen, 1995), and its unique endemic fauna (Press, 1995) contributed to the World Heritage listing of the adjoining Kakadu National Park. The area's status is, however, threatened by huge wildfires (Russell-Smith *et al.*, 1997b, 1998), which sweep into the escarpment region driven by prevailing dry season winds from the south-east. Fires apparently originating from a single ignition may affect huge areas (> 1 million ha^{-1}), including a substantial proportion of the escarpment (Jacklyn & Russell-Smith, 1998).

The incidence of such wildfires, especially under late dry season conditions that exacerbate their effects (Williams *et al.*, 1998), is thought to have increased because Aboriginal people were displaced from their lands or aggregated voluntarily in larger settlements (Levitus, 1995; Russell-Smith *et al.*, 1997a). Lowland tall-grass savannas, mostly

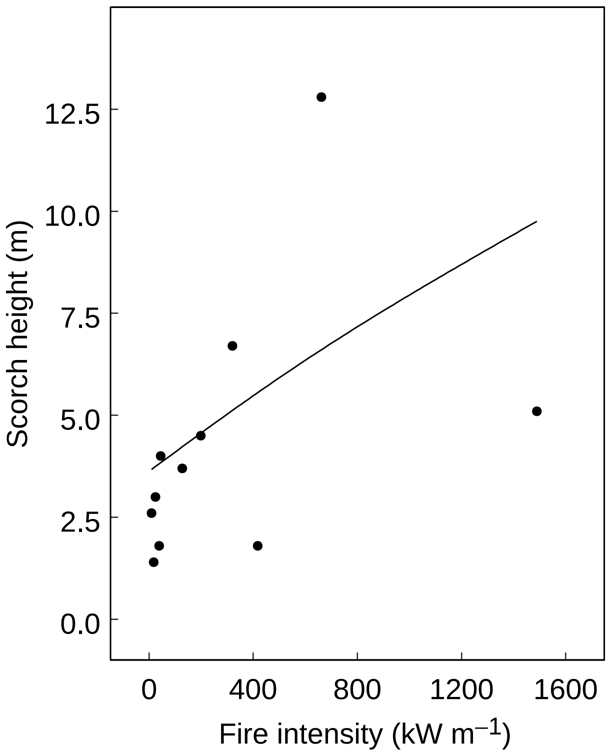

Figure 5 Relationship between scorch height and estimated fire intensity. Measures are shown for each fire plot (dots) and the line shows predictions from the equation of Williams *et al.* (1998).

dominated by native annual *Sorghum* (Wilson *et al.*, 1990) annually accumulate substantial fuel loads (averaging 5 t ha^{-1}; Williams *et al.*, 1998) capable of carrying fire over extensive distances and hence into the escarpment and these sensitive vegetation patches.

Within the sandstone and associated sand sheet environments, fuel loads capable of sustaining fires that will kill suites of fire sensitive woody plants accumulate 1–3 years after the last fire, a period much shorter than is required for regeneration of many obligate seeders, which require protection from fire throughout the period between seedling establishment and attainment of reproductive age (Russell-Smith *et al.*, 1998). Given ignitions by lightning (Bowman *et al.*, 1988) and frequent entry of fires deliberately lit outside the escarpment, large areas now experience late dry season fires at intervals shorter than the time required for re-establishment of these populations. Even in wetter, less exposed sites within the escarpment, similar processes are also damaging monsoon vine forests and thickets (Russell-Smith & Bowman, 1992). In the absence of renewed human intervention, frequent large and high intensity wildfires threaten widespread change in the status of the vegetation and its dependent biota with little prospect of recovery (Turner *et al.*, 1993).

