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Fire management and biodiversity of the western Arnhem Land plateau

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INTRODUCTION

This chapter describes the biodiversity values of the western Arnhem Land plateau (the ‘stone country’) and surrounds, attempts to provide a brief assessment of condition and trends in that biodiversity, describes the responses of some biodiversity components to fire regimes, and gives guidelines for fire management that may provide for biodiversity conservation within a broader management context. This review is constrained by the lack of substantial ecological studies that have been conducted in the stone country proper: in the Top End of the Northern Territory. With some notable exceptions, most of the autecological studies that relate to fire responses of individual plant and animal species, and all of the longer-term ecological experimental studies, have been conducted in the lowlands surrounding the western Arnhem Land plateau. These lowland studies provide invaluable context and are referred to here, but extrapolations and interpretations derived from such lowland studies should be applied with considerable care to the stone country.

The current fire regimes of the Arnhem Land plateau are described in detail elsewhere in this volume (e.g. Chapter 9, 10 and 14). The apparent detrimental impacts of those regimes upon biodiversity, as described in this chapter, have recently prompted the establishment of the most detailed and considered ecologically based fire-management strategy yet developed in northern Australia: the *Kakadu National Park Arnhemland plateau Fire Management Plan* (Petty *et al.* 2007a). This is an admirable attempt at the resolution of a very significant (and perhaps worsening) conservation management problem. However, its execution and outcomes remain untested, and it focuses specifically only on that 15% of the plateau area encompassed within Kakadu National Park. An analogous management approach that encompasses the rest of the plateau area is described in Chapter 9 (Edwards and Russell-Smith 2009).

BIODIVERSITY VALUES

The broad landscape context and description of the western Arnhem Land plateau area is provided in Chapters 1 and 4. There has been no consolidated documentation of its biodiversity values, although it has long been recognised as special (Specht 1958; Northern Territory

Reserves Board 1972; Christian and Aldrick 1977; Senate Standing Committee on Environment, Recreation and the Arts 1988). Aspects of its conservation significance were described in some of the reports prepared for the land-use planning process that resulted in the establishment and co-habitation of Kakadu National Park and a set of mining ventures (notably Ranger Uranium Environmental Inquiry 1976), subsequently in the description of the conservation values (including World Heritage status) of Kakadu National Park (e.g. Press *et al.* 1995a) and, most recently, in the case for the establishment of Wardekker Indigenous Protected Areas including those parts of the plateau area to the east of the Kakadu boundary.

Intricate knowledge of the biological values and ecological functioning of the plateau area has been held by Indigenous landowners for millennia. This still exists in the keeping of senior custodians (e.g. Russell-Smith *et al.* 1997; Telfer and Garde 2006; Chapter 5), but the generational transfer of this knowledge has been greatly interrupted over the last 50 years as the lands have become increasingly deserted, and the day-to-day need to hold and apply the knowledge has diminished (see Chapter 4). As described elsewhere in this volume, a range of recent initiatives has attempted to reverse this trend by attributing value to this venerable knowledge and seeking to maintain and apply it.

Since European settlement, biologists have been intrigued by the plateau and recognised the distinctiveness of its biota (e.g. Dahl 1897, 1926). Most of the early visits by biologists were brief collecting trips to the plateau margins. The American–Australian Arnhem Land Scientific Expedition of 1948 (Specht 1958) provided the first substantial assessment of the biodiversity of the plateau: particularly in the recognition of the antiquity of its biota, the relatedness of components of its vegetation to nutrient-poor heathlands in temperate Australia, and the distinctiveness of some of its components from the vegetation of the surrounding extensive lowlands. Specht (1958) recognised the melange of floristic elements: co-existing, intermingling or very finely partitioned because of the diversity of environments associated with the marked topographic variability of the plateau, including rock pavements, deeply dissected gorges, wetland systems and deep sandy plains. The elements include:

- relictual rainforest species (mostly in sheltered gorges)
- heathland species (mostly on rock pavements and sandsheets) with a comparably long history in this landscape
- more recent and vagile rainforest species
- components of the eucalypt-dominated savanna systems that form the dominant vegetation type across the surrounding lowlands.

Plant diversity is high: partly because of this fine-scale mix of environments and vegetation elements, but also because the highly dissected plateau allows for extensive localised speciation of groups with limited dispersal ability, such as the *Stylidium* trigger plants (Carlquist 1979).

Many of the relictual species are of outstanding evolutionary and conservation significance. The plateau has remained an unbroken (but often isolated) feature of the landscape for more than 100 million years, while the surrounding lowlands have intermittently been inundated (Needman 1988; Duretto and Ladiges 1997). The range of landscape elements (and hence microclimates) across the plateau has offered shelter to species faced with climate changes to which they would otherwise be ill-suited. The persistence of some of these species has been remarkable. In some cases, the persistence is now extraordinarily precarious. Figure 8.1 illustrates some examples of the distributions of some species for which the plateau provides refuge. From at least the Jurassic (136–195 million years ago), much of the Gondwana continent was dominated by conifers including the genus *Podocarpus* (Adam 1994); fossil pollen samples from

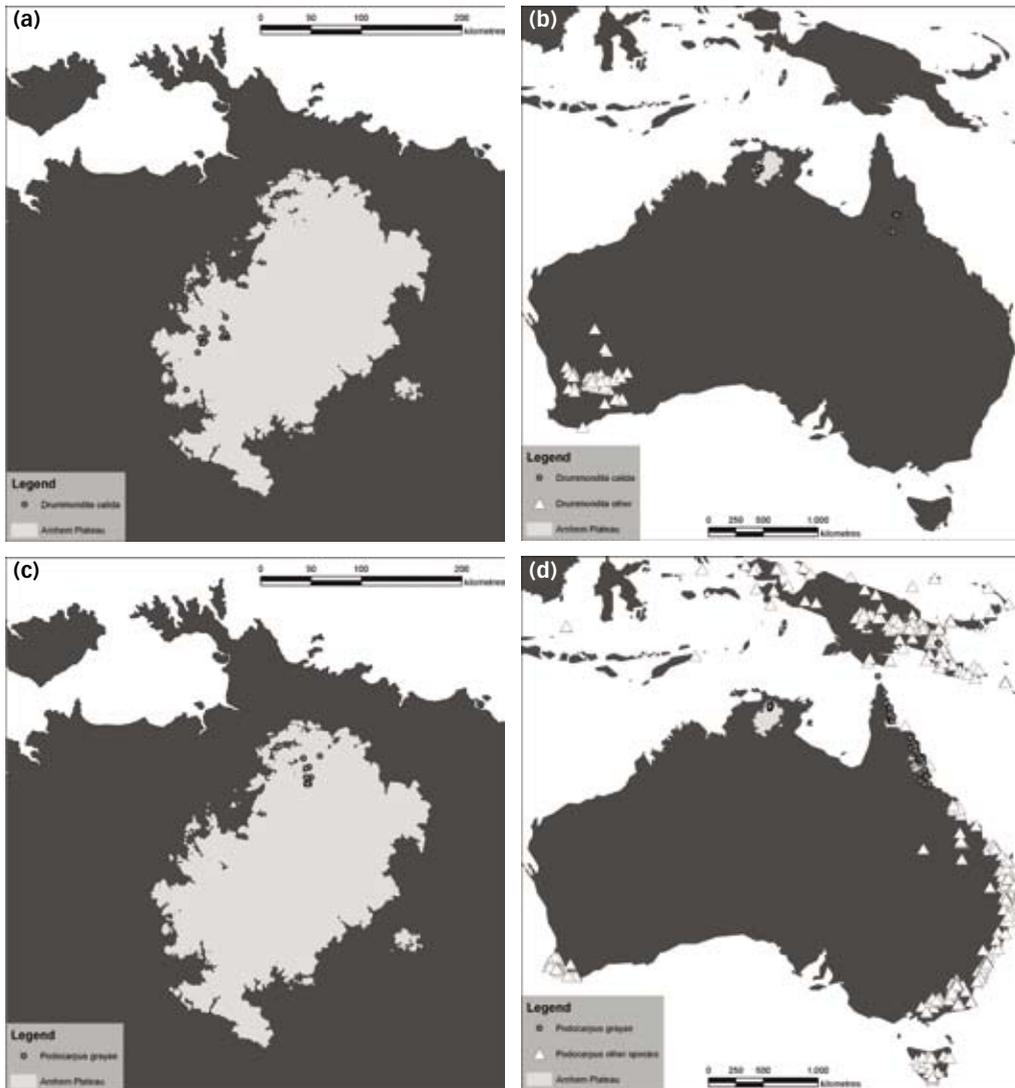


Figure 8.1 Maps of the current locations of (a) *Drummondita calida* in the western Arnhem Land plateau (b) all *Drummondita* species (c) *Podocarpus grayae* in the western Arnhem Land plateau (d) all *Podocarpus* species (but noting that records beyond Australia are not comprehensive). (Source: Australian Virtual Herbarium).

