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Assessing ecological performance thresholds in fire-prone Kakadu National Park, northern Australia

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Abstract. Contemporary fire regimes are recognized as a key threatening process for relatively immobile vertebrates and narrowly dispersing obligate seeder plant taxa in fire-prone Australian savannas. Here, we assess the efficacy of proposed ecological performance threshold metrics for evaluating the current state of fire management for biodiversity conservation outcomes in Australia's premier, and best publicly funded savanna reserve, Kakadu National Park. The assessment draws on available data describing Landsat-scale fire mapping over the period 1997–2015, habitat mapping, and mostly modeled responses of vegetation and faunal attributes. Despite conceptual and technical issues associated with various proposed performance thresholds and mapping products, the assessment demonstrates significant challenges with the current state of the reserve's fire management program. For example, by the end of 2015 it was observed that just 6% of woodland habitat in lowland and 23% in upland situations had remained unburnt for longer than three years and 98% of mapped fires in lowland and 87% in upland habitats were >1 km² in extent. Of 14 assessed performance threshold metrics, two were within acceptable thresholds at the end of 2015, and none had improved materially over the decadal assessment period. Given substantial resources evidently required to deliver effective, seasonally intensive, fine-grained adaptive fire management for biodiversity conservation outcomes in fire-prone Australian savannas, we suggest that alternative resourcing opportunities through market-based savanna burning greenhouse gas emissions abatement projects need to be explored.

Key words: adaptive management; biodiversity conservation; ecological threshold; fire management; fire regime; Kakadu National Park; long-term monitoring.

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INTRODUCTION

Savannas are the most fire-prone systems on Earth. Despite significant understanding of the role of fire regimes in regulating the composition, structure, and dynamics of savanna systems (Scholes and Archer 1997, Williams et al. 2002, Sankaran et al. 2004, Higgins et al. 2007, Lehmann et al. 2014), delivering effective fire

management for savanna biodiversity conservation purposes remains ecologically and operationally complex, uncertain, and challenging (van Wilgen et al. 2011, 2014, Russell-Smith et al. 2015). Ecological complexities reflect notably, but are not restricted to, understanding interactions between fire regime elements (e.g., fire frequency, intensity, interval, patchiness) and the responses of grazers and browsers (Bond and Archibald

2003, Holdo et al. 2009, Vanak et al. 2012), the habitat requirements of other faunal components (Andersen et al. 2005), and managing for woody plant encroachment (O'Connor et al. 2014, Smit et al. 2016) and other invasive plant species (Setterfield et al. 2010). In Australian contexts particularly, a key operational challenge includes implementing cost-effective, highly patchy fire management (at hectare scales) for narrowly dispersing obligate seeder plant taxa with long maturation periods, and relatively immobile vertebrates with small home ranges (Yates et al. 2008, Ziembicki et al. 2015).

An associated challenge for informing and guiding site-based fire management for biodiversity conservation purposes is the development of effective active adaptive management frameworks (sensu Walters 1986). For fire-prone savannas, the evolution of adaptive fire management policy in Kruger National Park, South Africa, over the past 20 yr is especially instructive (van Wilgen et al. 1998, 2014, Biggs and Rogers 2003, van Wilgen 2009). Significant changes to Kruger National Park fire management policy have occurred through this period: an emphasis on “natural” lightning ignitions from 1991 to 2001; combining point ignitions with natural and unplanned fires from 2002 to 2011 to meet burnt area targets assessed at the park-wide scale; and a current emphasis on delivering ecological outcomes in relation to five management zones defined by rainfall, historic fire patterns, and geology. Further, informed by substantial ongoing landscape-scale fire research, fire regime monitoring criteria (described as “thresholds of potential concern”) now explicitly reflect ecological management objectives as opposed to previous criteria based on fire regime patterning alone (van Wilgen et al. 2014).

The “thresholds of potential concern” concept has been influential in informing Australian biodiversity conservation management (Bradstock et al. 2012, Keith et al. 2014), including in Australia’s premier savanna protected area, Kakadu National Park (Australian Government 2016). Following extensive fires in fire-sensitive shrubland heath habitats in Kakadu’s rugged sandstone uplands in 2004 and 2006, a “sandstone country” fire plan incorporating explicit ecological fire frequency and interval thresholds for associated vegetation communities was developed (Petty et al.

2007a). Assessment of the efficacy of Kakadu’s sandstone country fire management program over the period 1984–2011 found that by 2011, the area of each of three major habitat types (monsoon rain forest, woodland, sandstone heath) experiencing acceptable (sensu Petty et al. 2007a) thresholds (fire interval; unburnt for ≥ 5 yr) was greater than at any earlier point in the fire record (Murphy et al. 2015). Although these thresholds criteria relate specifically to vegetation attributes, complementary criteria recently have been proposed for threatened regional savanna fauna (Woinarski and Legge 2013, Woinarski and Winderlich 2014).

In this paper, we report on and discuss the efficacy of applying proposed ecological thresholds criteria and associated metrics for assessing the current state of fire management for biodiversity conservation objectives in Kakadu National Park. For this assessment, we draw on available datasets describing (1) the fire history of the park based on mapping from relatively fine-scale Landsat imagery over the period 1997–2015; (2) mapping of major habitat and landscape units; and (3) modeled responses of key vegetation and faunal attributes to fire regime variables derived mostly from the park’s long-term (20-yr) fire effects monitoring program. This assessment is timely in that it coincides with the commencement of the park’s new decadal (2016–2026) management plan (Australian Government 2016) and, notwithstanding the positive benefits of the sandstone country fire management program described above, ongoing significant conservation management challenges associated with fire-vulnerable species and communities in Australia’s fire-prone savannas generally (Woinarski et al. 2010, Russell-Smith et al. 2015, Ziembicki et al. 2015).

MATERIALS AND METHODS

Regional context

Kakadu National Park (19,810 km²) is located in a coastal to sub-coastal region of the Top End of the Northern Territory, Australia (Fig. 1). Jointly managed by its traditional Aboriginal owners and an Australian Government conservation agency, the park is inscribed in the World Heritage list both for its cultural and natural values. The Kakadu region is recognized

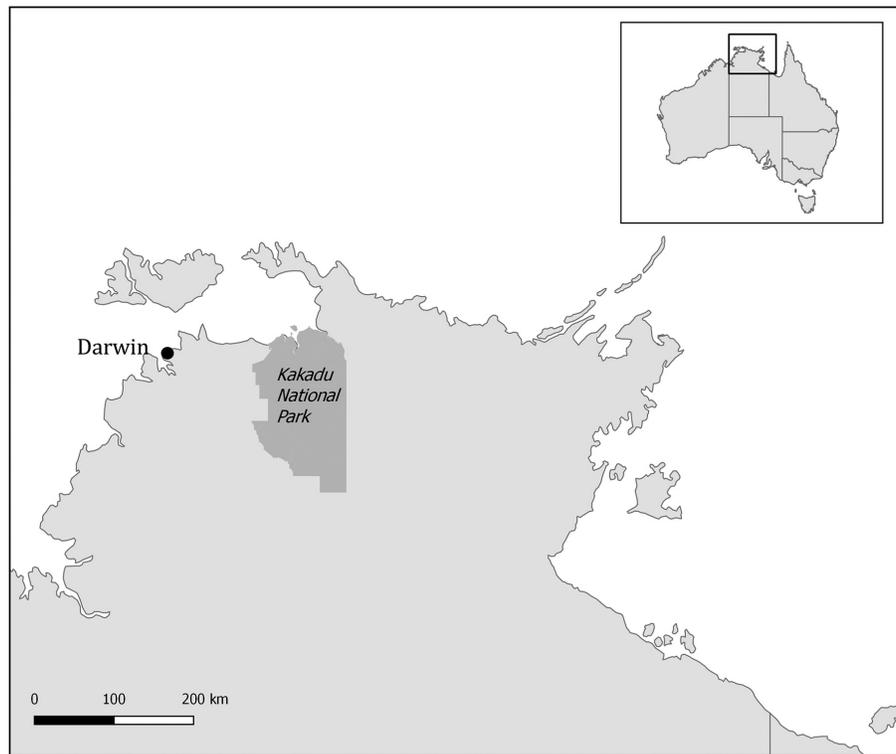


Fig. 1. Location of Kakadu National Park.

internationally as a center of plant and faunal diversity and endemism (Ingwersen 1995, Crisp et al. 2001, Woinarski et al. 2009).

The regional climate is characterized by marked rainfall seasonality, with over 90% occurring in the summer, wet season months, November through March. Mean annual rainfall declines from over 1500 mm in the northwest to ~1200 mm in the southeast. Although the amount of rainfall received in any one area is highly variable from year to year, the wet season is highly reliable (Taylor and Tulloch 1985).