The situation at Dukaladjarranj, where Aboriginal occupancy has been close to constant, contrasts strikingly with this regional picture. Annual *Sorghum* was all but absent; the traditional owners here regard the presence of dense *Sorghum* as evidence of poor fire management. Despite little grass (10%), total fuel loads (4 t ha^{-1}) were roughly comparable with other savannas in the region and of the order that can produce intense fires under late dry season climatic conditions (Williams *et al.*, 1998). Our experimental fires lit during early September were, nonetheless, of remarkably low intensity. Low intensities were not caused by elevated moisture content of fuels or aberrant weather conditions, all of which fell within the range producing very much more intense fires in Williams *et al.*'s (1998) experiments. Rather we attribute the difference to the physical configuration of fuel components (closely packed leaf litter) which constrained rates of combustion. Consistent with this interpretation, repeated ignitions were often required to achieve burns over the target area, and at the conclusion of experiments, fires most often quickly ceased to burn without further intervention. These observations illustrate the low risk of extensive wildfire in this landscape, and justified the landowner's confidence that the experiments could be conducted without undue risk in a season generally regarded as presenting high risk of damage and problems with containment (Williams *et al.*, 1998; Yibarbuk, 1998).

Moreover, the directly observable impacts of those fires on vegetation as indicated by leaf char height and leaf scorch height were comparable with the lowest measures made by Williams *et al.* (1998). We never recorded leaf char more than 0.7 m from ground level, and leaf scorch was only once recorded above 7 m. It is perhaps relevant to note that char heights were mostly slightly higher than predicted from the equations of Williams *et al.* (1998), while scorch heights were mostly lower (Figs 4 and 5). These patterns are consistent with the suggestion that the litter-based fuels that characterize this site tend to produce slower-burning, patchy fires capable of charring low vegetation but rarely producing large flames that carry into the upper canopy. This interpretation is also supported by the high incidence of healthy stands of *Callitris intratropica* in areas that had been recently burned, indicating that their persistence and good condition was not because of topographic protection from fire, but rather prevailing low fire intensity and spatial patchiness.

There was other evidence of strong congruence between the land owners' stated fire management goals and the outcomes we observed. Recurring themes in Aboriginal descriptions of their practice are the obligation to 'clean up' and renew the country. This is carried out to facilitate movement through the landscape and access to game, and to improve conditions for favoured species such as kangaroos. In addition, by burning at appropriate times and places, Aboriginal managers seek to protect fire sensitive sites such as jungles (Russell-Smith *et al.*, 1997a; Yibarbuk, 1998). In the study area, low grass biomass certainly facilitated movement through the landscape. We made no measures that might index the detectability of game. There have been no systematic aerial surveys of kangaroo abundance in the Northern Territory which would permit direct comparisons at a regional scale, but local densities were comparable

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Table 9 Correlation matrix between fuel characteristics, meteorological variables and fire characteristics measured on plots burnt in the late dry season at Maningrida

	Mean total fuel load	CV total fuel load	Grass fuel (%)	CV grass fuel	Fire intensity	Actual char height	Predicted char height	Actual scorch height	Predicted scorch height	Relative humidity	Air temperature
CV Total fuel load	0.181 (14)	–									
% grass fuel	−0.419 (14)	−0.021 (14)	–								
CV grass fuel	0.288 (14)	−0.259 (14)	−0.463 (14)	–							
Fire intensity	−0.282 (11)	−0.159 (11)	0.472 (11)	−0.482 (11)	–						
Actual char height	−0.324 (13)	−0.238 (13)	0.214 (13)	−0.032 (13)	**0.657*** (13)	–					
Predicted char height	−0.348 (11)	−0.178 (11)	0.532 (11)	−0.505 (11)	**0.996***** (11)	**0.656*** (11)	–				
Actual scorch height	−0.505 (13)	−0.287 (13)	**0.672*** (13)	−0.192 (13)	0.436 (13)	**0.587*** (13)	0.498 (13)	–			
Predicted scorch height	−0.303 (11)	−0.170 (11)	0.491 (11)	−0.481 (11)	**0.998***** (11)	**0.644*** (11)	**0.996***** (11)	0.464 (11)	–		
Relative humidity	0.354 (8)	−0.093 (8)	−0.543 (8)	0.100 (8)	−0.525 (8)	−0.243 (8)	−0.526 (8)	−0.536 (8)	−0.554 (8)	–	
Air temperature	−0.515 (8)	0.051 (8)	0.599 (8)	−0.133 (8)	0.522 (8)	0.395 (8)	0.524 (8)	0.683 (8)	0.546 (8)	**−0.966***** (8)	–
Wind speed	−0.240 (8)	0.149 (8)	0.610 (8)	−0.473 (8)	0.506 (8)	−0.079 (8)	0.516 (8)	0.253 (8)	0.527 (8)	**−0.780*** (8)	0.702 (8)

