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Published in:
International Journal of Wildland Fire

DOI:
10.1071/WF18126

Published: 01/01/2020

Document Version
Peer reviewed version

Citation for published version (APA):
Delivering effective savanna fire management for defined biodiversity conservation outcomes: an Arnhem Land case study

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Abstract

Given the recent history of frequent and extensive late dry season wildfire in Australia’s fire-prone northern savannas, regional contemporary conservation-based fire management programs typically aim to prescribe increased levels of fire patchiness in order to promote more diverse spatio-temporal fire regimes and habitat mosaics. However, it remains unclear as to the extent such environmental management concerns are being addressed by these renewed fire management efforts. While there is growing appreciation of the ecological significance of fire size distributions dominated by very extensive fires, and degree of internal patchiness (i.e. proportion remaining unburnt within the fire perimeter) on the maintenance of biodiversity in Australian tropical savannas, robust definitions of ecological fire regime targets and thresholds still require refinement. This study documents changes in fire regime in the western Arnhem Land region of northern Australia associated with the implementation of active fire management since 2006. Over a twelve year period, the regional fire regime has transitioned from late dry season, wildfire-dominated to being characterised by a majority of fires occurring as small early dry season prescribed burns. While overall area burnt has not significantly decreased, most ecological threshold metrics have improved, with the exception of those describing the maintenance of longer unburnt
habitat. Challenges involved with defining, delivering, monitoring and evaluating heterogeneity targets are discussed.
Introduction

Fire is intrinsic to the savanna landscape and is a major driver in the structure, composition and dynamics of savannas (Scholes and Archer 1997; Russell-Smith et al. 2003; Woinarski et al. 2004; Higgins et al. 2007; Scott et al. 2012; Murphy et al. 2013). In northern Australia Indigenous people have managed regional landscape resources utilising fire for millennia (Jones 1969; Preece 2013). The co-existence of both fire-resistant (e.g. Burrows et al. 2010) and -sensitive taxa and habitats in northern Australian savannas suggests that while fire has been a key disturbance factor throughout the late Tertiary (Hill 1994), pre-historic anthropogenic fire regimes have allowed the persistence of many relatively fire-sensitive components, e.g.: *Callitris intratropica* [northern cypress pine] forests and groves (Bowman and Panton 1993; Russell-Smith 2006; Trauernicht et al. 2015; 2016); closed forests (Russell-Smith et al. 1993); sandstone heaths (Russell-Smith et al. 2002); small mammals (Woinarski et al. 2010; Ziembicki et al. 2015; Lawes et al. 2015); granivorous birds (Woinarski and Legge, 2013; Legge et al. 2015); riparian species and communities (Douglas et al. 2015).

As illustration of the ecological importance of patch dynamics theory (Watt 1947; Wu and Loucks 1995) and the application of fine-scale fire mosaics (pyodiversity) for delivering and enhancing biodiversity (Brockett et al. 2001; Bradstock et al. 2005; Parr and Andersen 2006), accumulating evidence supports the contention that regional pre-colonial anthropogenic fire regimes, characterised by fine-scale spatial patchiness as a result of organised management (including the preparation of strategic firebreaks) as people traversed their traditional estates over the seasonal cycle (Russell-Smith et al. 1997; Garde et al. 2009), contributed to the persistence of regional fire-sensitive biota (Yibarbuk et al. 2001; Trauernicht et al. 2016). In contemporary regional savanna contexts the application of strategic fine (hectare) -scale
burning practices over the course of the dry season is considered to provide a practical means for reducing the incidence of severe and extensive fires, thus contributing to the persistence of fire-sensitive biota (Fraser et al. 2003; Woinarski et al. 2005, 2015; Yates et al. 2008; Radford 2012). Contemporary management practice aims to mitigate extensive late season fires through (1) ‘breaking up’ areas consisting mostly of homogenous grassy groundcover fuel loads with patchy burning (Dyer et al. 2001), (2) reinforcing permanent and natural firebreaks during benign fire weather conditions of the early dry season, and, increasingly, (3) wildfire suppression (Price et al. 2007; Price 2015). While these strategies have generally been successful in reducing the extent of late season wildfire (Price et al. 2012), contemporary fire regimes (as well as a variety of other factors—cattle grazing (Kutt and Woinarski 2007), cat predation (Leahy et al. 2016; McGregor et al. 2017) and cane toad toxins (Shine 2010)) remain implicated in the ongoing collapse of especially regional faunal biodiversity values (Woinarski et al. 2015; Lawes et al. 2015).