Throughout this paper, reference is made to three major landform units (Fig. 2a). The topographically dominant regional landform is the rugged Arnhem Land plateau, mostly at <400 m elevation, referred to hereafter as the “upland” unit. The uplands comprise resistant, flat-bedded Middle Proterozoic quartzose sandstones, crisscrossed by tensional joints which have been deeply weathered and eroded to form a maze of narrow valleys and gorges (Galloway 1976). Soils, where present, are typically skeletal and infertile sands (Aldrick 1976). Especially on its

northwestern and western perimeters, the uplands are bounded by sheer and spectacular escarpments. The two major lowland landform units comprise an undulating Cainozoic plain which stretches away from upland margins comprising deeply weathered (laterized), predominantly coarse-grained sediments; and extensive coastal/sub-coastal Holocene floodplains associated with the lower reaches of major river systems, comprising mostly fine-grained freshwater and estuarine sediments (Williams 1991).

The majority of the upland sandstone plateau vegetation comprises eucalypt-dominated savanna on shallow to deeper soils, interspersed with shrubby heath vegetation comprising a majority of obligate seeder taxa and flammable hummock grasses (e.g., *Triodia*) especially in skeletal rocky terrain (Fig. 2c). *Callitris* groves are common in relatively fire-protected sites. Although most shrub species are seeders which attain sexual maturity within three years following fire, many sites support seeder shrubs with substantially longer maturation periods (Russell-Smith et al. 1998, 2002, Russell-Smith 2006).

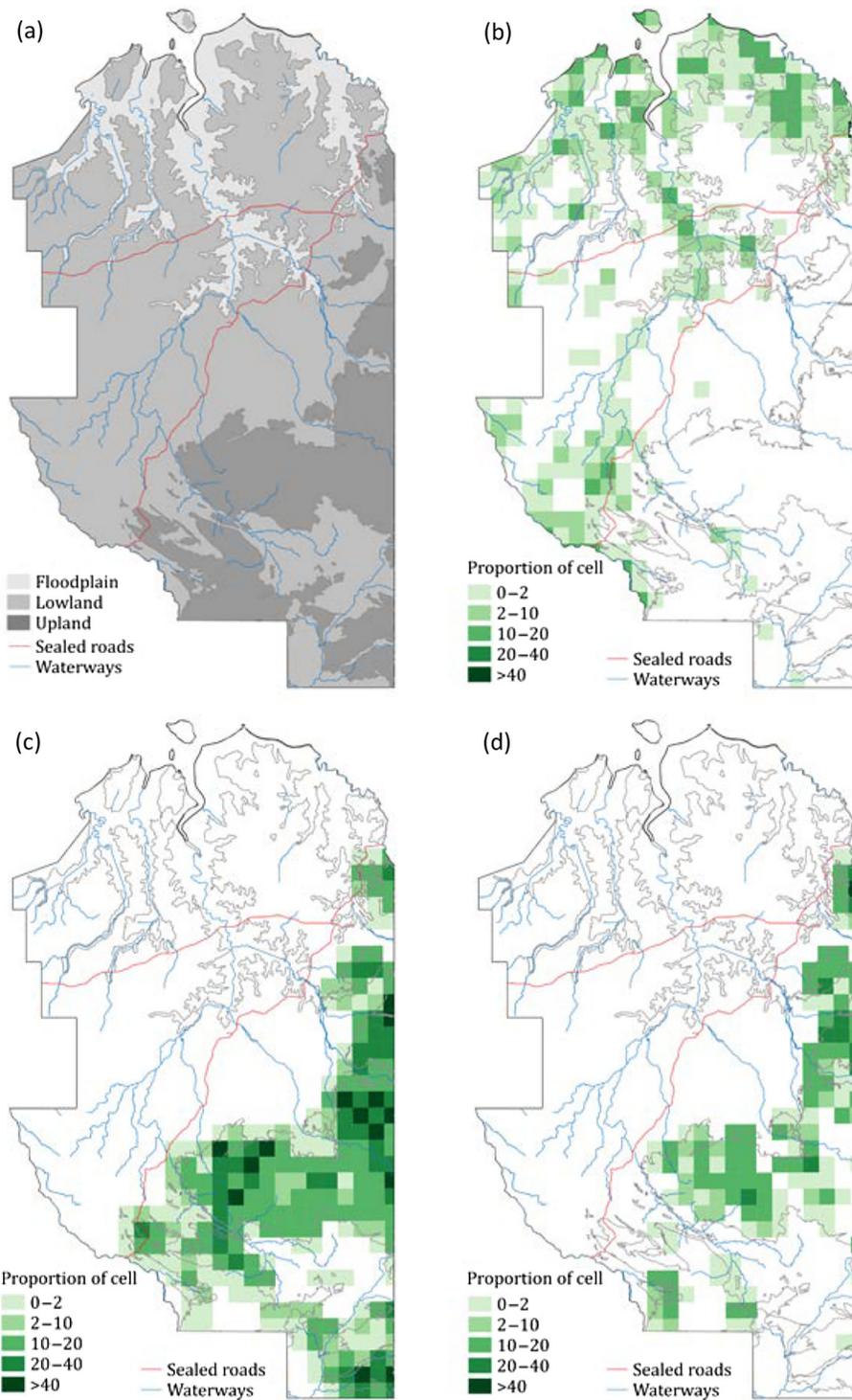


Fig. 2. Distribution of major habitats: (a) floodplain, lowland, and upland landform units; (b) lowland monsoon rain forests; (c) sandstone heaths; and (d) upland monsoon rain forests. Distributions of relatively small habitat areas (b, c, d) are displayed as the proportion of 5×5 km cells occupied by respective habitats.

Uplands also support substantial tracts (>100 ha) of an endemic monsoon rain forest type dominated by the hardy semievergreen myrtaceous resprouter, *Allosyncarpia ternata* (Fig. 2d). These forests occur typically in rugged, broken terrain affording some level of fire protection particularly along the northwestern rim of the Arnhem Plateau, but also as small isolates in occasional non-protected situations (Russell-Smith et al. 1993). Large *Allosyncarpia* stems (often >100 cm diameter at breast height [dbh]) typically are hollowed out by termite activity and susceptible to fire incursions on exposed patch margins (Prior et al. 2007).

Vegetation cover of the lateritic lowlands comprises mostly eucalypt (*Eucalyptus*, *Corymbia*)-dominated open forest or woodland over a typically grassy understory (Wilson et al. 1996). Eucalypts comprise the major proportion of the regional savanna stem basal area, but only a quarter of component tree species richness (Lawes et al. 2011). While most lowland tree and shrub taxa are relatively fire-resilient and resprout epically (e.g., all eucalypts) or at least basally (typically non-eucalypts) following severe fires, the savanna matrix also includes scattered copses of the long-lived, fire-sensitive obligate seeder conifer, *Callitris intratropica*. Mortality of *Callitris* usually results following canopy-scorching fires (Bowman and Panton 1993).

Small patches of lowland closed monsoon rain forest (typically <5 ha) occur in association with small springs and perennial streams, sites offering fire protection (e.g., rocky outcrops, beach ridges, river levees), and especially along the margins of riverine and coastal floodplains (Fig. 2b). These assemblages are particularly vulnerable to severe fires on exposed boundaries with high graminoid fuel loads (Russell-Smith and Bowman 1992).

Lowland floodplain communities range from fire-prone open sedgelands/grasslands to tall open forests dominated by *Melaleuca* associated with freshwater facies, to fire-protected samphires and mangroves under saline influence. In the latter part of the dry season, old-growth stands of *Melaleuca* are vulnerable to sub-surface smoldering peat fires.

Prevailing climatic conditions are conducive for rapid development of grassy fuels such that fires may recur within 1–2 yr (Williams et al.

2002). Fire severity generally increases with the progression of the dry season because of increasingly severe fire weather (stronger winds, higher temperatures, lower humidity), and lower fuel moisture conditions (Russell-Smith and Edwards 2006). Even in skeletal sandstone terrain, accumulation of fine grass and litter fuels is sufficient to support intense fires (>5000 kW/m) in all but exposed rock situations under late dry season climatic conditions within 1–3 yr of having been burnt previously (Russell-Smith et al. 1998).

Fires in the region are almost invariably anthropogenic in origin (Russell-Smith et al. 2007), and ground-borne, although lightning ignitions are associated with electrical storms at the start of the wet season. Contemporary fire regime patterns date from the early–mid-twentieth century associated with the breakdown of traditional Aboriginal modes of landscape management, followed by more recent attempts to implement fire management based largely on the traditional model (Russell-Smith et al. 2009, Lawes et al. 2015). Kakadu fire management policy emphasizes the undertaking of prescribed strategic burning in the early dry season (EDS) period under relatively mild fire-weather conditions (typically before August), as a means for restricting the incidence and extent of late season unplanned wildfires (Australian Government 2016). Nonetheless, contemporary regional fire regimes characterized by frequent, severe, and extensive fires are incurring significant impacts on fire-sensitive vegetation types (Bowman and Panton 1993, Russell-Smith et al. 2012) and are likewise implicated in the collapse of the small-mammal fauna (Woinarski et al. 2009, 2010, Griffiths et al. 2015, Lawes et al. 2015, Ziembicki et al. 2015).