Correlation coefficients that are significant are in bold and the level of significance is indicated by asterisks where *< 0.05, ***, 0.001. The number of samples (n) is shown in brackets.

Table 10 Population and density estimates for large native and feral animals in the survey area. The total density of large macropods was 9.3 animals km^2

Species	Population	Density	CV%
Euro	1110	4.93	3.64
Antilopine	448	1.99	25.55
Agile wallaby	545	2.42	20.20
Buffalo	593	2.64	13.10
Pig	142	0.63	28.45
Emu	67	0.30	24.70

with those reported by Croft (1987) on more fertile soils elsewhere in the Top End, and regional Wallaroo densities were higher than reported by Southwell *et al.* (1997) for north-east Queensland. The relatively lower grass biomass is explicable chiefly in the terms of the absence of *Sorghum*, and it would appear that abundance of palatable forage species is sufficient to sustain healthy populations of large herbivores, which aggregated on recently burned areas. Monsoon vine-forests and thickets showed no evidence of recent boundary retreat, and favoured human foods (yams) remained abundant and were readily located by landholders during the study. Given the extended period of human occupation of these landscapes (> 50,000 years; Roberts *et al.*, 1990), and the centrality of fire use to land management practice, traditional fire management can be understood as an important tool for achieving long-term, sustainable relationships with the landscape and its biota.

Fire use and ecosystem management

Construction and maintenance of long-term sustainable relationships with land are principal goals of ecosystem management. Proponents of ecosystem management characterize the concept as requiring a shift in focus from optimal management of individual natural resources to an emphasis on the management of the ecological processes or ecosystem functions that sustain those resources (e.g. Christensen *et al.*, 1996). But there remains much difficulty in operationalizing the concept to produce widely shared visions of appropriately managed ecosystems, largely because different interest groups vary in their view of how much social and economic goals should be permitted to shape ecological systems (Czech & Krausman, 1997).

From the perspective of professional ecologists, Christensen (1997) has defined ecosystem management as management 'whose goals, policies and practices are adaptable and based on our best understanding of the ecological interactions and processes necessary to sustain ecosystem function'. But like most of the numerous definitions of this concept, attempts at clarity succeed chiefly in shifting the ambiguity from one part of the lexicon to another (Grumbine, 1997). In the context of this study, an obvious question is whose understanding will be considered best and hence most influential in the application of ecological theory and other forms of knowledge to determine goals and practice? The Aboriginal managers at Dukaladjarranj and the clans with whom they cooperate in the management of fire regard their practice, and the knowledge that underpins it, as not only best, but mandatory if they are to meet their obligations to the land (Yibarbuk, 1998).

In contrast, Andersen (1999) characterizes contemporary fire management practice in northern Australia as ignorant and ill-judged. He appears to interpret an inability or unwillingness to articulate fire management objectives in terms that satisfy experimental ecologists to prevailing aimlessness or mysticism among indigenous and non-indigenous land managers. Clearly, there is a substantial gap to be bridged if we are to develop management regimes for northern Australia's fire-prone landscapes that address widely shared goals. Closing that gap is important because the Federal *Native Title Act* 1993 protects the right of native title holders to carry out hunting and gathering activities on lands with which they retain connection, and these activities may include the use of fire (Hughes, 1995). Recent decisions of Australia's High Court have confirmed elements of these rights, despite potentially conflicting legislation in the States and Territories. We consider that this study makes a potentially useful contribution to debate regarding fire management, because it shows that outcomes from well and consistently executed Aboriginal practice can be congruent with many widely shared land management and conservation goals.