For various savanna biota and habitats, a number of regional studies have proposed fire regime (frequency, severity, extent) thresholds required for the ongoing maintenance of biodiversity values and ecological function (Woinarski et al. 2005; Petty et al. 2007; Yates et al. 2008; Edwards and Russell-Smith 2009; Russell-Smith et al. 2012a, 2017; Woinarski and Legge 2013; Woinarski and Winderlich 2014). In the absence of detailed field-based assessments, trends in the performance of fire regime thresholds and other ecological drivers can inform pragmatic evaluation of the status and trajectory of ecological assets as part of the adaptive management cycle (van Wilgen et al. 2014; Commonwealth of Australia 2016). Although many fire managers specify the implementation of mosaic or patchy burning in their fire management plans, the spatio-temporal characteristics of appropriate fire patch size characteristics that best address diverse biodiversity conservation requirements typically
remain undefined. As a result (1) specific fire patchiness targets remain unspecified, and thus
(2) little attention is given to assessing the levels of resourcing actually required for
delivering mandated fire regimes.

In this assessment we document fire regime changes and, with respect to defined ecological
thresholds criteria, the purported biodiversity conservation effectiveness of the undertaking of
a major landscape-scale fire management project implemented primarily for reducing
greenhouse gas (GHG) emissions from savanna wildfires. Given the ecological importance of
fire patchiness and associated spatio-temporal habitat heterogeneity (i.e. the ‘hidden mosaic’;
Bradstock et al. 2005), we also assess the extent to which this savanna burning project has
been able to deliver enhanced fine-scale fire patchiness over the first 12 years of fire
management. Amongst the multitude of available metrics describing landscape patterning
(Turner 1989; McGarigal et al. 2009), here we describe trends in “burnt area patchiness” with
respect to two spatial heterogeneity metrics. In discussion we consider ongoing challenges
with regards to defining, delivering, monitoring, and evaluating appropriate levels of fire
regime heterogeneity.

Regional setting
The West Arnhem Fire Abatement (WALFA) project area encompasses 28,000km² of
remote, sparsely populated tropical savanna adjacent to the World Heritage-listed Kakadu
National Park, northern Australia (Fig. 1). The north-west of the region comprises a large
portion of the Arnhem Plateau, a deeply fissured, rugged, flat-bedded Middle Proterozoic
sandstone formation supporting a rich and endemic biodiversity (Woinarski et al. 2006,
2009), including a matrix of fire-sensitive components (e.g. Allosyncarpia ternata closed
monsoon forests, Callitris intratropica groves, and the Endangered Arnhem Plateau
Sandstone Shrubland (heath) Complex (Russell-Smith et al. 1993; Russell-Smith 2006; Commonwealth of Australia 2012), interspersed with highly flammable perennial *Triodia* spp. (spinifex) hummock grasses. The south-west of the WALFA area comprises undulating *Eucalyptus* (including *Corymbia*)-dominated woodlands and open woodlands, over mixed tussock grasses (Edwards and Russell-Smith 2009). Rainfall is highly seasonal with 90% of 1000mm – 1500mm occurring in the summer months. High wet season rainfall, combined with a typically seven-month dry season, supports rapid accumulation and subsequent annual curing of both grass and litter fuels (Williams et al. 2002).

The region was largely depopulated from the early decades of the twentieth century, with residents moving to larger settlements around the periphery (Cooke 2009). As a result, systematic traditional Indigenous landscape-scale fire management mostly ceased and, prior to the development of the WALFA program, the regional fire regime was characterised by extensive late dry season (LDS; typically from August until the start of the rainy period beginning in October-November) wildfires recurring on a three to four-year ‘boom and bust’ cycle (Bowman et al. 2001; Russell-Smith et al. 2009).

Previous studies have described the regional fire regime and assessed associated ecological threshold trends for commensurate ‘pre-management’ time periods for subsets of the Arnhem Plateau, including in adjacent Kakadu National Park (Edwards and Russell-Smith 2009; Russell-Smith et al. 2012). These studies found that the contemporary unmanaged fire regime for the region comprised mainly large (>10km²) LDS fires occurring at high frequencies (mean = 3.7 years), and that, for a commensurate time period, an average of 35.1% of the Arnhem Plateau was burnt with two thirds of these fires occurring during the LDS. Both
studies found that for this period ecological thresholds for fire-sensitive components were substantially exceeded.