Until the early 1990s, densities of browsing and grazing feral animals (especially Asian water buffalo, cattle) were uncontrolled and high in many locations (especially floodplains, riparian corridors, and adjacent habitats), but controlled and consistently low thereafter (Petty et al. 2007b).

Ecological thresholds

The development of ecological thresholds criteria for informing and guiding fire management is a requirement of the current Kakadu 2016–2026 Management Plan (Australian Government

2016). A guiding principle for establishing these criteria is that they need to parsimoniously represent, and reliably and cost-effectively measure, prioritized ecological management objectives as identified in the management plan. The Kakadu management plan also sets out a range of related requirements that need to be considered as part of developing a fire management strategy for the park, including Indigenous engagement, employment, cultural requirements and aspirations for appropriate fire management to maintain healthy country, as well as operational and regulatory requirements. These issues are not addressed in this assessment.

The criteria and associated metrics assessed here (Table 1) have been developed based on substantial research undertakings, particularly as outputs from (1) analyses of Kakadu's long-term fire effects monitoring program (e.g., Russell-Smith et al. 2010, 2012, Woinarski et al. 2010, Lawes et al. 2015, Murphy et al. 2015), and (2) a variety of key pertinent north Australian studies (e.g., Bowman and Panton 1993, Russell-Smith et al. 1993, 1998, Andersen et al. 2005, Woinarski et al. 2005, Yates et al. 2008, Edwards and Russell-Smith 2009, Radford 2012, Woinarski and Legge 2013). The criteria also need to meet related park management and monitoring requirements, especially those associated with monitoring habitat health in line with the park's threatened species and communities management strategy (Woinarski and Winderlich 2014).

Three performance thresholds (effects of severe fires on (1) monsoon rain forest and (2) woodland habitats; (3) effects of frequent fires on sandstone heath; Table 1) apply metrics based on statistically significant simple linear response models derived from long-term monitoring plots established as part of the Three Parks (Kakadu, Litchfield, and Nitmiluk National Parks) fire effects monitoring program (Russell-Smith et al. 2010, 2012, 2014). Respective models are based on 15 yr of observations (1995–2009) from 48 plots located in Kakadu's sandstone uplands (Russell-Smith et al. 2012). Although the woodland performance threshold metric is based foremost on the modeled effect of severe fires on *Callitris* stems, this metric more broadly encapsulates the effects of frequent severe fires on woodland stem densities generally. For example, statistically significant relationships were

observed between the variable "time since burnt by a severe fire" with densities of saplings/tall shrubs (200 cm tall to <5 cm dbh) both in the Kakadu upland study and in an earlier larger study involving 10 yr of observations based on 122 Three Parks woodland savanna plots (Russell-Smith et al. 2010). We assume that for long-term sustainability, thresholds criteria for these response groups (and for fire size criteria below) should not be exceeded by more than 10% over any 5-yr period.

Two further key performance thresholds relate to maintaining significant areas of relatively long unburnt habitat (particularly for dependent small mammals and birds), and for maintaining the sizes of burnt patches within spatial scales compatible with the restricted home range and dispersal limitations of many small vertebrates and plant taxa, respectively (Table 1). As addressed in *Discussion*, challenges are associated with defining appropriate ecological thresholds in both instances, and with significant technical measurement issues in the latter.

Based on decadal-scale fire mapping for Kakadu, Litchfield, and Nitmiluk National Parks, Andersen et al. (2005:162) observed that there was a "serious lack of long unburnt habitat catering for relatively fire-sensitive species"—with ~10% of lowland savannas remaining unburnt for at least 3 yr, and <3% remaining unburnt after 10 yr. Although critical ecological assessments of minimum unburnt area threshold requirements for maintaining fire-sensitive vertebrate fauna are currently lacking, for savanna birds Woinarski and Legge (2013:343) propose that "(1) at least 25% of the savanna landscape... is at least 3 yr unburnt and, in some contexts, such as rocky landscapes... the target proportion should be much higher, [and] (2) at least 5% is at least 10 yr unburnt (and again, in some contexts this proportion should be much higher)." In the absence of other data, we apply minimum unburnt area thresholds following Woinarski and Legge (2013), and as recommended by Woinarski and Winderlich (2014) for application in Kakadu.

With respect to fire size, various studies concerning individual fauna taxa demonstrate that required spatial scales of burning (patch sizes) are often substantially less than threshold sizes addressed here (patches <1 km²), for example,

Table 1. Ecological basis for determining quantitative performance thresholds.

Response group	Fire regime vulnerability	Performance threshold
1. Monsoon rain forest	<ul style="list-style-type: none"> • Comprising mostly resprouter species (Russell-Smith et al. 1998), regional monsoon rain forests are nonetheless vulnerable especially on exposed margins to severe fires 	<p><10% of monsoon rain forest boundaries should be affected by severe fires in any 5-yr period</p>
1.1 Lowland monsoon rain forest		
1.2 Upland monsoon rain forest	<ul style="list-style-type: none"> • This applies particularly to upland monsoon rain forests dominated by <i>Allosyncarpia ternata</i>, whose large stems are often scarred from previous fires, and/or hollowed by termites (Russell-Smith et al. 1993, Prior et al. 2007) • Modeling shows that large <i>Allosyncarpia</i> stems (>30 cm dbh) are susceptible to severe fires (Russell-Smith et al. 2012) 	
2. Woodland	<ul style="list-style-type: none"> • Long-lived, fire-vulnerable obligate seeder tree, <i>Callitris intratropica</i>, occurring in stands within woodland matrix, with juvenile periods likely to substantially exceed 10 yr in many situations (Bowman and Panton 1993, Russell-Smith 2006) • Modeling shows that <i>Callitris</i> stems (>5 cm dbh) vulnerable to severe fires at frequencies >0.2 (Russell-Smith et al. 2012) • Similar negative effects of frequent, especially severe, fires are observed on densities of woodland saplings (>200 cm tall and <5 cm dbh) in regional modeling studies (Russell-Smith et al. 2010, 2012) 	<p><10% of woodland habitats should be affected by two or more severe fires in any 5-yr period</p>
2.1 Lowland woodland		
2.2 Upland woodland		
3. Sandstone heath	<ul style="list-style-type: none"> • Comprises a majority of obligate seeder shrub taxa (Russell-Smith et al. 1998) • Most taxa with juvenile periods ≤3 yr, but others ≤5 yr, and a few (including some <i>Acacia</i> spp., and the serotinous <i>Petraecomyrthus punicea</i>) c. 10 yr (Russell-Smith et al. 1998, 2002, Russell-Smith 2006) • Modeling shows that longer-maturing (3+ yr) obligate seeder shrub taxa vulnerable to fires at annual frequencies >0.2 (Russell-Smith et al. 2012) 	<p><10% of sandstone heath habitats should be re-burnt within a 5-yr period</p>
3.1 Sandstone heath		
4. Savanna woodland small mammals and many bird taxa	<ul style="list-style-type: none"> • Fire-mediated habitat heterogeneity, including the development of species-diverse shrub and mid-canopy food resources, is considered critical for many small mammals and bird taxa (Andersen et al. 2005, Woinarski et al. 2005) • Evidence indicates that such fire-mediated habitat diversity is lacking in Kakadu woodlands (e.g., Andersen et al. 2005, 2012) • Although modeling of the required proportion of fire-prone habitat that should remain unburnt for effective conservation of vulnerable taxa is unavailable, recommended guidelines are provided by Woinarski and Legge (2013) and Woinarski and Winderlich (2014) • Strategic prescribed burning has been shown to deliver effective long-unburnt habitat outcomes in Kakadu sandstone habitats (Murphy et al. 2015) 	<ul style="list-style-type: none"> • At least 25% of fire-prone floodplain and lowland woodland, and 40% of upland woodland habitats, should remain unburnt for 3 yr • At least 5% of floodplain and lowland woodland habitats, and 10% of upland woodland habitats, should remain unburnt for 10 yr
5. Fauna with restricted home ranges, and narrowly dispersed obligate seeder plants	<ul style="list-style-type: none"> • Many small mammals and invertebrates with restricted home ranges (e.g., <1–10 ha) are vulnerable to large fires (Fraser et al. 2003, Woinarski et al. 2005, Yates et al. 2008, Radford 2012) • Limited dispersal capacity is also exhibited by many component obligate seeder taxa, especially those possessing non-dormant seed banks (Russell-Smith 2006, Yates et al. 2008) • Although few pertinent studies exist exploring relationships between fire size and population response, Radford (2012) showed that low-intensity fires <1 km wide had little impact on small mammals, and Lawes et al. (2015) observed that fires <10 km² had limited effect on small mammals, but those >10 km² resulted in catastrophic declines 	<p><10% of all major landscape units should be burnt by individual fires >1 km² in extent within a 5-yr period</p>

≪1 ha for dry season nymph colonies of the spectacular Leichhardt's grasshopper, *Petasida ephippigera* (Lowe 1995, Barrow 2009); and <10 ha for many small frugivorous and granivorous birds (Franklin 1999, Fraser et al. 2003, Woinarski and Legge 2013), and many small marsupials and rodents (Kerle 1998, Oakwood 2002, Woinarski et al. 2005, Firth et al. 2006, Hohnen et al. 2015, 2016). Associated technical challenges include the capacity to measure the sizes of individual fires or, as pragmatically assessed here, contiguously burnt areas over the seasonal cycle.