sediment cores near the source of the South Alligator River attest to the presence of *Nothofagus* and *Podocarpus* in this region around 50 million years ago (Truswell 1982; Wilson *et al.* 1996). *Podocarpus* persists in Australia now mostly in high-altitude rainforests of the Great Dividing Range. In the Northern Territory, it exists only in a small number of highly incised gully systems draining into the East Alligator River, with the species present, *P. grayae*, separated by more than 1500 km from its nearest other populations, in northern Queensland (Figure 8.1, 8.2a). Another example of a relictual species is the heathland plant *Drummondita calida*, which is known in the Northern Territory only from the western escarpment of the Arnhem Land plateau. This population is remarkably disjunct from its only other occurrence in the Einasleigh



Figure 8.2 Examples of some plants and animals endemic to – or with relictual distribution including the – Arnhem Land plateau (a) *Podocarpus grayae* (b) *Hildegardia australiensis* (photo: A. Fisher) (c) *Boronia viridiflora* (photo: D. Liddle) (d) Leichhardt's grasshopper on one its main host plants, *Ptyrodia jamesii* (photo: P. Barrow). This specimen carries a plastic disk used as marker during a recent study of dispersal.

Uplands area of north Queensland, which is itself separated from all other species in the genus, which are all restricted to heathlands in south-western Australia (Figure 8.1). A contrasting example is the spectacular tree *Hildegardia australiensis* (Figure 8.2b; Figure 8.4e), which is restricted to a few sites of the Arnhem Land plateau; the other 10 members of this genus occur in Africa, south-eastern Asia and the Caribbean. A feature of many of the most localised plant species of the Arnhem Land plateau is their current restriction to some of the most rugged habitats, such as cliffs, waterfalls, gorges and defiles. Such sites offer the greatest possible protection from fire and/or buffering from otherwise largely unfavourable climatic conditions.

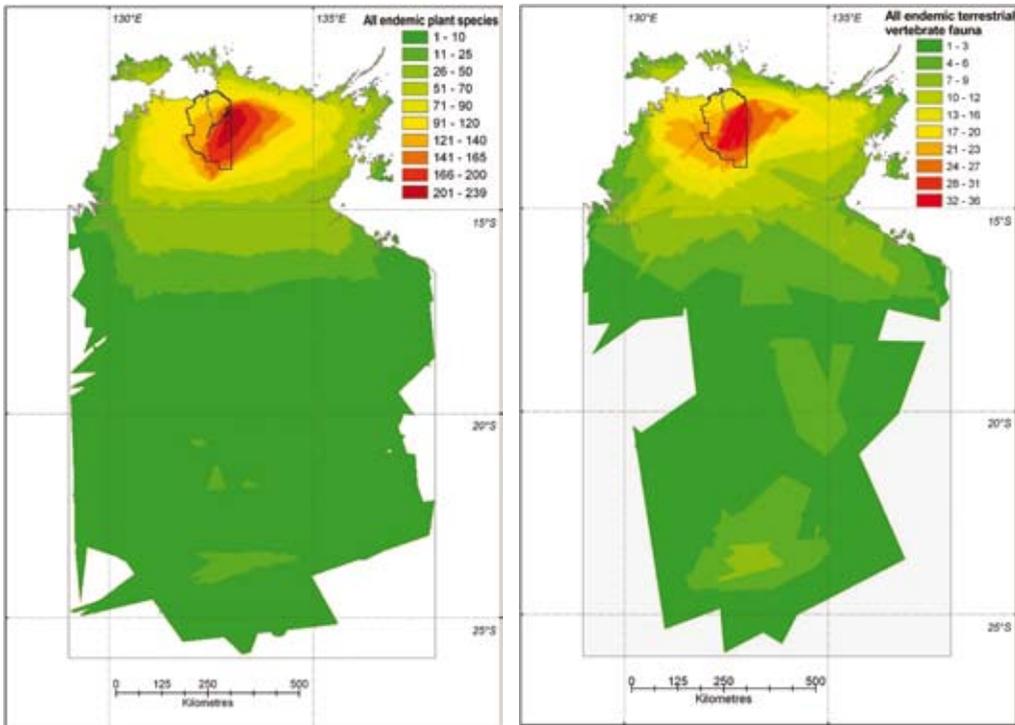


Figure 8.3 Contour maps showing the richness of Northern Territory endemic (a) plant species and (b) terrestrial vertebrate species, with the boundary of Kakadu National Park superimposed. (Sources: Woinarski *et al.* (2006) for plants; Woinarski and Hempel (unpublished) for vertebrate species.)

Examples include the threatened plants *Boronia viridiflora* and *Sauropus filicinus*, which are both now restricted to crevices in a small set of sandstone cliff faces (Figure 8.2c).

The international significance of the Arnhem Land plateau as a centre for plant biodiversity was explicitly recognised through the inclusion of a ‘Kakadu–Alligator Rivers Region’ in a listing of the world’s centres for plant diversity (Ingwersen 1995), in part on the basis that this area included about 40 endemic plant species. This region was also recognised in a more systematic analysis of Australian sites of plant endemism by Crisp *et al.* (2001). More recently, and with access to a far more comprehensive information base, Woinarski *et al.* (2006) analysed the distributions of all plant species endemic to the Northern Territory, and established that the Arnhem Land plateau was by far the most significant Northern Territory site for endemic plants (Figure 8.3a), with 172 plant species restricted to the plateau and a further 25 species mostly restricted (>90% of their range) to it. Although some sites in the Arnhem Land plateau area are botanically well explored, many other areas remain largely unvisited and the number of plant species (including endemic plant species) known from the plateau continues to increase substantially (Brennan 1996; Harwood and Dessein 2005).

As with plants, the Arnhem Land plateau is of international significance for endemic, relictual and threatened fauna. To date, there has been no comprehensive assessment of the richness and endemism of the invertebrate fauna of the western Arnhem Land plateau. Our knowledge of the terrestrial invertebrate fauna is especially poor, and is based primarily on opportunistic collections from that part of the Arnhem Land plateau and escarpment in Kakadu National Park (Press *et al.* 1995b). Two of the best known groups are ants and grasshoppers. The ant

fauna of Australia's monsoonal tropics is exceptionally rich (Andersen 2000), and there are likely to be several hundred species in the western Arnhem Land plateau. The ant fauna of the sandstone country associated with the western Arnhem escarpment is highly distinctive, with perhaps 30% of species not occurring in the lowland savanna to the west. Many of these are undoubtedly endemic to the immediate region, and this endemism occurs at higher taxonomic levels, including an undescribed species group of *Meranoplus* that is most unlike other members of the genus (Andersen 2006). Such endemism is clearly true more generally for the western Arnhem Land plateau, with, for example, *Aphaenogaster reicheli* known only from Podocarpus Canyon (Shattuck 2008), and a species of *Rhytidoponera* recently collected from the Mann River region representing a new and highly unusual species group (AN Andersen, unpublished). The western Arnhem Land region is also notable for the occurrence of species with highly disjunct distributions. For example, the only record of the infamous bull-ant genus *Myrmecia* in the Top End is an undescribed species from the Marrawal plateau in the southwest of the region, which also occurs on the Mitchell plateau in Western Australia's northern Kimberley (AN Andersen, unpublished.). The coastal Queensland species *Dolichoderus scrobiculatus* has also been collected from the Marrawal plateau – about 2000 km from its previously known range; this is the only record of the genus in the Northern Territory. The only record of the ant genus *Prionopelta* in the Northern Territory is from a rainforest patch associated with the Arnhem Land plateau (Reichel and Andersen 1996).