In 2006, the WALFA project was formalised as an agreement between regional Indigenous land management organisations and a multinational energy corporate, with the aim of offsetting GHG emissions from a major liquefied natural gas project (Whitehead et al. 2009). The agreement requires the contractual abatement of 100,000 t.CO₂-e yr⁻¹ over a 17-year period through imposition of strategic early dry season (EDS—typically from April to July, when temperatures are coolest and overnight dews are common) fire management as a means for reducing the extent, severity, and substantial GHG emissions associated with, LDS fires. The WALFA project has fostered the recommencement of a concerted fire management program where, over the first decade of operations, there has been general improvement in the regional fire regime, reflected in a change to burning occurring primarily during the EDS, an increase in longer-unburnt vegetation, and a decline in extensive LDS fires (Russell-Smith et al. 2015).

The GHG emissions accounting process for WALFA (Russell-Smith et al. 2009) was formalised in 2012 as a national methodology for generating carbon credits under the Australian Government’s Carbon Farming Initiative (CFI). In 2014 WALFA was registered under the CFI and continues to benefit from increased resourcing made possible through both the industry partnership, participation in the carbon market, and Australian Government funding for Indigenous rangers and related programs (Russell-Smith et al. 2015). While WALFA consists of five cooperating Indigenous land management organisations, these partners nonetheless maintain similar aims with regards to achieving cultural and biodiversity conservation goals (Ansell et al. 2019, this edition).
Methods

Characterisation of the WALFA fire regime was undertaken by analysis of a 28-year spatial fire history archive derived mostly from Landsat imagery. Defined ecological thresholds and fire regime metrics derived from other regional ecological studies were assessed with reference to available vegetation habitat mapping. Temporal trends in the status of thresholds were assessed with respect to periods representing pre- and post-recommencement of fire management. Respective layers were sourced or constructed as follows.

Habitat Mapping

WALFA comprises two broad landform units; rocky uplands (UPL) and undulating lowlands. Uplands comprise the sandstone Arnhem plateau and its matrix of woodlands (10-30% canopy cover) and open woodlands (<10% canopy cover; UWL), shrubby sandstone heath (SSH), and interspersed closed canopy Allosyncarpia ternata-dominated monsoon forests, herein referred to as upland closed forests (UCF). Lowlands comprise mostly Eucalyptus-dominated woodlands and open woodlands (LWL), smaller extents of open forests (30 – 70% canopy cover), making up the remainder (40%) of the project area (Fig. 2).

Mapping of all landform and habitat layers, with the exception of UCF distribution, is derived from Edwards and Russell-Smith 2009, with an additional area of ~4000 km² included in 2012 to meet expanded WALFA area requirements, using the same object-based classification method. UCF mapping (Freeman et al. 2017) was derived predominantly through manual digitisation of ~1 x 1m resolution imagery, delineating all forest patches greater than ~1 ha. All analyses except those addressing UCF boundaries were undertaken
with layers resampled to 1 ha cell size. Thresholds and metrics for UCF were assessed with reference to an inner buffered 50m forest perimeter. As a pragmatic compromise between preserving the fine-scale mapping integrity of this habitat type relative to the fire history data, and computational efficiency, analyses involving UCF were undertaken using layers at a cell size of 10 x 10m.

**Fire History**

The dataset used was a refined and updated version of an existing sixteen year (1990-2005), primarily Landsat-derived fire history (Edwards and Russell-Smith 2009). Excessive cost of satellite data at that time necessitated the archive being derived partly from digitised hard copy media or spatially degraded thumbnail (quicklook) files. Since that study however, all historic Landsat data archives have become freely available. For those years where sub-optimal data sources were utilised, re-mapping was undertaken using standard full resolution (30 x 30m) imagery. A minimum of three sample dates encompassing the dry season, one being the latest suitable cloud free image for the year, were utilised. Similar classification methods were applied, and while no additional validation was performed, the original mapping was found to have a mean overall accuracy of 87% (Kappa co-efficient = 0.72) (Edwards and Russell-Smith 2009). Some mapped burned areas (particularly for LDS periods where the onset of cloudy weather conditions restricts the use of relatively low temporal resolution Landsat data), were augmented with spatially coarse but more frequently available (daily) Advanced Very High Resolution Radiometer (AVHRR, 1.1 x 1.1km pixel), and Moderate Resolution Imaging Spectroradiometer (MODIS, 250 X 250m pixel) data. This use of coarser spatial resolution data sources can result in the omission of smaller (<10km²) fires (Yates et al. 2008). However, during the LDS, fires tend to be substantially more extensive (Yates et al. 2008) and severe (Williams et al. 1998; Russell-Smith and Edwards 2006),
resulting in spatially more complete (i.e. less patchy) burns (Russell-Smith et al. 1997; Price et al. 2003). Relatively coarse resolution mapping of extensive LDS fires can have >90% agreement with finer resolution sources (Edwards et al. 2018).