Spatial datasets

For the purposes of assessing the status of performance thresholds (Table 1) in 2015, and to track respective annual trends over the preceding nine years from 2006, annual fire mapping covering the period 1997–2015 was assembled for the entire park. Annual fire mapping was derived from Landsat satellite imagery (rescaled to 25 m pixels), sampled at least three times annually as follows: a first image obtained ideally early in the dry season (approximately late May/early June), a second obtained around the end of the main burning early season period (late July/early August), and a third image obtained as late in the year as possible before the onset of extensive cloudy conditions associated with the developing wet season. To account for cloudiness issues in the late dry season period, Landsat imagery was augmented with coarser-resolution, daily imagery available from Advanced Very High Resolution Radiometer (1.1 km pixels) and, since 2000, Moderate Resolution Imaging Spectroradiometer (MODIS, 250 m pixels) sensors. Fire mapping data have been validated annually based on stratified aerial transect assessments. Methodological details, including for validation assessments, are given in Edwards et al. (2003) and Russell-Smith et al. (2009, 2012).

Lowland and sandstone upland land units were derived from an available 1:250,000 vegetation mapping coverage (Schodde et al. 1987). Floodplains, excluding mangroves and samphires (after Schodde et al. 1987), were delineated from lowlands based on high water inundation level mapping (Ward et al. 2014). With the exception of lowland monsoon rain forests, mapping of all other major vegetation structural types was

derived from Landsat TM imagery. Mapping (~1:100,000) of sandstone woodlands and heaths and lowland woodlands (using 2008 and 2014 imagery, respectively) was compiled using an object-based classification approach, following procedures (including validation) detailed in Edwards and Russell-Smith (2009). Mapping of sandstone monsoon rain forests (~1:25,000) was derived from 2005 Landsat TM imagery by on-screen digitizing (J. Freeman, *personal communication*). Mapping of lowland and floodplain monsoon rain forest patches was undertaken by mapping (1:100,000) manually from 1980s 1:25,000 aerial photographs and subsequent digitization (Russell-Smith 1991).

With the exception of lowland monsoon rain forest mapping (addressed in *Discussion*), we consider the accuracy of all other vegetation mapping surfaces applied in this assessment to be generally reliable as follows: upland monsoon rain forest—>98% based on recent independent validation using 295 km of high-density aerial videography (J. Freeman, *personal communication*); 89% for upland woodland and sandstone heath structural mapping (Edwards and Russell-Smith 2009); and >80% for lowland woodlands (Darwin Centre for Bushfire Research, *unpublished manuscript*).

Analyses

Using standard geographic information system techniques, fire metric values for each performance threshold (Table 1) were calculated for respective habitats as a whole and mapped spatially with respect to relevant 5 × 5 km grid cells. In the absence of a separate fire severity layer, following procedures outlined in Russell-Smith et al. (2012) we applied previously established relationships between fire seasonality and severity derived from 10 yr (1995–2004) of plot-based observations ($n = 719$; Russell-Smith and Edwards 2006). For calculations involving fire severity, we assumed the probability of a severe fire as 0.28 in the EDS and 0.81 in the late dry season (LDS) for all upland habitats, and 0.19 and 0.78, respectively, for lowland habitats. Fire effects on monsoon rain forest boundaries were assessed with reference to a 50-m (i.e., two-pixel) buffer encompassing the boundary pixel plus that on the inner closed forest margin. For respective habitat types, fire patches were defined as areas of contiguously burnt pixels,

including those on diagonals, burnt throughout any one year; as such, any one patch can represent the cumulative effects of a number of individual fires.

RESULTS

By the end of 2015, only two key performance thresholds (both relating to maintenance of long-unburnt floodplain habitat; see 4.1 in Table 2, Fig. 3) of 14 addressed here had been met consistently over the preceding decadal period. One other performance threshold (concerning upland

woodland) was achieved at least partially over the decadal assessment period (see 4.3 in Table 2, Fig. 3). The spatial distribution (in 5 × 5 km cells) of performance thresholds illustrates that in the five-year period ending 2015, fire thresholds were exceeded generally uniformly throughout the lowlands, whereas thresholds in upland habitats were exceeded especially in southwestern and southern sectors (Fig. 4).

With respect to the fire patch-size threshold, assessment of the underlying fire patch-size distribution illustrates that although 5-yr means of 91% of mapped fire patches in lowland

Table 2. Observed responses vs. performance threshold criteria.

Response variable	Performance threshold	Observed response	
		2015†	2006–2015
<i>1. Severe fire impacting monsoon rain forest boundary</i>			
1.1 Lowland monsoon rain forest	<10% should be affected by one severe fire in 5 yr	57%	Consistently >40%
1.2 Upland monsoon rain forest	<10% should be affected by one severe fire in 5 yr	40%	Consistently >15%
<i>2. Severe fire affecting fire-prone woodland habitat</i>			
2.1 Lowland woodland	<10% should be affected by two or more severe fires in 5 yr	37%	Consistently >20%
2.2 Upland woodland	<10% should be affected by two or more severe fires in 5 yr	24%	Consistently >10%
<i>3. Frequent fire affecting fire interval-sensitive sandstone heath</i>			
	<10% should be affected by more than one fire in five years	75%	Consistently >40%
<i>4. Long-unburnt habitat for fire-vulnerable small mammals and birds</i>			
4.1 Floodplain			
	At least 25% should remain unburnt for 3 yr	Achieved 41%	Achieved: consistently >40%
	At least 5% should remain unburnt for 10 yr	Achieved 15%	Achieved: consistently ≥15%
4.2 Lowland woodland			
	At least 25% should remain unburnt for 3 yr	6%	Consistently <25%
	At least 5% should remain unburnt for 10 yr	1%	Consistently <5%
4.3 Upland woodland			
	At least 40% should remain unburnt for 3 yr	23%	Achieved for 6 yr
	At least 10% should remain unburnt for 10 yr	7%	Consistently <10%
<i>5. Small fire sizes for fauna with restricted home ranges, and narrowly dispersed obligate seeder plants</i>			
5.1 Floodplain			
	<10% should be burnt by fires >1 km ² in extent within a 5-yr period	67%	Consistently >50%
5.2 Lowlands			
	<10% should be burnt by fires >1 km ² in extent within a 5-yr period	98%	Consistently >95%
5.3 Uplands			
	<10% should be burnt by fires >1 km ² in extent within a 5-yr period	87%	Consistently >65%

† At end of 5-yr assessment period.