The Kakadu region supports a highly diverse and significant grasshopper fauna, with at least 11 species endemic to sandstone country of the Arnhem escarpment and plateau (Andersen *et al.* 2000). One notable plateau species is Leichhardt's grasshopper *Petasida ephippigera* (Figure 8.2d), which is restricted to the plateau of western Arnhem Land and a few other sandstone isolates in the Top End and Victoria River region (Lowe 1995). It is notable for its dependence on a narrow set of host food plants: primarily two aromatic shrub species of *Pityrodia* (Verbenaceae), and for its bright and spectacular orange and blue aposematic coloration. It has special spiritual significance for the local Jawoyn people, who relate it to the 'lightning man' (Namarrgon) responsible for the wet season storms.

The biogeographic significance of the west Arnhem Land plateau for ants and grasshoppers is also shown in other insect groups for which there is sufficient information. For example, several species of dragonfly are endemic to streams associated with the plateau (Press *et al.* 1995b). Some of these, such as *Hemigomphus magela*, are most closely allied to species from temperate southern Australia. All other species of *Hemigomphus* occur in coastal and montane streams of eastern Australia (Watson *et al.* 1991), with closely allied genera occurring in south-western Australia and South America.

The freshwater invertebrate fauna of the western Arnhem Land plateau also includes a substantial and significant endemic component, although quantification of this is currently hampered by very limited collection and taxonomic scrutiny (such limitations are now being addressed by a range of collaborative studies involving the Environmental Research Institute of the Supervising Scientist). The fauna includes an endemic family of shrimps – the Kakaducarididae, comprising two endemic genera (Bruce 1993, Bruce and Short 1993, Page *et al.* 2008) – as well as an endemic genus of phreatoicidan isopod (*Eophreatoicus*) that has exceptional species-level diversity (Dr G Wilson, Australian Museum *pers. comm.*). Most of these macro-crustacean species have very restricted distributions: often limited to single streams, seeps or springs.

Twenty-two vertebrate species (three fish, two frog, eleven reptile, four bird and two mammal) are endemic to the Arnhem Land plateau (Table 8.1) – by far the highest concentration in the Northern Territory (Figure 8.2b), and very high by Australian standards. The list includes one of Australia's largest snakes (the Oenpelli python *Morelia oenpelliensis*) and a large macropod (the

Table 8.1: List of vertebrate species endemic to the Arnhem Land plateau.

Fish	
<i>Hypseleotris barrawayi</i>	
<i>Craterocephalus marianae</i>	Mariana's hardyhead
<i>Pingalla midgleyi</i>	Midgley's grunter
Frogs	
<i>Uperoleia arenicola</i>	Jabiru toadlet
<i>Litoria personata</i>	masked rock-frog
Reptiles	
<i>Chelodina burrungandjii</i>	sandstone long-necked turtle
<i>Gehyra pamela</i>	Arnhem Land spotted dtella
<i>Oedura gemmata</i>	jewelled velvet gecko
<i>Pseudothecadactylus lindneri</i>	giant cave gecko
<i>Ctenotus arnhemensis</i>	Arnhemland ctenotus
<i>Ctenotus coggeri</i>	Cogger's ctenotus
<i>Ctenotus gagudju</i>	Kakadu ctenotus
<i>Ctenotus kurnbudj</i>	Alligator Rivers ctenotus
<i>Egernia obiri</i>	Arnhemland egernia
<i>Menetia concinna</i>	neat menetia
<i>Morelia oenpelliensis</i>	Oenpelli python
Birds	
<i>Petrophassa rufipennis</i>	chestnut-quilled rock-pigeon
<i>Ptilinopus cinctus</i>	banded fruit-dove
<i>Amytornis woodwardi</i>	white-throated grass-wren
<i>Meliphaga albilineata</i>	white-lined honeyeater
Mammals	
<i>Macropus bernardus</i>	black wallaroo
<i>Zyzomys maini</i>	Arnhem Land rock-rat

black wallaroo *Macropus bernardus*). Other vertebrate species thought to be restricted to the plateau (notably a small carnivorous marsupial, *Planigale* sp.) remain undescribed.

Some of the plateau endemics form disjunct populations of more wide-ranging superspecies, whose level of taxonomic divergence provides some indication of the duration of their current isolation and the extent to which they can or cannot disperse through less-suitable lowland habitats. For example, the white-lined honeyeater *Meliphaga albilineata*, chestnut-quilled rock-pigeon *Petrophassa rufipennis*, white-throated grass-wren *Amytornis woodwardi*, Arnhem rock-rat *Zyzomys maini* and giant cave gecko *Pseudothecadactylus lindneri* all have closely related species (allospecies) occurring in comparable sandstone environments in the Kimberley, Western Australia (*M. fordiana*, *P. albipennis*, *A. housei*, *Z. woodwardi* and *P. cavaticus*, respectively). There are also comparable 'sister' species for the grass-wren (Carpentarian grass-wren *A. dorotheae*) and rock-rat (Carpentarian rock-rat *Z. palatalis*) in the smaller isolated sandstone ranges of the Gulf of Carpentaria hinterland. Other species occur broadly across almost all of the disjunct ranges of northern Australia (e.g. common rock-rat *Z. argurus*), or patchily across some (e.g. nabarlek *Petrogale concinna*, sandstone antechinus *Pseudantechinus*

bilarni: Fisher *et al.* 2000). More idiosyncratic is the banded fruit-dove *Ptilinopus cinctus*, which is restricted in Australia to rainforests of the Arnhem Land plateau, but variably considered specifically distinct from a closely related form occurring from Bali to the Lesser Sunda Islands (including Timor) in Indonesia (Christidis and Boles 2007).

The plateau supports not only the Northern Territory's largest concentration of endemic species but also one of the largest concentration of threatened species (Table 8.2; Department of Natural Resources Environment and The Arts 2005). This latter factor reflects two somewhat opposing factors:

1. that some species have declined substantially across their broader (lowland) range, but are now finding their last refuge in the stone country: in part because there may be some moderation in the plateau region of some otherwise pervasive threats (such as weeds and feral animals)
2. that some species largely restricted to the plateau are now declining in the area (e.g. white-throated grass-wren).

In fact, these two variants are probably just truncated versions of a process that has been operating over very long timeframes: that species ebb and flow from the plateau to the lowlands, retreating to the plateau during inhospitable periods and expanding out in favourable times. Where the periods are particularly inhospitable, there will also be retreat to the most rugged (insulated) parts of the plateau and, at times of particular stress, even these may prove inadequate.

The plateau supports not only threatened and relictual species but also distinctive assemblages of species (Figure 8.4). The most notable of these are sandstone rainforests and sandstone heathlands. There is a range of rainforest assemblages in and around the Arnhem Land plateau that are floristically dissimilar to the rainforests of the lowlands (Russell-Smith 1991). The most extensive and notable are closed forests dominated by the ancient proto-eucalypt *Allosyncarpia ternata*. This majestic tree is restricted to the Arnhem Land plateau, and its closest relatives comprise similarly old relictual genera isolated in mountain tops in Queensland (*Stockwellia*), eastern Malesia (*Eucalyptopsis*) and New Caledonia (*Arillastrum*) (Ladiges *et al.* 2003; Crisp *et al.* 2004). On the Arnhem Land plateau, *Allosyncarpia ternata* can form monospecific forests (especially on slopes and sandsheets) or mix with a diverse set of rainforest trees (especially in gullies, watercourses and springs) (Bowman 1991; Russell-Smith 1991), with a total extent of 1138 km² (Russell-Smith *et al.* 1993).

Heathland communities are scattered across Australia, especially on nutrient-poor sandy substrates. They are typically dominated by woody shrubs belonging to the families Myrtaceae, Fabaceae and Ericaceae. Many of the heathland species reproduce only by seed, and characteristically require a disturbance-free interval of several to many years before they are sufficiently mature to produce seeds. Across the Arnhem Land plateau, open heathlands and shrublands occur on rock platforms, skeletal sandy soils and deeper sand sheets – occupying a total area of between 18 920 and 24 510 km², depending upon the inclusion, or not, of sandstone isolates outside the periphery of the core plateau area (Blake 2004). These formations may be floristically diverse, and many of the constituent plant and animal species are endemic. More species-poor versions of these communities extend from the Arnhem Land plateau to smaller and less-dissected sandstone isolates elsewhere in northern Australia, or to sandsheets and coastal dunes of the lowlands; these attenuated heathlands typically include lower representation of obligate-seeding species.