Statistical assessment (two-tailed t-tests, P<0.05) of change in mean extent of fires, and in fire size-age-classes, was undertaken for four consecutive assessment periods (two pre-project periods—1990 – 1997, 1998 – 2005; two with-project periods—2006 – 2011, 2012 – 2017).

In the absence of available fire severity mapping products, for the purpose of assessing impacts of severe fires we categorise all LDS fires as being severe (leaf scorch >2 m) based on strong relationships observed between fire seasonality and severity described in various regional studies (Williams et al. 2003; Russell-Smith and Edwards 2006; Edwards et al. 2015). However, as addressed in Discussion, relatively small proportions of EDS fires are observed to be of high severity (total leaf scorch), and <20% of LDS fire extent is recorded as being of low severity.

We describe fire mosaic patchiness with respect to two indices, aggregation (Clumpiness Index), and disaggregation or dispersion (Landscape Shape Index), using the FRAGSTATS v4 package (McGarigal et al. 2012). Clumpiness Index describes fragmentation of a focal class as a range from -1 to 1 being maximally disaggregated to maximally clumped, with a random distribution returning zero. Landscape Shape Index is a standardised measure of class edge density that increases from ≥ 1 as total edge within the landscape increases and becomes more irregular. As for these purposes FRAGSTATS defines a contiguous patch as adjoining
orthogonal (not diagonal) cells of the same class, we have also followed this rule for summarising patch–size-class distributions.

Ecological thresholds

Ecological thresholds for above habitats and associated biodiversity values were drawn from the literature. Three performance threshold metrics (effects of severe fires on UCF and woodland habitats including Callitris stands; effects of frequent fires on SSH—Table 1), are based on statistically significant simple linear response models derived from long-term monitoring plots (Russell-Smith et al. 2010, 2012, 2014). Respective models are based on 15 years of observations (1995-2009) from 48 plots located in Kakadu’s sandstone uplands (Russell-Smith et al. 2012). We assume that, for long-term sustainability, thresholds criteria for these response groups (and for fire extent criteria below) conservatively should not be exceeded by more than 10% over any 5-year period.

Two further key performance thresholds address requirements for 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 ranges and dispersal limitations of many small vertebrates and plant taxa, respectively (Table 1). In an earlier assessment of the extent of burning in Kakadu National Park, Andersen et al. (2005) observed that there was a ‘serious lack of long unburnt habitat catering for relatively fire-sensitive species’—with ~10% of lowland savannas remaining at least 3-years unburnt, and <3% remaining unburnt after 10 years. For savanna birds Woinarski and Legge (2013) have proposed that ‘(1) at least 25% of the savanna landscape…is at least 3 years 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 years 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).

With respect to fire patch size, for assessment purposes we have adopted a threshold size of 1 km$^2$ as a pragmatic compromise based on a large and variable literature. As noted in Table 1 and Discussion, however, many regional studies illustrate that required spatial scales of burning (patch sizes) for fire-vulnerable fauna and flora taxa may be at hectare-scales, or even less, for example: <<1 ha for dry season nymph colonies of the spectacular Leichhardt’s grasshopper, *Petasida ephippigera* (Lowe 1995; Barrow 2009); <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).

**Results**

*Fire regime*

The fire regime for the WALFA region has undergone considerable change since concerted fire management recommenced in 2006 (Fig. 3). For the two consecutive unmanaged assessment periods, 1990-1997, and 1998-2005, annual means of 40% and 43% of the WALFA region were burnt, translating to mean fire return periods (FRP) of 2.5 and 2.3 years respectively (Fig. 4). Over both of these periods approximately three times the area was burnt in the LDS (FRP = 3.1 and 3.2 years, respectively), than during the EDS (FRP = 13.8 and 8.8 years, respectively). For the two management assessment periods, 2006-2011, and 2012-2017, 35% and 37% of WALFA was burnt on average (FRP = 2.9 and 2.7 years,
respectively: Fig. 4). Over these managed periods between 2 and 3 times the area was burnt during the EDS (FRP = 4.5 and 3.3 years), than in the LDS (FRP = 7.9 and 15.6 years).