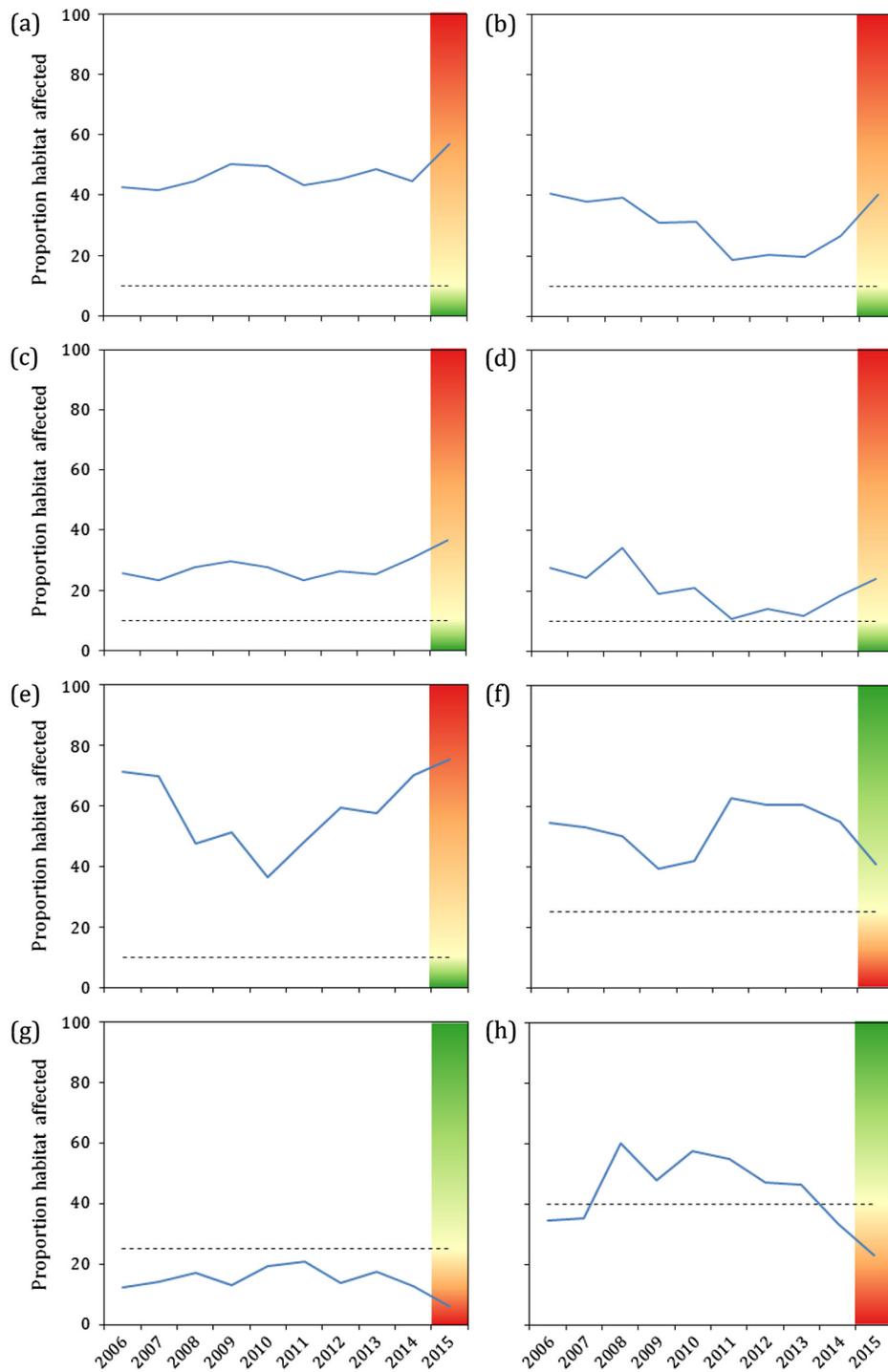
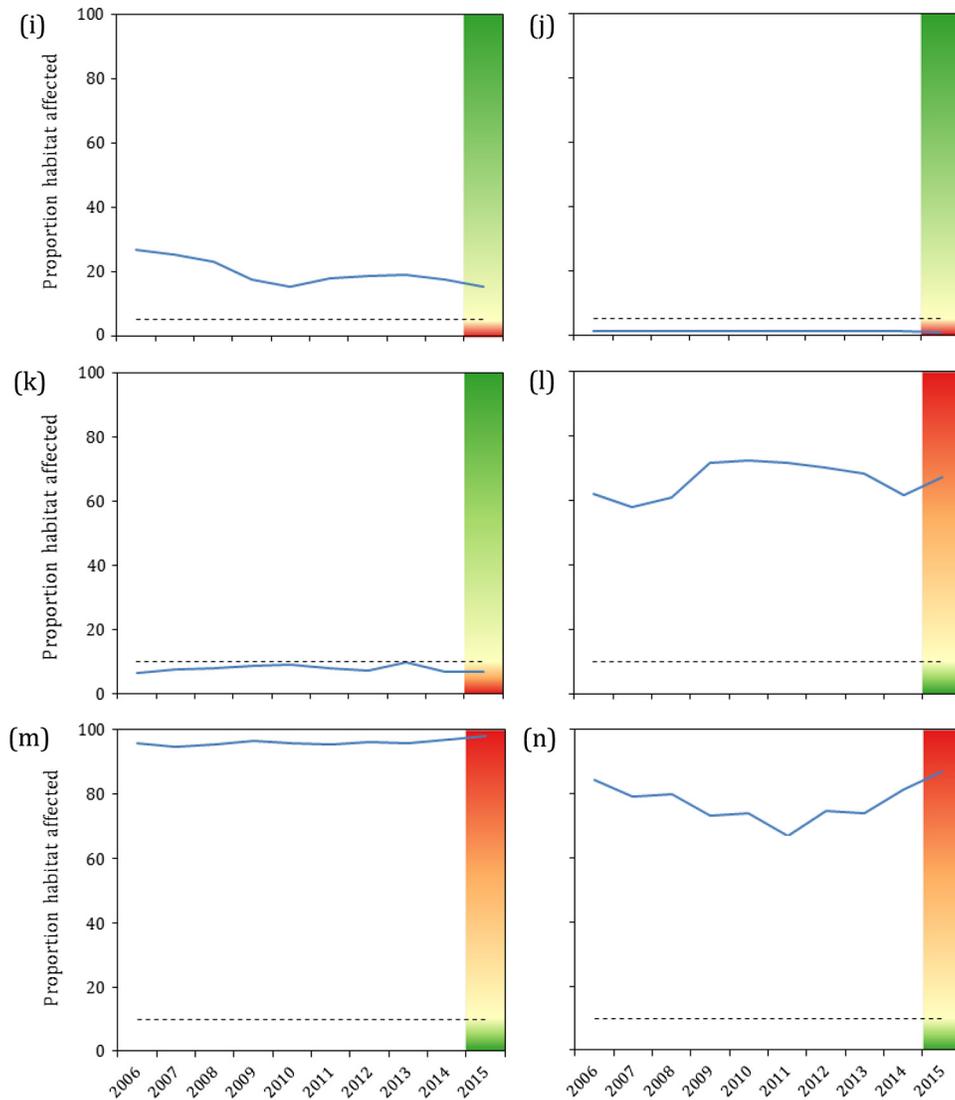


Fig. 3. Ten-year trends for respective ecological performance threshold metrics, for (a) severe fires affecting lowland monsoon rain forest, (b) severe fires affecting upland monsoon rain forest, (c) severe fires affecting lowland woodland, (d) severe fires affecting upland woodland, (e) frequent fires affecting sandstone heath, (f) floodplain habitat unburnt for at least 3 yr, (g) lowland woodland unburnt for at least 3 yr, (h) upland woodland unburnt for at least 3 yr,



(Fig. 3. Continued)

(i) floodplain vegetation unburnt for at least 10 yr, (j) lowland woodland unburnt for at least 10 yr, (k) upland woodland unburnt for at least 10 yr, (l) floodplain habitat burnt by fires <1 km², (m) lowland habitat burnt by fires <1 km², and (n) upland habitat burnt by fires <1 km². Dotted line indicates threshold, with green colored ramping indicating that performance threshold is within acceptable limits, and red colors indicating that it is not.

woodland ($n = 2563$) and 91.7% of fire patches in upland woodland ($n = 673$) were <1 km², these contributed just 1.9% and 1.6% of mean fire-affected area in respective habitats (Fig. 5).

DISCUSSION

It is axiomatic that adaptive management for biodiversity conservation purposes needs to be

guided by clear, measurable, and agreed objectives and targets, and cost-effective monitoring processes (Walters 1986, Williams 2011). However, seemingly straightforward targets may not always be either unambiguous or appropriate, nor readily measured. Here, we first assess methodological issues and ecological assumptions underpinning fire management performance targets under consideration for Australia’s premier savanna reserve.