The Arnhem Land plateau supports an unusually diverse set (13 species) of hummock grasses *Triodia* spp.: in part mixed among the sandstone heathlands, as the dominant component of

Table 8.2: List of threatened species occurring in the Arnhem Land plateau. Trend assessment (for the Arnhem Land plateau area) is based on interpretation from Woinarski (2004a) and Woinarski et al. (2007). Codes for conservation status: CR=critically endangered; EN=endangered; VU=vulnerable; DD=data deficient; – = not listed. Northern Territory (NT) status is as defined in regulations of the *Territory Parks and Wildlife Conservation Act 2000*; Australian (Aust.) status is under the *Environment Protection and Biodiversity Conservation Act 1999*.

Species		Status (NT)	Status (Aust.)	Trend
Plants				
<i>Acacia</i> sp. Graveside Gorge		CR	CR	unknown
<i>Boronia quadrilata</i>		VU	VU	unknown
<i>Boronia viridiflora</i>		VU	VU	unknown
<i>Cephalomanes obscurum</i>		EN	–	unknown
<i>Eleocharis retroflexa</i>		DD	VU	unknown
<i>Freycinetia excelsa</i>		VU	–	unknown
<i>Hibiscus brennanii</i>		VU	VU	unknown
<i>Lithomyrtus linariifolia</i>		VU	–	unknown
<i>Sauropus filicinus</i>		DD	VU	unknown
<i>Toechima</i> sp. East Alligator		EN	EN	unknown
<i>Utricularia singeriana</i>		VU	–	unknown
Reptiles				
<i>Varanus mertensi</i>	Mertens' water monitor	VU	–	declining
<i>Varanus panoptes</i>	yellow-spotted monitor	VU	–	declining
<i>Egernia obiri</i>	Arnhemland egernia	EN	EN	declining
<i>Morelia oenpelliensis</i>	Oenpelli python	VU	–	unknown
Birds				
<i>Dromaius novaehollandiae</i>	emu	VU	–	declining
<i>Erythrotriorchis radiatus</i>	red goshawk	VU	VU	stable
<i>Ardeotis australis</i>	Australian bustard	VU	–	declining
<i>Geophaps smithii smithii</i>	partridge pigeon	VU	VU	unknown
<i>Tyto novaehollandiae kimberli</i>	masked owl (north Australian mainland)	VU	VU	unknown
<i>Amytornis woodwardi</i>	white-throated grasswren	VU	–	declining
<i>Falcunculus frontatus whitei</i>	crested shrike-tit (northern)	VU	VU	unknown
<i>Erythrura gouldiae</i>	gouldian finch	EN	EN	stable
Mammals				
<i>Dasyurus hallucatus</i>	northern quoll	CR	EN	declining
<i>Phascogale pirata</i>	northern brush-tailed phascogale	VU	–	declining
<i>Isoodon auratus</i>	golden bandicoot	EN	VU	declining, probably extinct
<i>Hipposideros diadema inornatus</i>	Arnhem leaf-nosed bat	VU	–	unknown
<i>Conilurus penicillatus</i>	brush-tailed rabbit-rat	VU	–	declining
<i>Mesembriomys macrurus</i>	golden-backed tree-rat	CR	VU	declining
<i>Notomys aquilo</i>	northern hopping-mouse	VU	VU	unknown
<i>Zyomys maini</i>	Arnhem Land rock-rat	VU	VU	declining

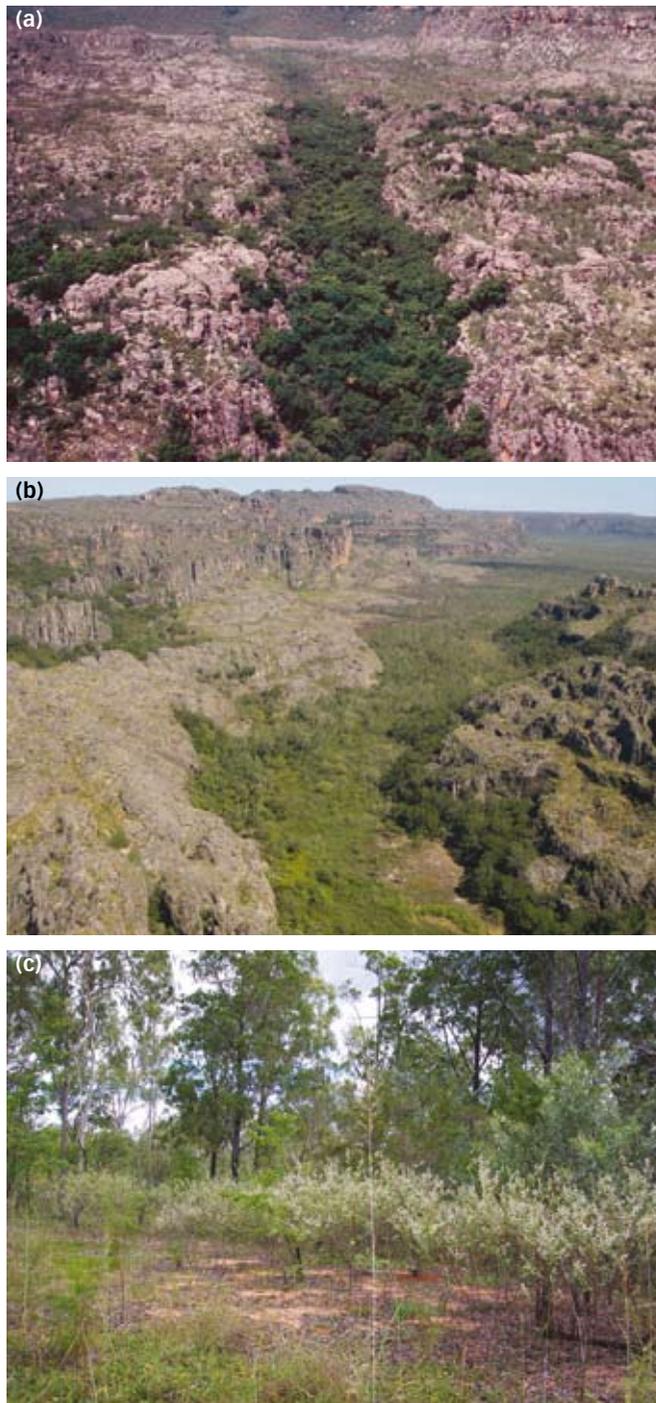


Figure 8.4 Characteristic landscapes and ecological communities of the stone country. (a) fissures and gullies offering shelter for fire-sensitive plants and rainforests (b) escarpment (c) edge of sandstone heathland with woodland dominated by *Callitris intratropica* (d) rainforest (e) rainforest patch on rocky hilltop, here dominated by *Hildegardia australiense* (all photos by Jeremy Russell-Smith).

understorey in some woodlands, or as one of the few plants growing in boulder fields and rock pavements. As with hummock grasses elsewhere (Morton and James 1988), these may be ecologically pivotal: providing key habitat, shelter and food resources for many fauna species. For the western Arnhem Land plateau, the white-throated grass-wren is probably one such example. The plateau is also the centre of diversity (11 species) of resurrection grasses *Micraira* spp. – all with unusual moss-like appearance, spirally arranged leaves, and mostly occurring on rock pavements with no or skeletal soil. These typically highly localised species are particularly adapted to the extreme seasonal challenge of such sites, notably by showing very rapid response of dried-off foliage to rehydration ('revival': Lazarides 1979).