Variation in area burnt was greatest in the unmanaged era; for example, ranging from 16% in 1992 (with 9% burnt in the LDS) to 70% in 2004 (with 61% burnt in the LDS). Conversely, in the managed era the areal extent has ranged between 27% in 2007 (with 7% burnt in the LDS) to 52% in 2012 (with 11% in the LDS). These general trends are reflected in both uplands and lowlands, although lowlands have been burnt approximately twice as much on average as uplands.

As the overall area of burning occurring during the EDS has increased (Fig. 4), this burning has also comprised a substantial increase (400%) in relatively smaller EDS fires; conversely LDS fires have decreased both in size and number of contiguous burnt patches (Fig. 5, 6).

Size classes of fires occurring across the WALFA project area in unmanaged years were defined as comprising ~0.5% (<1km²), ~1% (>1 to 10 km²), and ~10% (>10km²) during the EDS, and ~0.5%, ~0.8%, and ~30% during the LDS. Since the commencement of WALFA, EDS size classes have comprised ~2%, ~4%, and ~20% in respective size classes, while LDS size classes have comprised just ~0.5%, ~1%, and ~8% in respective size classes.

Over the assessment period, fire patches have become smaller with management (Fig. 5), more dispersed (Fig. 3), less clumped (Fig. 6c), and more disaggregated (Fig. 6d). However, despite the mean frequency of burning having declined overall (albeit non-significantly) since the commencement of the WALFA program (Fig. 4), the distribution of time-since-fire has not changed markedly over the entire assessment period with the notable exception of vegetation unburnt for >5 years in the LDS (Fig. 7).
**Ecological thresholds**

Thresholds criteria were mostly substantially exceeded during the pre-WALFA years and, after 12 years with-management, the WALFA project met four of ten assessed thresholds criteria (Fig. 8). While criteria for maintaining at least three years unburnt habitat in woodland settings was met for some years this was interspersed by years where the threshold was exceeded. The most marked improvement in ecological thresholds metrics has concerned the incidence of severe (LDS) fires; by 2017, <10% of both UWL and LWL were affected by more than one LDS fire over the preceding five years (Fig. 9c, d). This criterion was also met for seven years from 2011 for UWL, but only marginally for LWL.

The other two criteria that were met by 2017 both address maintaining relatively longer unburnt habitat. At least 25% of LWL was unburnt for three years for 2016 and 2017, and has been maintained at an average of 25.9% since 2006. UWL has been maintained at an average of 53% unburnt for 3 years or more since 2006, addressing this threshold in all managed years. All criteria relating to upland habitats displayed a spike away from desired thresholds during the late 1990’s to early 2000’s. This corresponds to the high occurrence of extensive LDS fires during the period 1998-2005, notably in the uplands (Figs. 4,5).

The spatial distributions of calculated metrics expressed as a proportion of affected 1 x 1 km cells (see supplementary material), generally show that where thresholds are either being met, or there appears to be a trend toward attaining desired thresholds, this has occurred more or less evenly across the range of the habitat. Notable exceptions include metrics representing the frequency of extensive but concomitantly localised LDS fires, as illustrated in Figs. 9a,b for the years 2005 and 2017, respectively.
**Discussion**

Data presented demonstrate that, after a decade of organised fire management, substantial advances have been made in addressing the previous wildfire regime and at the same time delivering putative ecological benefits. These results reiterate previous findings that, prior to the commencement of WALFA, the region was experiencing deleterious fire regimes dominated by LDS wildfires with negative impacts on component fire-sensitive habitats and taxa (Bowman and Panton 1993; Bowman *et al.* 2001; Edwards and Russell-Smith 2009; Woinarski *et al.* 2009).

Over the 12-year WALFA management period the seasonality of burning, and associated reduction in LDS wildfires, has substantially improved. Additionally, it is evident that the annual patterning of burning is becoming inherently patchier, illustrated especially by the rapidly increasing trend in numbers of individual burnt patches (Fig. 6a). However, the contributions of larger-sized fires (>1 km²), although having diminished markedly in the LDS in the management period, continue to overwhelmingly dominate the mean areal extent of the WALFA region that is burnt (Fig. 5).