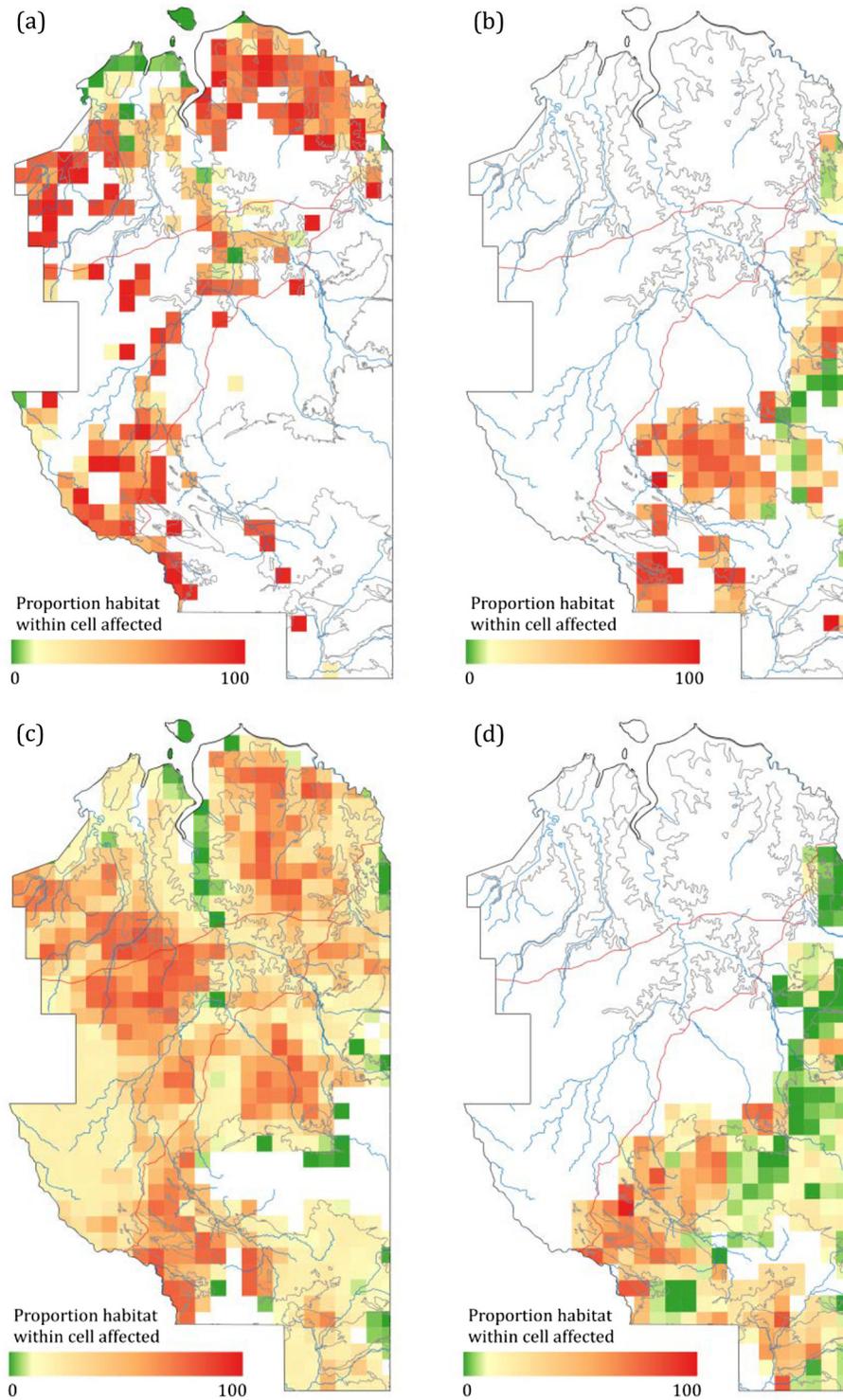
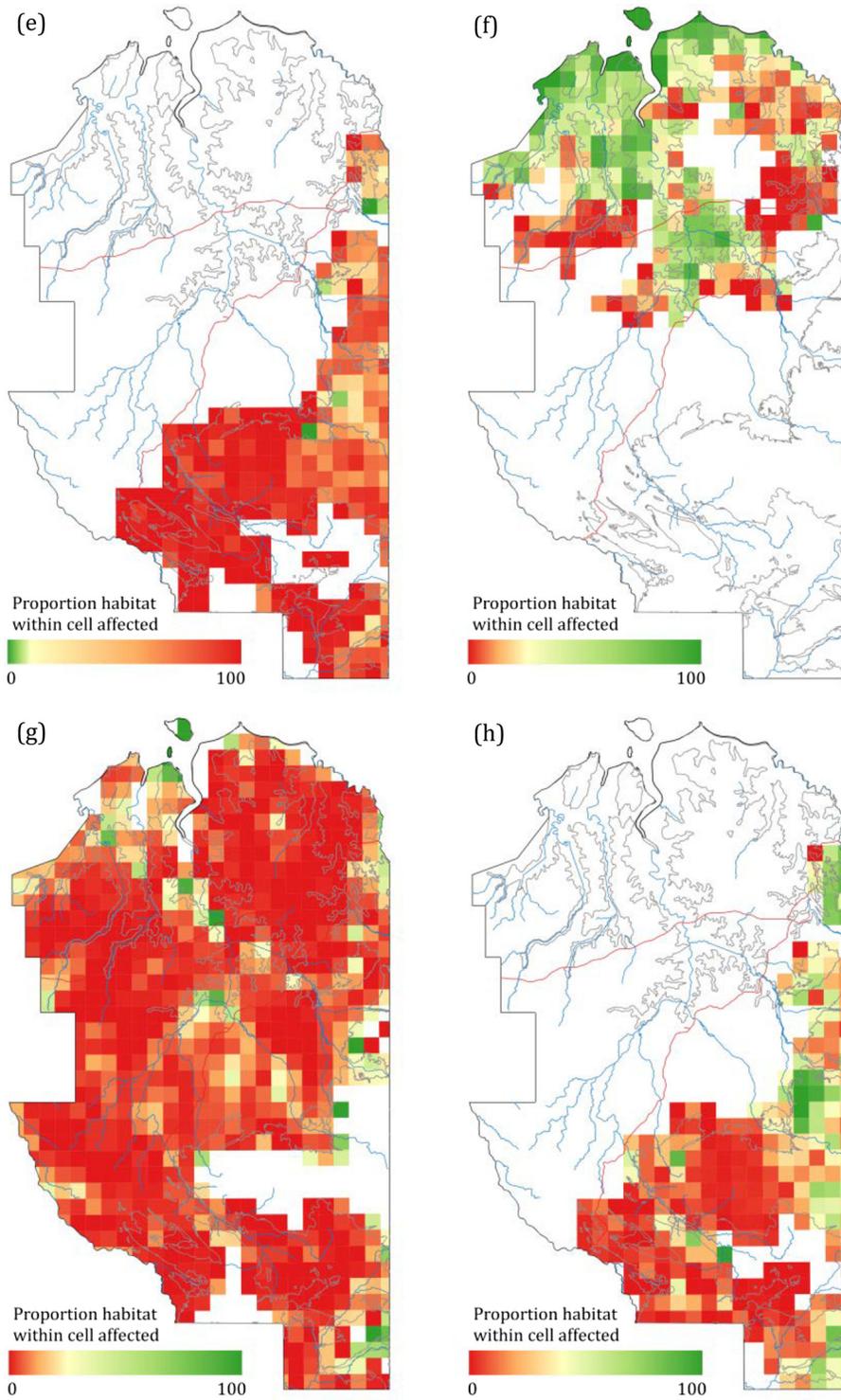
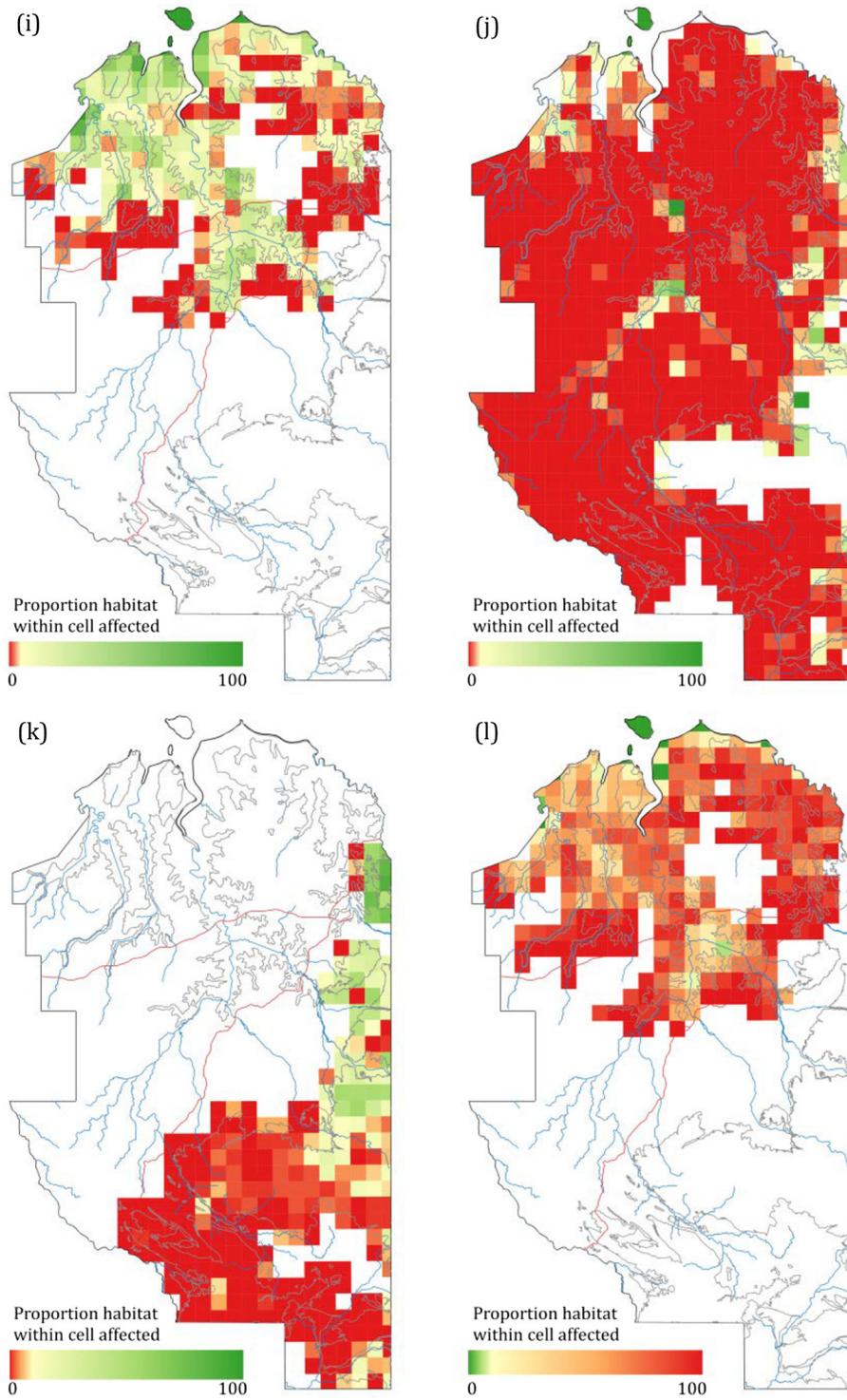