Some plateau habitats are far less distinctive. As with the lowlands, eucalypt open forests and woodlands occur over extensive areas (about 75% of the Western Arnhem Land Fire Abatement project area: Edwards and Russell-Smith 2009) of the plateau. While there is some degree of floristic variation between upland and lowland eucalypt woodlands (e.g. Taylor and Dunlop 1985; Wilson *et al.* 1990; Dunlop and Webb 1991), they also show many structural and floristic similarities. Dominant plateau woodland trees include *Eucalyptus tetradonta*, *E. miniata* (and its endemic close relative *E. gigantangion*), *E. tintinnans*, *Corymbia dichromophloia*, *C. arnhemensis*, *C. bleeseri*, *C. kombolgiensis*, *C. ferruginea*, *C. setosa*, *C. latifolia*, *Erythrophleum chlorostachys* and *Xanthostemon paradoxus*, with overstorey and understorey species composition and forest/woodland structure varying subtly or markedly in response to variation in soil depth and texture, geology and landscape position (Burgman and Thompson 1982; Bowman *et al.* 1990, 1993; Wilson *et al.* 1990, 1996). These woodlands support many endemic understorey species, particularly in the genera *Acacia*, *Boronia*, *Calytrix*, *Gardenia*, *Goodenia*, *Grevillea*, *Hibiscus*, *Jacksonia*, *Lithomyrtus* and *Spermacoce*, with many of these understorey plants also forming dominant components in adjacent or intermingling heathlands (Wilson *et al.* 1996; Woinarski *et al.* 2006). As scattered copses, isolated individuals or occasionally dominant in patches, the northern cypress-pine *Callitris intratropica* is a notable feature of these woodlands, and provides a signal of the complex evolutionary mix in the flora of this region. As with many of the heathland plant species, *Callitris* is an obligate seeder, with, in at least some habitats, new plants requiring a fire-free period of at least 10 years to mature sufficiently to set seed (Russell-Smith 2006).

CONDITION AND TRENDS IN BIODIVERSITY

As evident in the assessment of trends for threatened species given in Table 8.2, there is little or no information on trends in the status of many of the plateau species of most conservation significance. Even for those species for which some trend is provided, the monitoring or other data from which this assessment is based is notably meagre. There is no substantial or sustained targeted monitoring program for any threatened species in the plateau area, although some limited monitoring programs have been established recently, with baseline sampling (Kerrigan 2003, 2004; Woinarski 2004a; Cowie 2005; see also Chapter 10).

Limited trend information on some components of biodiversity is available from a range of sources, including Indigenous knowledge, targeted monitoring, incidental by-product from other monitoring, and some ecological studies. In some cases, trends on the plateau reflect more general trends across the broader region (e.g. golden bandicoot *Isoodon auratus*, northern quoll *Dasyurus hallucatus*: Woinarski *et al.* 2001); in other cases, trends on the plateau are more muted or delayed than sharper trends in the lowlands (e.g. *Callitris intratropica*); but in no case is the plateau unequivocally defying trends in the lowlands.

Species have been lost from the plateau. Spectacular depictions of thylacines *Thylacinus cyanocephalus* and Tasmanian devils *Sarcophilus harrisi* in rock art (and, in the latter case, as

subfossils) attest to the regional extinction of these species sometime in the last 20 000 years, and perhaps as recently as 3000 years BP (Calaby and White 1967; Brandl 1972; Calaby and Lewis 1977; Lewis 1977). Bones from surficial archaeological sites demonstrate the presence of northern hopping-mouse *Notomys aquilo*, probably as recently as post-European contact (Jones 1985). This species appears to be now absent from the plateau area, with the sole exception of a 1973 record from the upper Caddell River (Woinarski *et al.* 1999). The golden bandicoot was probably present across the plateau (there are a few isolated records from the nearby lowlands, including as recently as 1967, near Goodparla: Woinarski 2004a), but appears now to have disappeared from the entire Top End mainland. The golden-backed tree-rat *Mesembriomys macrurus* is recorded in the Northern Territory from only three specimens: the most recent being in 1969 from Deaf Adder Gorge (Woinarski 2004a). It has not been trapped since, notwithstanding substantial survey effort including in many areas of the plateau. However, a recent study using stuffed mammals as props to record traditional knowledge (M. Ziembicki *pers. comm.*) has indicated that this species may persist in at least one site on the plateau.

Several species have declined precipitously over the last decade following the colonisation of the plateau by cane toads *Chaunus marinus* (mostly from 2000–01). The most extreme of these declines has been for the northern quoll (Watson and Woinarski 2003), which has suffered consequential local extinction over most of its range. Bucking the general trend, a population of quolls has persisted (so far) following toad colonisation near the East Alligator ranger station at Kakadu National Park (J. Woinarski unpublished). This persistence mirrors some examples of local populations of quolls persisting with toads in parts of Queensland, typically in the most rugged areas, which are the highest habitat quality for northern quolls (Woinarski *et al.* 2008). Other species known to have suffered at least short-term declines in the plateau (and throughout the broader Top End region) because of toads include Mertens' water monitor *Varanus mertensi*, yellow-spotted goanna *V. panoptes*, freshwater crocodile *Crocodylus johnstonei* and king brown snake *Pseudechis australis* (e.g. Watson and Woinarski 2003; Doody *et al.* 2006; Smith and Phillips 2006; Griffiths and McKay 2007). Declines are also suspected for some small dasyurid marsupials (e.g. northern brush-tailed phascogale *Phascogale pirata*), some agamid lizards and some other snake species (Watson and Woinarski 2003). It is possible that the significant endemic invertebrate fauna of the plateau wetlands may also have been detrimentally affected by toads (Dr C. Humphries *pers. comm.*).

Across much of the Top End, there has been substantial decline of the mammal fauna over the last century, with declines continuing over the last decades (e.g. Woinarski *et al.* 2001). Anecdotal information from Indigenous landowners and other experienced observers suggest that there has also been a substantial decline of medium-sized mammals in the plateau, which has been reported for some other conspicuous and significant animal species notably including emu *Dromaius novaehollandiae* (Yibarbuk and Cooke 2001) and nabarlek *Petrogale concinna* (D. Pearson *pers. comm.*). Results from fauna sampling at 63 fire monitoring plots at Kakadu (including 26 plots in the Arnhem Land plateau and sandstone outliers) indicate a substantial decline between the baseline sampling (in 2001–02) and most recent sampling (2007–08), with decline occurring both in lowlands and uplands, although perhaps more muted in uplands (Figure 8.5). Such downward trends are also evident from more-intensive sampling at particular sites (e.g. Nawurlandja: Watson and Woinarski 2003).

In contrast, the numbers of some feral animals are increasing in the plateau area, although there are few population estimates available. Gorman *et al.* (2007) reported substantial recent increases in the abundance of feral water buffalo *Bubalus bubalis* from central Arnhem Land, which have been building up again following a significant reduction in their numbers during the Brucellosis and Tuberculosis Eradication Campaign between 1985 and 1989. The cane toad

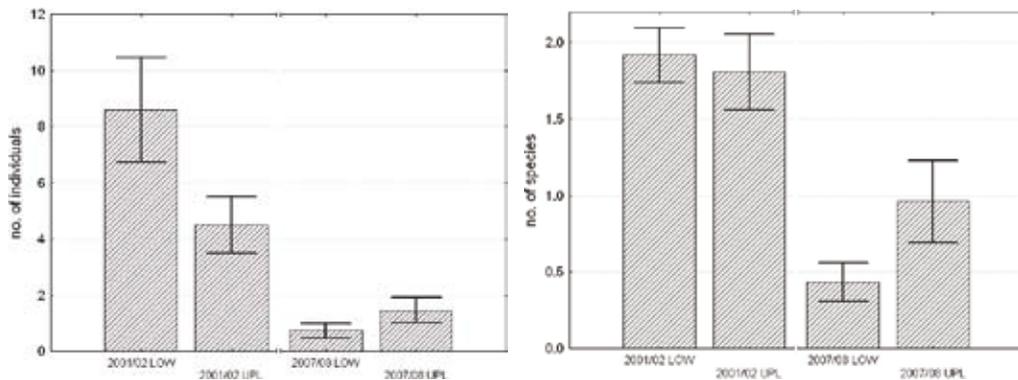


Figure 8.5 Trends in native mammal populations (a) number of individuals (b) number of species in Kakadu National Park, derived from repeat sampling at 63 fixed monitoring plots. Columns represent means, whiskers represent standard errors; 'LOW' indicates lowland sites; 'UPL' indicates upland sites.

invaded the entire plateau area over a 3–5 year period after 2000. The first records of the exotic black rat *Rattus rattus* have been reported (from several areas) in the stone country over the last 5 years. There is some anecdotal evidence for increases in feral cats *Felis catus* and pigs *Sus scrofa*; and feral cattle *Bos taurus/indicus* and horses *Equus caballus* remain at least locally abundant, despite some localised hunting and more systematic aerial shooting programs (Franklin *et al.* 2008).