*Implications – Monitoring and thresholds*

The fire mapping surfaces presented here reflect simple EDS and LDS binary classifications (burnt, unburnt pixels) derived from a minimum of three sample dates throughout the dry season. Given the restricted temporal sampling, a first implication is that presented mapped fire patch sizes do not represent the total number of individual fire events, but rather contiguous burnt areas occurring between sampling dates. This could include separate but
abutting fire events. While available satellite data sources are improving with increased
spatial resolution and revisit times (e.g. Sentinel-2 satellites providing 10-m multispectral
imagery at ~5-d return periods), these spatially and temporally refined imagery sources have
only become available in recent years (e.g. Sentinel-2 since 2015). Such scale issues have
significant implications for resultant patch-size assessments. For example, a recent study
assessing the patch-size distributions for the WALFA area but based on daily but coarser
resolution MODIS imagery (250 X 250m pixels), observed a mean annual number of 194 fire
patches for the pre-management period 2000 – 2004, and an annual mean of 464 fire patches
for the with-management period 2010 – 2014 (Russell-Smith et al. 2015). By contrast, using
Landsat imagery, in the three most recent years of our assessment the numbers of observed
burnt patches exceeded 10,000 annually (Fig. 6a).

A second qualification with presented fire mapping data is the assumption that all regional
landscape fires in the EDS are of low, and LDS fires are of relatively high, intensity /
severity. As a generality this assumption is supported by a number of studies (Williams et al.
2003; Russell-Smith and Edwards 2006; Edwards et al. 2015). Based on 19 years of
observations of fire severity effects at 79 permanent sandstone vegetation plots in Kakadu
and Nitmiluk National Park, Edwards et al. (2015) recorded that 75% of EDS fires, and 17%
of LDS fires, were of low severity (ie. scorch height <2 m). EDS fires can be of high intensity
/ severity (e.g. if ignited under hot windy conditions), and conversely, LDS fires can be of
low intensity / severity (e.g. under night-time, low wind and relatively high humidity
conditions). It has also been suggested that EDS fires, when extensive and frequent, could
negatively impact fauna with small home ranges (Fraser et al. 2003; Radford et al. 2015;
Lawes et al. 2015). However, it is the case that the majority of EDS fires are in fact of far
lower severity and carry substantially greater internal unburnt patchiness than LDS fires
(Price et al. 2003), with obvious beneficial ecological implications (Oliveira et al. 2015). An operational system for automated mapping of fire severity, which will provide for a more accurate spatio-temporal representation than the current arbitrary seasonal classification, is likely to become available for regional savannas in the near future (Edwards et al. 2015, 2018).

Elsewhere, we have recently discussed challenges associated both with adequately ecologically defining, and technically measuring, the performance thresholds assessed here (Russell-Smith et al. 2017). Salient issues include uncertainties associated with: (a) defining appropriate proportions and configurations of unburnt habitat required for conservation of a variety of relatively immobile small vertebrates (Friend 1985; Petty et al. 2007; Woinarski and Legge 2013; Woinarski and Winderlich 2014) —or perhaps rather, the maintenance of structurally diverse habitat conditions through landscape-scale imposition of relatively benign, small patchy fires (Kerle 1998; Fraser et al. 2003); and (b) defining appropriate threshold fire patch sizes for a range of fire-vulnerable taxa, and then being able to effectively map these.

Mounting evidence indicates that extensive, especially frequently recurring, fires with limited internal unburnt patchiness (even in the EDS) have significant deleterious impacts on fire-vulnerable, relatively immobile savanna fauna with restricted home ranges, and poorly dispersed obligate seeder plant taxa (Fraser et al. 2003; Woinarski et al. 2005; Yates et al. 2008; Oliveira et al. 2015; Radford et al. 2015; Lawes et al. 2015). Noting the above qualification that our fire mapping surfaces do not necessarily depict individual fire events, here we have applied a fire patch size performance threshold of 1 km$^2$ essentially as a trade-off between the varied ecological patch-size requirements of fire-vulnerable taxa, and the
evident significant challenges associated with achieving even this threshold in practice (Fig. 5). Many regional fauna and plant taxa are vulnerable to fire-patch sizes significantly smaller than 1 km² (Woinarski et al. 2005; Yates et al. 2008; Table 1), notably including vulnerable dry season colonies <<0.1 ha of immature nymphs of the spectacular Leichhardt’s grasshopper (*Petasida ephippigera*; (Lowe 1995; Barrow 2009)).