Ecological Integrity

A recurring theme in a number of treatments of the concept of ecosystem management by ecologists is the imperative to maintain ecological integrity. This notion is not much easier to pin down than others (Woodley *et al.*, 1993) but a number of criteria seeking to index a state of ecological integrity have been proposed. Many authors argue that, in seeking to deal with human management of ecosystems, measures of system integrity must deal with social and

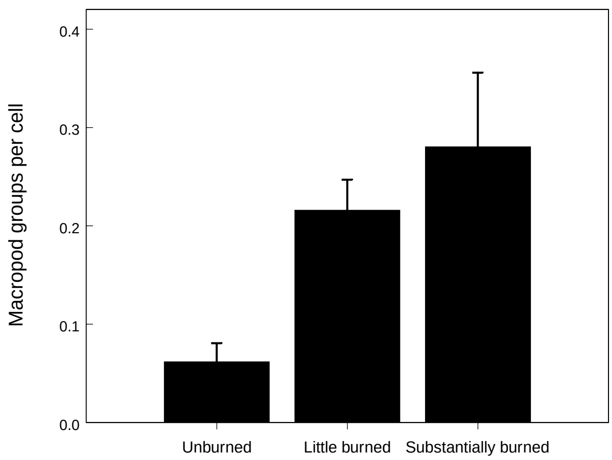

Figure 6 Mean (+ 2 SEM) number of kangaroos groups sighted during aerial survey in cells with different 1997 (year of survey) fire histories.

cultural criteria (values) as well as biophysical (e.g. Regier, 1993). We do not propose to enter that debate here, but rather deal briefly with the match between outcomes of Aboriginal management at Dukaladjarranj and commonly invoked biophysical indicators of ecological integrity. There are as many indicators of integrity as there are monitoring systems and their proponents, so we have matched the phenomena for which we have data against indicators from a number of sources, most notably Woodley *et al.* (1993) and the proposed Core Environmental Indicators for monitoring the state of the Australian environment (ANZECC, 1998). On a number of those measures, the clan estate has been managed to maintain its ecological integrity:

Biological diversity. Vertebrate diversity is high, and a number of species that are absent or have declined elsewhere remain abundant (e.g. the Northern Quoll and Partridge Pigeon). Aboriginal informants report no recent loss of species from the region. While our surveys were not comprehensive, all of the available evidence suggests that the site supports the suite of species that would be expected to occur in such habitats, and substantial populations of species that are infrequently encountered in similar environments elsewhere.

Presence of rare and range-restricted native fauna. The site supports a number of vertebrate wildlife species that have been classified as rare or insufficiently known (notably the Oenpelli Python *Morelia oenpelliensis* and the Northern Hopping Mouse *Notomys aquilo* Thomas). There was also a substantial number of additional species with geographically restricted distribution. Many of these observations extended the known range of these species from sites in or adjoining Kakadu National Park, suggesting that management regimes are at least as favourable as those applying within the Kakadu World Heritage conservation area.

Threatened communities. Fire-sensitive vegetation types that are in serious decline elsewhere (Bowman & Panton, 1993; Price & Bowman, 1994; Russell-Smith *et al.*, 1998) are well represented (e.g. *Callitris intratropica*-dominated open forests, sandstone heaths), and the relative abundance of individual plant species (obligate seeders) that require long fire-free intervals to maintain populations is high compared with other sites in similar environments. For example, despite the small plot sizes used in this study, species richness of obligate seeder shrubs in plots that had not been burned within the last few months was nearly twice as high (5.3 species per 78 m^2 plot) as in similar sandstone environments in Kakadu National Park (3.1 species per 125 m^2 plot: Russell-Smith *et al.*, 1998).