There is far more information available for trends in (at least some) plants, most particularly for some key charismatic plant species considered indicators of fire regimes – *Allosyncarpia ternata* and *Callitris intratropica* – with adequate information also from specific studies for some other fire-sensitive plant species (e.g. *Petraeomyrtus punicea*: Russell-Smith 2006). More generally, there is a very considerable body of data also available for nearly 1000 plant species from the large set of established fire-monitoring plots at Kakadu and Nitmiluk National Parks (Edwards *et al.* 2003; see also Chapter 10), although only about 10% of these have been recorded in a sufficient number of plots to allow for robust monitoring. Trends in the status and condition of some key vegetation communities are also determinable from a range of studies including fire-plot monitoring (Edwards *et al.* 2003; Chapter 10), remotely-sensed and field-based measurements of boundaries and extent (Bowman 1994; Banfai and Bowman 2006; Bowman and Dingle 2006; Prior *et al.* 2007), correlative distributional and fire-response studies (Bowman and Panton 1993; Russell-Smith *et al.* 1998, 2002; Yibarbuk *et al.* 2001; Bowman *et al.* 2001a, 2004; Bowman and Prior 2004), small-scale experiments (Russell-Smith *et al.* 2002), and demographic modeling (Price and Bowman 1994; Prior *et al.* 2007).

The pattern for broad-scale decline of *Callitris* in the plateau area is clear (Haynes 1985, 1991; Bowman and Panton 1993; Bowman *et al.* 2001a; Bowman and Prior 2004; Russell-Smith 2006; Prior *et al.* 2007), although the prevailing decline may be reversed or stabilised at sites having benign fire regimes (Yibarbuk *et al.* 2001; Edwards *et al.* 2003; Bowman and Prior 2004; Edwards and Russell-Smith 2009). Dead *Callitris* trees remain conspicuous in the landscape for decades, and hence allow for an unusually good assessment of trends from aerial survey or ground-based monitoring (Bowman and Panton 1993; Bowman and Prior 2004; Edwards and Russell-Smith 2009), although the diffuse distribution of *Callitris* across the plateau makes it difficult to map extent and boundary changes effectively, with the exception of some larger and sharply edged copses (e.g. Round Jungle: Bowman and Dingle 2006; Prior *et al.* 2007; Edwards and Russell-Smith 2009). Demographic modelling studies have demonstrated highly significant

(current and) expected declines for *Callitris*, under most prevailing fire regimes (e.g. Price and Bowman 1994; Prior *et al.* 2007).

In contrast, forests dominated by *Allosyncarpia* are more tractably mapped and monitored, and changes over time in the position of their edges – and of the demographic fingerprint of these edges – provide robust and explicit measures of their trend and condition. However, long-term monitoring of the boundaries and (hence) extent of *Allosyncarpia* forests has suggested some inconsistent trends (Bowman and Dingle 2006), probably resolvable with reference to site-specific fire histories. Across much of northern Australia, there are general trends for expansion of rainforests (and increased ‘thickening’ in some savanna woodlands), probably due to some increases in rainfall over the last few decades and to the effects of increased atmospheric CO₂ (Sharp and Whittaker 2003; Sharp and Bowman 2004; Vigilante and Bowman 2004a; Banfai and Bowman 2005, 2006; Bowman and Dingle 2006; Brook and Bowman 2006), albeit modulated (or in some cases reversed) by local factors (notably unfavourable fire regimes).

Beyond estimation of trends in the boundaries of *Allosyncarpia* patches, there have also been assessments of the condition of those patches, and of other plateau rainforests. Russell-Smith and Bowman (1992) reported that: 67% of the sandstone rainforests were ‘severely disturbed’ by fire; 47% ‘severely disturbed’ by feral buffalo and/or cattle; and 27% ‘severely disturbed’ by feral pigs. These measures of health are unlikely to have shown substantial improvement in the 16 years hence.

The overall condition and trend of sandstone heathlands is poor: the general trend has been for diminution of area, and local-scale loss of plant species richness, particularly of the functionally significant obligate seeder species (Russell-Smith *et al.* 1998, 2002; Edwards *et al.* 2003; Edwards and Russell-Smith 2009). This pattern has been variable, and is closely associated with the prevailing fire regime and site location. Overall decline in sandstone heathlands has been of such concern that they have recently been nominated for listing as an endangered ecological community under the Australian *Environment Protection and Biodiversity Conservation Act*. If approved, it would be the first listed community in the Northern Territory, and only the second of the 38 listed Australian ecological communities whose qualification for listing has not been linked to threats from intensive development: the only other exception being for a 5 km² community in south-western Australia threatened primarily by the cinnamon fungus *Phytophthora cinnamomi*. It would also be one of very few listed threatened ecological communities with such a large proportion included in protected areas (23% of its extent is included in Kakadu National Park, 3.5% in Nitmiluk National Park, and almost all of the remainder is within the proposed Warddeken Indigenous Protected Area: A. Edwards *pers. comm.*).

There is some information focused on the spread of undesirable plants into, around and in the plateau, including some consideration of the ongoing increase in extent and dominance of exotic weeds (including mission grass *Pennisetum* spp.: P. Cooke *pers. comm.*). Weed incidence in the plateau area is generally low relative to the surrounding lowlands and pastoral lands (Franklin *et al.* 2008); however, there is increasing incidence or risk of incursions, mostly along road corridors and from current or abandoned sites of more intensive development (such as mine sites), with feral animals (and macropods) responsible for some dispersal. The current incidence of some weeds is so localised that eradication and prevention of further incursions may be possible, given appropriate resourcing for targeted management (P. Cooke *pers. comm.*).

RESPONSES TO FIRE REGIMES

From the discussion above about trends in biodiversity on the plateau, it is evident that many environmental values are in decline. Much of this decline is attributable to inappropriate fire

regimes. To some extent, there is a contrast between plants and animals in this pattern. Decline has been far more marked for many animal species than for plant species. Fire is the principal agent for decline for plants and vegetation communities, but there may be a broader range of threatening factors that have complex and compounding impacts for animals. Furthermore, the response to fire regimes may be more intricate and contextual for animals than for plants.

In terms of conservation management, the most straightforward responses of species to fire regimes are those of obligate seeder plant species. For many such species, adult plants are readily killed by a single fire event, and the survival of the species at a burnt location is totally dependent upon regeneration from seed shed before (or in some cases, after) the fire event. This life history strategy will work only when there is no more fire at that site until after the new plants have reached maturity and produced seeds. (However, the system can be tweaked a little by variable seed dormancy and survival periods, variable susceptibility of adult plants to fires of varying intensity and timing, and by variability in the rate and mode of dispersal of propagules between burnt and unburnt patches: Vigilante and Bowman (2004a).) For many heathland plants – and some plants from other vegetation types – such a required fire-free interval is generally at least 3 years, and may more optimally be 5–8 years. For some of these species, life history also demands another fire requirement – because of shading or other resource exploitation by adult plants, and because plants have a limited lifespan, recruitment may require fire to remove the mature (or senescent) generation; that is, there may be a maximum fire-free period. The models developed for *Callitris intratropica* (Price and Bowman 1994; Russell-Smith 2006; Prior *et al.* 2007) provide the most informed such fire requirements currently available for obligate seeder species in the plateau, and suggest a preferred regime with shortest interval between successive fires of at least 10 years (Edwards and Russell-Smith 2009).

Plant species that can reproduce vegetatively after fire (such as most eucalypts and *Allosyncarpia*) typically have greater flexibility and resilience in persisting through a range of fire regimes. However, these species can still be severely disadvantaged by fires that are so intense that the mature plant is killed, or by fires that are so frequent that regeneration is fatally compromised. For rainforest patches, losses can be extreme (from intense fire); or gradual accretion may occur over decades from retreat at the margins or through openings in the canopy that allow for invasion by fire-promoting grasses (Russell-Smith and Bowman 1992; Prior *et al.* 2007). More broadly, in the savanna woodlands, frequent ‘hot’ fire may decrease the woody basal area and increase grass fuels, leading to irrevocable change in overall vegetation structure – and fire regimes – and render fire management increasingly challenging (Bowman *et al.* 2001a, 2004; Russell-Smith *et al.* 2003; Bowman and Prior 2004; Liedloff and Cook 2007). For the Arnhem Land plateau, such a fire–grass cycle currently involves almost entirely native grasses (particularly spinifex *Triodia* spp. and sorghums *Sarga* spp.: (Franklin *et al.* 2008); but any further incursions by the exotic pasture grasses of the surrounding lowlands may ratchet up the pace and extent of this detrimental change (Rossiter *et al.* 2003; Rossiter-Rachor *et al.* 2008).