The spatial configuration of burnt area can be assessed in different ways, for example: Radford (2012), Radford et al. (2015) and Lawes et al. (2015) assessed mammal species richness and abundance with respect to distance to mapped fire perimeter. It was shown that fires (including those occurring during the EDS) <1 km wide can have negligible impact on small mammals (Radford 2012), whereas increasingly extensive fires can have a negative impact (Radford et al. 2015), and those >10 km² can have catastrophic impact (Lawes et al. 2015). Fraser (2000) and Fraser et al. (2003) defined habitat suitability for the mostly ground-dwelling granivorous Partridge Pigeon (*Geophaps smithii*), with respect to the proportion of a pair’s home range (typically ~8 ha) that was burnt (ideally 40-60%). Evidently, a variety of fire patch-size thresholds can be applied to cater for the requirements of different taxa (where known).

Given the implicit importance of developing fire-induced patch heterogeneity for biological conservation outcomes (Bradstock et al. 2005; Parr and Andersen 2006), we have presented annual assessments of two patch dispersion indices (Fig. 6c, d; McGarigal et al. 2009) to illustrate and quantify the trajectory of fire patchiness development. Price et al. (2005) developed a broadly similar conceptual approach to illustrate developing fire patchiness heterogeneity in Kakadu National Park over four consecutive 5-year periods. Three patch-based heterogeneity indices were calculated from assembled Landsat-scale fire history data
for the central 1 ha cell of a 5 X 5 cell array; effectively at a spatial scale relevant to the home
ranges of many small native mammals. The assessment demonstrated: that fire-induced
heterogeneity in Kakadu increased overall in each successive five-year periods from 1981;
the mappable trajectory of individual cells; and the amenity for undertaking statistically based
analyses of the geo-spatial correlates of change. Additionally, Legge et al. (2011) applied a
measure of the average distance between unburnt areas to describe patch dispersion. Given,
however, the diverse spatio-temporal patch-size (and configuration) requirements of regional
flora and fauna, defining appropriate heterogeneity metrics for informing biodiversity
conservation assessments presents evident challenges.

Implications - Management

The WALFA project is succeeding in addressing some ecological thresholds of concern. The
continued strategic implementation of relatively fine-scale prescribed burning has effectively
reduced the impact of severe fires on closed canopy monsoon forests and woodlands
generally, and enabled the maintenance of moderately long unburnt (≥3 yr) vegetation
especially in upland woodlands. Challenges remain with regard to restricting frequent fires in
sandstone heath, reducing fire size particularly in structurally homogenous lowland
woodlands, and maintaining longer unburnt sandstone heath and upland habitats generally.

Similar analyses undertaken for WALFA and three regional national parks (Russell-Smith et
al. 2015) revealed that while WALFA has made some progress in addressing the maintenance
of longer unburnt vegetation, these thresholds also remain problematic elsewhere.

Since 2006 fire management of the region has been resourced primarily through contractual
obligations to abate emissions. The mechanistic accounting method developed as part of the
founding WALFA agreement (that underpins the national savanna burning scheme) means
credits are earned for reducing the total area burned annually relative to the pre-project 10-
year baseline, weighted by (a) season (approximately twice the quantum of GHG emissions
per unit area are generated in the LDS relative to the EDS), (b) vegetative fuel type (differing
fuel loads per vegetation-fuel class), and (c) fuel age (older, greater fuel loads carry greater
emissions potential) (Russell-Smith et al. 2009). Along with socio-cultural aspirations
(Ansell et al. 2019 this edition), the financial incentive to reduce especially LDS burnt area
requires strategic prescribed burning that effectively can only be implemented during the
EDS under relatively benign fire weather conditions. Ultimately this results in an increasingly
patchy mosaic of annual burning patterns and commensurate ecological benefits.

The prevailing management approach is to prescribe fire breaks by prioritising, where
feasible, strategic EDS burning parallel to riverine and road corridors, and along open grassy
valleys dissecting rugged rocky plateaux. In the absence of natural fire breaks, extensive
areas of homogenous fuel loads are secured with iterative, progressive burning during
weather conditions that inhibit fire spread. Burning commonly ceases prior to the defined
EDS/LDS cut-off (31 July) if seasonal fire weather conditions deteriorate before this date and
prescribed burns are deemed to be becoming too extensive. The efficacy of this approach has
been demonstrated in sandstone uplands of Kakadu National Park where, for the period 2007
– 2011, targeted burning of natural fire-break features resulted in significant increase in
longer-unburnt habitat although the annual extent of prescribed burning remained more-or-
less constant (Murphy et al. 2015). This approach has also been demonstrated to be effective
in mitigating the spread of LDS fires when effort is made to implement continuous burnt
breaks (Price et al. 2007), especially when combined with fire suppression (Price 2015).
However, while the proportion of longer unburnt fuel is increasing in WALFA, maintaining
longer unburnt habitat for various fire-vulnerable taxa (e.g. White-throated Grasswrens
[Amytornis woodwardi], Leichhardt grasshopper [Petasida ephippigera] colonies, various long-lived obligate seeder heath taxa) remains a challenge under these arrangements.