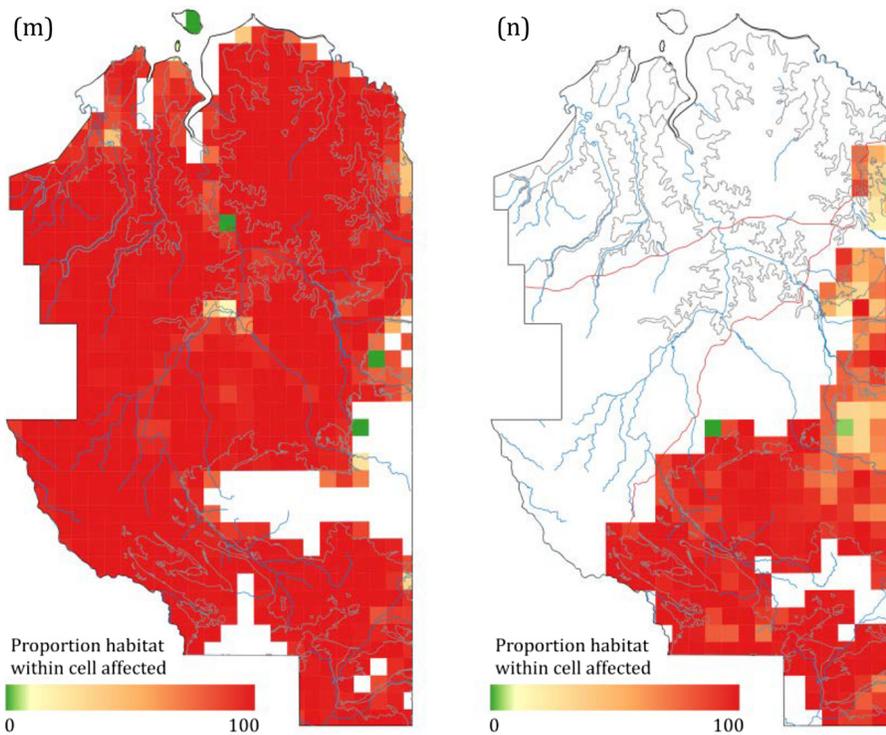
Fig. 4. Status and geographic distribution of ecological performance threshold metrics at end of 2015 in relevant 5×5 km cells. Labels (a–n) as per Fig. 3. Green colored ramping indicating that performance threshold is within acceptable limits, and red colors indicating that it is not.



(Fig. 4. Continued)



(Fig. 4. Continued)



(Fig. 4. Continued)

We then consider the broader fire management challenges both for Kakadu specifically, and fire-prone savanna reserves across northern Australia more generally, posed by assembled data.

Mapping and performance thresholds

While we consider levels of mapping reliability for most habitats to be sufficient for our assessment purposes (see *Materials and methods*), we acknowledge that potentially large mapping inaccuracies are likely to be involved with the locations of lowland monsoon rain forest boundaries, attributable to the following: changed boundary conditions since the capture of the underpinning 1980s aerial photography, especially given the dynamism of this forest type (Russell-Smith 1991, Banfai and Bowman 2006); and more particularly, substantial challenges involved with delineating mature “core” lowland monsoon rain forest from surrounding seral and ecotonal elements (Bowman 1992, Banfai and Bowman 2006). By contrast, the margins of regional upland monsoon rain forests, dominated almost monotypically by the evergreen tree *Allosyncarpia ternata*, are characteristically

abrupt (Bowman 1991, Russell-Smith et al. 1993), slow-growing (Russell-Smith et al. 2009, 2012), and hence readily mapped. We note however that for monitoring and deciphering monsoon rain forest boundary changes at decadal scales (e.g., fire impacts vs. climate effects), a key challenge is to develop mapping products with \pm meter-level accuracy.

Annual fire mapping derived from Landsat imagery sampled at least three times per year is available for Kakadu from 1980. For the 20-yr period covering this assessment, fire mapping has been undertaken derived from imagery mostly with 30 m pixels. Applying standard fire mapping procedures, Edwards and Russell-Smith (2009) reported that mean overall accuracy of Landsat-based fire mapping for complex terrain in western Arnhem Land was 87%, based on assessment against independent ground truth data, 1999–2004. While opportunities for routine fire mapping from finer-scale imagery sources are now, or will soon be available (e.g., Sentinel-2 satellites providing 10-m multispectral imagery at \sim 5-d return periods), we consider the applied imagery scale to be

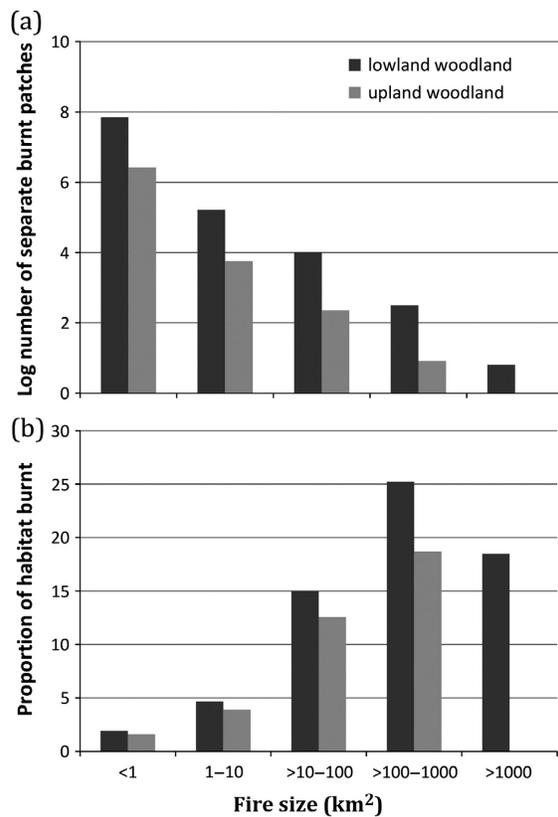


Fig. 5. Five-year (2011–2015) mean burnt patch-size distribution for (a) number of burnt patches, and (b) contribution to total mean area burnt, in lowland woodland and upland woodland habitats.

generally adequate for all our performance thresholds, including mapping of generally small fire patch sizes. In a comparable regional assessment, but using fire mapping derived from MODIS imagery with 250 m pixels, it was observed that this imagery scale was generally inappropriate for mapping small fires and describing resultant patchiness (Russell-Smith et al. 2015).

We have applied 10% as a benchmark threshold figure not to be exceeded in respective habitats in four of five performance metrics, in any one 5-yr period. While this benchmark is admittedly arbitrary—after all, it could reasonably be argued that any negative impact is unacceptable—we accept that management cannot account for all unforeseen circumstances. Tolerating higher levels of negative fire impact accumulative over time can readily be shown to degrade the natural resource itself. The fifth performance

metric, relating to maintaining minimum proportions of unburnt habitat, is considered below.

As described in *Materials and methods*, performance thresholds concerning the effects of severe fires on monsoon rain forest margins and fire-vulnerable woodland plant taxa, and too frequent fires on fire interval-sensitive sandstone heath taxa, are based on robust models derived from long-term observations established in three regional conservation reserves, including Kakadu (Russell-Smith et al. 2010, 2012, 2014). The effects of too frequent fires, at intervals less than three to five years, on long-lived obligate seeder shrubs in regional sandstone heaths are well documented (Russell-Smith et al. 1998, 2002, Russell-Smith 2006). Regional sandstone heath communities, including those occurring in Kakadu, are listed nationally as an Endangered Community given contemporary fire regime impacts (CoA 2012).

For assessments involving fire severity, we determined seasonality relationships derived from long-term monitoring plot observations for the decadal period 1995–2004 (Russell-Smith and Edwards 2006), supported by a subsequent assessment for the period 2005–2013 (Edwards et al. 2015a). Despite this, application of a remotely sensed trial method for directly assessing fire severity in savanna conditions (Edwards et al. 2013, 2015a) illustrates that in recent years at least, substantial areas of Kakadu appear to have been burnt relatively severely in the EDS period (A. C. Edwards, *unpublished manuscript*). While application of a remotely sensed method for assessing fire severity is likely to become operational in the near future, for present purposes we can conclude that our assessments of performance thresholds based on fire severity (Table 2, Figs. 3a–d, 4a–d) are likely conservative.

Proposed performance thresholds relating to maintaining significant areas of relatively long unburnt habitat, and for maintaining the sizes of burnt patches <1 km², pose significant assessment challenges—not the least because, to date, critical empirical observations are either lacking or ambiguous and, in the instance of appropriate burnt patch sizes, technically complex to assess. Addressing and refining such uncertainties, however, is integral to the adaptive management cycle (Biggs and Rogers 2003, van Wilgen et al. 2014).