To the extent that floristic composition and vegetation structure contribute to habitat suitability for most fauna species, the responses of vegetation to fire regimes also directly affect fauna. But the responses of fauna to fire may be far more complex. In general, the greater mobility of fauna than flora species allows them to work around the highly deterministic site-specific responses to fire of plants. Animals may be able to move from (fire-dictated) unsuitable to suitable patches, although this capability varies depending upon the scale of fire and the dispersal ability of the animals. Leichhardt’s grasshopper, whose individuals may have a dispersal range of less than hundreds of metres, may be far more readily overcome by a 1 km² patch of recently burnt habitat than would a masked owl *Tyto novaehollandiae*, which

may see such a patch as simply another layer of heterogeneity readily encompassed within a far larger territory.

Most information on faunal responses to, and requirements for, fire are derived from studies in the lowlands adjacent to the western Arnhem Land plateau. But, with due caveats, the conclusions from such studies are broadly applicable to the plateau. In the next few paragraphs, we briefly describe some of the fire responses demonstrated in these studies. Most of these studies describe species that are in decline, with a generally inappropriate fire regime a factor contributing to that decline.

In lowland open forests, the partridge pigeon *Geophaps smithii* prefers habitat that has both patches of dense grassy understorey in which to provide shelter for the nest (little more than a scrape in the ground), but also open (burnt) areas in which to forage most efficiently (for fallen seeds) (Fraser *et al.* 2003). The optimal fire regime for this threatened species thus appears to be cool patchy fires at a scale smaller than its territory (1–10 ha), with fires early in the dry season (in order to provide extensive access to the resource of fallen seeds), although fires at this time come with the risk of destruction of nests.

The frilled lizard *Chlamydosaurus kingii* forages most effectively in open (more recently burnt) areas, but suffers greater mortality from fires in the late dry season (Griffiths and Christian 1996). As with the partridge pigeon, its optimal fire regime is probably spatially variable patchy fine-scale and early dry season fire, with most detriment arising from frequent late dry season fires (Brook and Griffiths 2004).

The brush-tailed rabbit-rat *Conilurus penicillatus* dens mostly in fallen hollow logs and hollows in large old trees (Firth *et al.* 2006a). Such sites are likely to be susceptible to intense fires (Williams *et al.* 2003), particularly those late in the dry season. The preferred diet of rabbit-rats comprises grass seeds, particularly those of perennial grass species (Firth *et al.* 2005). These are more likely to dominate in areas that are not exposed to annual fires; and indeed, broad-scale correlative distributional studies have demonstrated a negative association of rabbit-rats with areas that have been most recently burnt (Firth *et al.* 2006b). Thus a preferred fire regime for this species is likely to be one of infrequent fire, with any such fire preferably being relatively fine scale and of low intensity.

The related, but larger, black-footed tree-rat *Mesembriomys gouldii* also dens in tree hollows (and the tangled fronds of pandanus), but its diet comprises a higher proportion of fruits, and it is likely to find it difficult to forage on the ground when the ground layer is encumbered with dense grasses. Its habitat quality is likely to be greatest when the woodlands support a dense tall shrub layer (for these provide the greatest abundance and diversity of fruits) and sparse grass layer (Rankmore 2006). Such a structure occurs only where fire is infrequent (e.g. intervals of 5 or more years) (Taylor and Dunlop 1985; Russell-Smith *et al.* 2003; Vigilante and Bowman 2004b; Woinarski *et al.* 2004).

In an intensive radio-tracking study of the northern quoll in the Kakadu lowlands (before the arrival of cane toads), Oakwood (2000) found that the main cause of mortality was predation, and that this rate of predation increased proportionally to the extent of the home range that was burnt. Consistent with this assessment, more broad-ranging distributional studies have demonstrated that the occurrence of northern quolls was most likely in longer-unburnt sites (Woinarski *et al.* 2008). The management of fire for this species in the Arnhem Land plateau area may be a moot point anyway, given its recent broad-scale demise following cane toad invasion.

The Gouldian finch *Erythrura gouldiae* occurs particularly on stony hillsides, including around the margins of the western Arnhem Land plateau. Like the partridge pigeon, it forages most efficiently in open (burnt) areas, but has a special requirement for a sequential set of seeds of differing grass species to eke it through a resource bottleneck in the early wet season

(Dostine *et al.* 2001). This bottleneck arises because the first rains trigger the loss (through germination) of the fallen grass seed resource on which this species depends (Garnett and Crowley 1994). The Gouldian finch then works on an ecological knife-edge: relying on the local spatial and temporal juxtaposition of both seeds from grass species that are slower to germinate and fast-growing species that most quickly produce the new season's seeds. Perennial grasses are critical to this strategy, and these are most likely to persist in the landscape when fires are relatively infrequent (Woinarski *et al.* 2005). Finally, this species nests in tree hollows, particularly on hill slopes. The availability of such hollows and hollow-bearing trees is likely to decrease with frequent fire.

There have been no substantial management-related studies on an iconic endemic bird of the Arnhem Land plateau: the white-throated grass-wren. However, it is thought to be closely associated with large and extensive (i.e. longer unburnt) hummocks of spinifex *Triodia* spp. and, given the current regime of frequent fire (Edwards and Russell-Smith 2009), likely to be in considerable decline (Woinarski 1992; Woinarski *et al.* 2007; but cf. Noske 1992). A closely related species, the Carpentarian grass-wren, appears to have undergone a very major decline over the last few decades because of too frequent and extensive fire in similar habitat in the Gulf of Carpentaria hinterland (Perry 2005; Woinarski *et al.* 2007).

There is a diverse macropod fauna in the Arnhem Land plateau and surrounds. Indigenous people have long used fire to manipulate habitat suitability for (and hunt) these 'game' species, with some burning designed specifically to stimulate grass production ('green pick') to increase the abundance (or concentrate the distribution) of macropods (Bowman *et al.* 2001b; Telfer and Garde 2006; Murphy and Bowman 2007). Although frequent patchy fires probably do enhance habitat suitability for some of the larger macropod species (including the endemic black wallaroo) in open forests and sandsheets, it is not clear that this is beneficial for smaller macropods (e.g. short-eared rock-wallaby *Petrogale brachyotis*) and in rocky areas (Murphy and Bowman 2007).

Experimental manipulation of fire regimes has provided valuable insight into the responses of plant and animal species, particularly those of the savanna lowlands. The most substantial of these studies was that at Kapalga in the Kakadu lowlands, where four different experimental fire regimes (ranging from fire exclusion to annual, high-intensity fires in the late dry season) were imposed on replicated sub-catchments over a 5-year period (Andersen *et al.* 2003, 2005). The responses of most insect, reptile and bird species were muted (perhaps because of the short timeframe), but there were pronounced responses by small mammals, with particular reductions in abundance in areas burnt annually in the late dry season. For vertebrate fauna, the most detailed analysis of the Kapalga study was for northern brown bandicoot *Isodon macrourus* (Pardon *et al.* 2003). This demonstrated that bandicoots declined under all uniformly applied fire regimes – most particularly under that of annual late dry season fire – with least decline in unburnt and regular early dry season fires. Pardon *et al.* (2003) concluded that fine-scale patchy fires were most beneficial for bandicoots, and frequent intense fires were most detrimental. A conclusion from the Kapalga fire experiment was that the overall proportion of land burnt each year should be reduced. Such reduction could be achieved through spatially explicit long-term protection from fire of some areas and/or more generally through landscape-wide reduction in the annual extent of fire (Andersen *et al.* 2005).

For the longer-lasting (26 years), but unreplicated and smaller-scale, fire 'experiment' (unburnt versus annually burnt) at Solar Village, Woinarski *et al.* (2004) reported substantial change in the vegetation structure and floristic composition of savanna woodlands (towards rainforest elements), with consequential changes in assemblages of vertebrates (Woinarski *et al.* 2004) and ants (Andersen *et al.* 2006). This study demonstrated that a set of species including *Glaphyromorphus* skinks, bar-shouldered dove *Geopelia humeralis*, white-gaped honeyeater

Lichenostomus unicolor, white-throated honeyeater *Melithreptus albogularis*, dusky honeyeater *Myzomela obscura*, northern fantail *Rhipidura refiventris*, yellow oriole *Oriolus flavocinctus*, common brushtail possum *Trichosurus vulpecula* and black-footed tree-rat is clearly disadvantaged by frequent fire, and some of these species are in decline (and/or are occurring at abundances far below their potential) because frequent fire in lowland woodlands has left little long unburnt habitat (Woinarski 2004b). Conversely, other species were more common in the frequently burnt savanna woodlands at Solar Village, and most of these species are flourishing broadly across the landscape.