Harvested species. The estate continues to produce natural products (e.g. large macropods, including the spectacular endemic Black Wallaroo, a range of yams, and many other items (D. Jackson & L. Williams, unpublished data)) that are highly valued by the land managers.

Threatening processes. The region is little affected by late-dry season destructive wildfires that are damaging many other parts of the escarpment (Russell-Smith *et al.*, 1997b, 1998), including sites like Kakadu National Park, which is actively managed by a large staff to protect its recognized World Heritage values.

Exotic plants. Diversity of exotic plants is extraordinarily low. In more than 94 vegetation quadrats including 157 species of woody and herbaceous plants, no exotic species were recorded.

Exotic animals. Diversity of exotic animals is low, being dominated by large domestic stock (buffalo, cattle and horses) which have been actively introduced or tolerated in the region for many decades. The exotic rodents (*Rattus rattus* Linnaeous and *Mus musculus* Linnaeous) that are well-established over much of the Australian mainland were not recorded during the study.

The region of which this estate is a part has been classed as possessing high wilderness values (Robertson *et al.*, 1992). Yet maintenance of many of the region's natural values has and continues to be dependent on human intervention. Sustained Aboriginal management and fire use has imposed a regime of high frequency, low intensity disturbance at fine spatial scales, to which the contemporary biota appears highly resilient. Provided levels of intervention and the associated skills are maintained, the unique sandstone landscape can be maintained in a state of equilibrium or, at least, relatively low variance in the mix of landscape elements. In contrast, a low intervention fire regime characterized by intense wildfires that are (i) frequent relative to the time required for re-establishment of fire sensitive communities (Russell-Smith *et al.*, 1998) and (ii) large relative to the total area of the ecosystem (Jacklyn & Russell-Smith, 1998), are likely to push landscape dynamics to a position of instability and increased probability of ultimate collapse (Turner *et al.*, 1993). Treating the region as a wilderness and so minimizing the human presence invites extreme outcomes.

The important role of humans in conservation of these natural and cultural landscapes is likely to best be fostered by cooperative programmes that encourage continued Aboriginal association with, and responsibility for, traditional lands. Such programmes are currently being implemented through short-term (3 years) funding. However, given that the outcomes include the protection of attributes that are valued globally, longer-term support would also appear to be justified. Moreover, in tandem with the implementation of such programmes, the opportunity should be taken to involve non-indigenous managers in implementation and evaluation, so that, subject to the agreement of the traditional custodians of relevant knowledge, the benefits of indigenous practice can be extended to the wider landscape.

The actions and motivations of Aboriginal people in regard to fire management extend well beyond the strictly utilitarian. Fire and its use also play important social and spiritual roles (Langton, 1998; Yibarbuk, 1998), and hence in sustaining a culture characterized by intimate association with land. Premature or otherwise clumsy attempts to 'transplant' the technology without supporting the ongoing participation and recognition of its most skilled practitioners is not only likely to produce inferior results, but also squander an important opportunity to develop genuine partnerships among Australia's indigenous and non-indigenous natural resource managers.

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BIOSKETCHES

All authors are associated with the **Tropical Savannas Cooperative Research Centre**, a partnership among north Australian research groups with a goal to promote improved land management.

Dean Yibarbuk and **Charles Godjuwa** lead the Djelk Rangers, a team of Aboriginal land managers. Dean has a particular interest in fire use and in linking traditional knowledge and western science to address land management problems. Other authors are presently affiliated with universities, Government agencies and non-government indigenous organizations. Together they have collected ecological research and wildlife management experience exceeding 70 years. Interests range through wetland management, fire management, indigenous use of wildlife and influences of Aboriginal and European land management practice on the structure and function of landscapes.

The study arose from an initiative of the Parks and Wildlife Commission to bring together at remote sites, with comprehensive logistic support, large groups of agency and non-Government personnel for intensive study of biodiversity issues.