While appropriate proportions of required unburnt woodland habitat, and their spatial

configuration, still requires assessment, it is recognized that a number of threatened small-mammal species occurring in Kakadu, including the Northern Brush-tailed Phascogale (*Phascogale pirata*), Fawn Antechinus (*Antechinus bellus*), Brush-tailed Rabbit-rat (*Conilurus penicillatus*), Black-footed Tree-rat (*Mesembriomys gouldii*), and Pale Field-rat (*Rattus tunneyi*), are favoured by longer-unburnt habitat conditions (Friend and Taylor 1985, Friend 1987). Whether the critical issue is maintaining unburnt habitat per se, or the maintenance of structurally diverse habitat conditions (e.g., species-rich, multilayered ground cover and shrubby sub-canopies) through landscape-scale imposition of relatively benign, small patchy fires (Kerle 1998, Woinarski et al. 2005, Yates et al. 2008), remains a key research issue.

For present purposes, however, we accept the guidelines proposed by Woinarski and Legge (2013) and Woinarski and Winderlich (2014), concerning requirements for maintaining significant proportions of unburnt woodland for periods from at least three to ten years (Table 1). Based on analysis of 32 yr of fire records for Kakadu's sandstone uplands, Murphy et al. (2015) show that such stipulated requirements appear achievable; by the end of their study period, ~30% of savanna woodlands were unburnt at least for 5 yr.

With respect to fire size, a first issue concerns determining an appropriate threshold patch size. As noted in *Materials and methods*, the home range and habitat requirements of many fauna taxa are vulnerable to fires which are substantially smaller than the performance threshold (1 km²) adopted here. This applies also to many plant taxa, especially narrowly dispersed obligate seeders (Russell-Smith 2006, Yates et al. 2008). Radford (2012) observed that low-intensity fires <1 km wide had little impact on small mammals in northwest Australia, whereas, in Kakadu, Lawes et al. (2015) found that fires <10 km² had limited effect on small-mammal abundance, but those >10 km² resulted in catastrophic declines. Felderhof (2007) also considered fire patch sizes <1 km² as appropriate for maintaining suitable habitat for a number of sedentary, territorial, ground-dwelling bird species in western Queensland savanna. As applied here, however, the adopted fire patch-size threshold is essentially a tradeoff between such varied ecological requirements, and significant current

challenges associated with achieving even this threshold in practice (Table 2, Fig. 5).

Mapping of fire patch sizes poses conceptual and technical challenges. Firstly, our mapping does not distinguish between individual fires (given that it is based on sampling Landsat imagery three times per year); hence, any one mapped patch may combine a number of separate fires occurring at different times. In the near future, this issue could be better addressed with fire mapping undertaken at more frequent intervals—for example, with higher resolution imagery sources such as the aforementioned Sentinel-2 system. However, such increased fire mapping frequency (and associated requisite validation) does come with increased resourcing costs. Additionally, it is appropriate to consider, as implied by Radford's (2012) study, alternative metrics which describe maximum width of individual fires, or distance to nearest unburnt patch, to cater at least for small mammals with restricted home ranges. Finally, there is a significant difference in effect on habitat conditions between a highly patchy, low-severity, typically early season fire and one occurring more typically in the late dry season that consumes all ground, shrub, and sub-canopy cover over an extensive area. Clearly, further research on appropriate, cost-effective fire patch-size metrics is required.

Management implications

Over almost four decades since the inception of Kakadu National Park in 1979, substantial strides have been made in fire management, particularly the early change from a formerly LDS wildfire-dominated fire regime to one where more strategic prescribed burning has been undertaken from earlier in the dry season (Russell-Smith et al. 1997a). The prescribed fire management program has also recorded substantial achievements over various periods with respect to increasing fire regime heterogeneity (Price et al. 2005), targeted habitat management (Murphy et al. 2015), and more generically in specific fire seasons including 2016. Effectively implemented, a prescribed regime potentially can deliver relatively small, patchy, low- to moderate-severity fires (Price et al. 2003, Williams et al. 2003, Russell-Smith and Edwards 2006), and hence, if strategically implemented along roadsides and natural barriers such as

streams, provide for effective buffers against more extensive severe fires later in the dry season (Price et al. 2007, 2012). Undertaken at landscape scales, such a program ideally can allow for implementing a variety of fire regime options catering to the ecological requirements of a range of different fauna and flora (Andersen et al. 2005, 2012, Yates et al. 2008, Woinarski and Legge 2013).

Notwithstanding these successes, and despite qualifications expressed previously concerning the application of some performance threshold metrics, data presented here illustrate ongoing significant challenges facing delivery of effective fire management for biodiversity conservation outcomes in Kakadu. This applies especially to Kakadu's extensive lowland woodland habitat which, as others also have observed, continues to be burnt by extensive fires at ecologically unsustainable frequency and likely severity (Andersen et al. 2005, Yates et al. 2008, Woinarski et al. 2010, Lawes et al. 2015). Trend data (Fig. 3) indicate that fire performance thresholds affecting this habitat type did not improve in lowland situations over the decadal assessment period.

Effects of severe fires on lowland monsoon rain forest boundaries were also found to be substantial, although, as discussed, significant mapping errors are likely. Based on an aerial photographic assessment of canopy cover change for 50 lowland monsoon rain forest patches in Kakadu, Banfai and Bowman (2006) observed that study patches expanded by a mean 29% over the period 1960–2004. Expansion was found to be strongly mediated by fire effects.

Of the lowland habitat types considered here, floodplain vegetation exhibited the best management outcomes, exceeding threshold targets for 3 and 10 yr unburnt vegetation (Table 2). In part, this is likely attributable to significant floodplain areas remaining inundated in all but the driest years (Fig. 4f, i), with the remainder being subject to extensive burning (Fig. 4l). Significantly, Kakadu's floodplain vegetation has been subject to, and substantially fashioned by, at least three thousand years of intensive Aboriginal fire management (Hope et al. 1985, Clark et al. 1992), which continues to the present (McGregor et al. 2010). Current understanding of traditional regional Aboriginal fire management practice, including on floodplains, is that extensive areas

were burnt progressively over the seasonal cycle generally in small individual fires (Jones 1980, Russell-Smith et al. 1997b, McGregor et al. 2010).

Recent assessments (Petty et al. 2007a, Russell-Smith et al. 2012, 2015, Murphy et al. 2015), and that presented here, indicate that delivering fire management outcomes for biodiverse upland communities also remains challenging. Major recurrent wildfires in the early–mid-2000s prompted a concerted adaptive management response including the establishment of an initial set of ecological thresholds criteria (Petty et al. 2007a). Murphy et al. (2015) found that by the end of 2011, fire management outcomes as measured against those criteria were in the best state since the establishment of the park. In particular, Murphy et al. (2015) demonstrated that the proportion of longer-unburnt (≥ 5 yr) vegetation could only have been achieved by implementing a strategic fire management program focused on reinforcing natural barriers with prescribed EDS burning. Since that time, however, most thresholds criteria describing upland habitat conditions have deteriorated substantially (Fig. 3).

A particular challenge for park management is to significantly reduce the ecological impacts of very large and severe fires across all habitats (Fig. 5). Data presented in Fig. 5 are analogous to those reported by Yates et al. (2008) under an effectively unmanaged wildfire regime for the adjoining western Arnhem Land region, 1997–2005, where 98.2% of fires < 1 km² contributed just 1.4% of the fire-affected area.

The substantial technical and operational fire management challenges identified here are not unique to Kakadu. More than 50% of substantial areas of north Australia's 460,000-km² mesic savannas (> 1000 mm mean annual rainfall) currently are burnt annually, mostly in extensive late dry season fires (Felderhof and Gillieson 2006, Russell-Smith et al. 2007, Edwards et al. 2015b). Although Kakadu is Australia's best resourced (both in staffing, funding) savanna conservation reserve, the resources required to consistently deliver effective, seasonally intensive, fine-grained adaptive fire management for biodiversity conservation outcomes are evidently well beyond the means of the publicly funded conservation estate.

Opportunities for engaging in market-based savanna fire management projects which deliver abatement of accountable greenhouse gas

emissions and, in the near future, carbon sequestration in biomass pools afford a tangible resourcing solution with realizable biodiversity conservation and broader social benefits. Such a program has been undertaken to significant effect over 28,000 km² of Aboriginal-owned lands in the contiguous western Arnhem Land region since 2006 (Russell-Smith et al. 2013, 2015). Comparison of ecological fire management performance between the Aboriginal-run West Arnhem Land Fire Abatement (WALFA) program with three major regional reserves (including Kakadu) over the ten-year period 2005–2014 illustrated that WALFA was outperforming the national parks when measured against three performance criteria (seasonal fire frequency, proportion of long-unburnt vegetation, fire patch-size distribution; Russell-Smith et al. 2015). At the time of writing, Kakadu's traditional Aboriginal landowners and the park management agency are in the process of establishing two such landscape-scale fire management projects in Kakadu. We acknowledge, however, that the prime responsibility for conservation management in this World Heritage property remains with the Australian Government and that appropriate resourcing is required to meet both international obligations and lease arrangements with traditional owners.

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