The only experimental fire study in the plateau is that by Begg *et al.* (1981) at Little Nourlangie Rock (Nawurlandja): a plateau outlier in Kakadu. This study examined the abundance and distribution of four mammal species (sandstone antechinus, Arnhem rock-rat, common rock-rat and northern quoll) before and for a year after a single (intense) fire. For the first three species, abundance decreased markedly following fire. Based on these results, Begg *et al.* (1981) recommended that fire regimes should comprise management burns in the early to mid dry season, at intervals of at least 3–5 years.

There have been some studies that have compared biodiversity at sites subject to ongoing traditional fire management with sites without such management. The forerunner of such studies was that of Haynes (1985, 1991), who noted that there was more successful regeneration of *Callitris* in clan estates near Maningrida that were traditionally managed than in areas not subject to such management. This theme – with particular reference to *Callitris* – has been reported in several subsequent studies (e.g. Yibarbuk *et al.* 2001; Bowman *et al.* 2001a, 2004; Bowman and Prior 2004), and reinforces and corroborates demographic-based modelling. From such studies, mixed-age *Callitris* stands, with successful recruitment, are recognised as the product of, and a marker of, landscape health and benign fire management.

But healthy *Callitris* stands are not a principal objective of traditional management. Rather, they happen to be a collateral consequence of such management that aims to promote the availability of useful resources and to serve a range of other utilitarian ends. The extent to which such management provides the optimal regime for biodiversity more generally remains unresolved (as indeed is the composition of such a biodiversity nirvana). The most assertive claim for such a conjunction has been made by Yibarbuk *et al.* (2001), who reported that one traditionally managed clan estate had unusually high biodiversity values. The generality of such a claim is arguable, given its basis on but one site and the lack of explicit quantitative comparisons with comparable areas subjected to contrasting fire regimes. But although maybe not yet proven, there is obvious sense in such an assertion: millennia of Indigenous management presumably produced some sort of equilibrium for contemporary patterns of biodiversity, and withdrawal of that management will destabilise that equilibrium: inevitably with some species finding the new regime less suitable.

Indigenous knowledge of fire is intricate (see Chapter 5), and its application provides considerable opportunity for improving the management – and conservation outlook – for the plateau region. However, although such application of traditional fire management is necessary, it may no longer be sufficient. Across most of the plateau area, there has been a 50–70 year hiatus in traditional management (Bowman *et al.* 2001a; Chapter 4). This may have shifted the ecological state to a condition (characterised particularly by increase in flammable grass fuels: Bowman *et al.* 2001a) that no longer responds in the same way to customary management. Such a shift may have been catalysed or further compounded by the spread of invasive weeds and changes in vegetation structure, floristics and fuel loads caused by feral animals (Werner 2005; Petty *et al.* 2007b). The difficulty of, or advantage in, (re-)imposing traditional management on such an altered state may be further blurred in at least some areas by changes in the rationale and practice of that management: away from traditional to ‘corrective’ fire management (Lewis

1994; Bowman *et al.* 2004). The broad-scale application of traditional fire management is also compromised by the reduced population of Aboriginal people across the plateau area, the reduced need to manage the land to provide sustenance, and the erosion of traditional knowledge – although recent initiatives aim to counter all of these constraints (Chapter 11).

CONCLUSION

This chapter demonstrates the extraordinarily high value of the Arnhem Land plateau for biodiversity conservation, and notes that this value is being diminished. There is a need to identify these values more comprehensively (particularly for poorly known invertebrate groups, and more generally for biota in less-well inventoried areas of the plateau), and to establish more systematic monitoring that will provide more robust data on population trends and more insight into the factors that drive those trends.

Notwithstanding the incompleteness of our knowledge, it is clear that current fire regimes are a major driver of detrimental change in the plateau's biodiversity. To some extent, this is typical of environments in the Northern Territory or northern Australia generally, but there is also exceptionality about fire and biodiversity in the plateau. We examined this issue by developing a matrix of threatened species by threatening factors for all listed species in the Northern Territory (from Woinarski *et al.* 2007), and then tallying, for every subregion, the number of species affected by each threat. The matrix revealed that fire was by far the most frequently listed threatening factor for threatened species (particularly plants) in the plateau area (see Chapter 10), and that fire affected unusually many threatened species in the plateau area relative to most other regions in the Northern Territory (Figure 8.6). That is, fire is the most

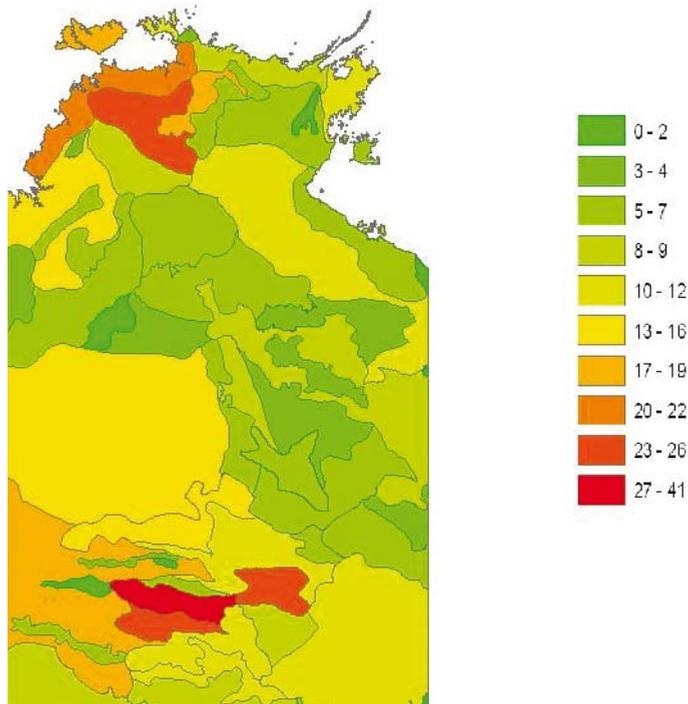


Figure 8.6 The number of threatened species for which fire is a recognised threat, for Northern Territory IBRA subregions (tallied from information in Woinarski *et al.* 2007).

significant problem for threatened species in the plateau, and that this problem is worse in the plateau area than almost anywhere else in the Northern Territory.

Most species suffering decline in the plateau are disadvantaged by the high frequency and large extent of the current fire regime. A management goal of a substantial reduction in fire frequency, especially the frequency of large (e.g. >10 km²) fires should be defined and implemented. Fire frequency data (from 1990) presented in Chapter 9 provide such an opportunity. Even with the limited knowledge base, this would be a 'no regrets' policy. This goal may be put into operation either through setting aside some substantial fire refuge areas, or reducing the overall fire frequency across the entire landscape. While noting the need for lower fire frequency and impact, other chapters consider the institutional and practical mechanisms for achieving this. In part, this can be done through resourcing Indigenous landholders to manage fire at landscape scales, informed by traditional practice.

There may be a happy conjunction in conflating the virtue of traditional fire management (with its associated implication for Indigenous employment and management direction) with optimal management for explicit biodiversity priorities. But uncritical acceptance or assertion of such symmetry will ultimately lead to disappointment for all interests, because:

- Significant components of faunal biodiversity are declining for reasons other than inappropriate fire regimes (alone)
- It is not yet clear that re-imposition of traditional modes of fire management will be sufficient to re-establish the ecological equilibrium that existed prior to European settlement
- At least some fauna species probably have habitat requirements that are poorly met by either the currently prevailing or more traditional fire regimes.

For these reasons, we recommend that management for biodiversity conservation in the plateau area complements enhanced fine-scale fire management (with targets and thresholds of reduced fire frequency) with management of other threatening factors (such as control of feral buffalo and cats, and elimination of any outbreaks of invasive pasture grasses), and that management resourcing is explicitly linked to biodiversity outcomes and is of a magnitude commensurate with the area's international significance for biodiversity.

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