Given considerable financial and human resourcing, and accumulated recent management experience, it is unlikely that the significant advances made by WALFA to date can continue to be enhanced over the entire project area. Realistically, prioritised investment in fire management for biodiversity conservation outcomes needs to target specific locations—requiring considered assessment of the costs and trade-offs associated with different or refined management options. Over recent decades substantial biodiversity survey data and enhanced mapping products have been assembled which can assist with identifying the distributions of critical Arnhem Plateau biological assets and prioritise appropriately-scaled fire management requirements. In this context we note that the extensive endemic sandstone heathlands of the Arnhem Plateau (including in Kakadu National Park) are formally listed as an Endangered Community (the Arnhem Plateau Sandstone Shrubland Complex) under national legislation, with contemporary fire regimes identified as the major threat (Commonwealth of Australia 2012). To date, that listing has provided no funding assistance to develop the legislatively required Recovery Plan, nor help address the regional fire management challenge.

Acknowledgements

The authors acknowledge the immense dedication and hard work shown by all of the people and organisations involved in making the Western Arnhem Land Fire Abatement project a success in contributing to the protection of one of Australia’s cultural and natural icons, the
Arnhem Plateau. Jeremy Freeman, Andrew Edwards and Cameron Yates contributed habitat and burnt area mapping products. This contribution is part of a broader research collaboration addressing enhancing savanna fire management in northern Australia, undertaken under the auspices of the Bushfire & Natural Hazards Cooperative Research Centre. Thanks also to anonymous reviewers for their constructive comments.
References


Table 1. Summary of applied ecological thresholds.

<table>
<thead>
<tr>
<th>Response group</th>
<th>Rationale</th>
<th>Threshold exceeded when:</th>
</tr>
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<tbody>
<tr>
<td>Severe fires affecting closed canopy monsoon forests (UCF)</td>
<td>Closed canopy monsoon forests dominated by the regional endemic, <em>Allosyncarpia ternata</em>, are susceptible to incursions particularly from severe fires (Russell-Smith et al. 2012).</td>
<td>&gt;10% of forest boundaries impacted by any severe (LDS) fires over five years. Refer Figs 8, 9 (a), 'UCF severe'.</td>
</tr>
<tr>
<td>Frequent fire affecting sandstone heath (SSH)</td>
<td>Regional heath assemblages comprise many obligate seeder shrub taxa with juvenile periods ranging from 3 to &gt;10 years, including various <em>Acacia</em> spp. and the serotinous shrub, <em>Petraeomyrtus punicea</em> (Russell-Smith et al. 1998, 2002; Russell-Smith 2006; Russell-Smith et al. 2012).</td>
<td>&gt;10% of sandstone heath habitats remain less than five years unburnt. Refer Figs 8, 9 (b), 'SSH frequent'.</td>
</tr>
<tr>
<td>Severe fire affecting woodlands (UWL, LWL)</td>
<td>Eucalypt-dominated regional woodland formations supporting stands of the long lived, fire vulnerable obligate seeder tree, <em>Callitris intratropica</em> (Bowman and Panton 1993, Russell-Smith 2006; Edwards and Russell-Smith 2009, Russell-Smith 2012).</td>
<td>&gt;10% of woodland habitats impacted by more than one severe (LDS) fire over five years. Refer Figs 8, 9 (c) and (d), 'UWL severe' and 'LWL severe'.</td>
</tr>
<tr>
<td>Maintaining structural diversity in woodlands (UWL, LWL)</td>
<td>Fire-mediated habitat heterogeneity, especially the development of diverse shrub and mid-canopy food resources in the absence of burning, is critical for many small mammal and bird taxa (Andersen et al. 2005, Fraser et al. 2003, Radford et al. 2015, Woinarski et al. 2005, Woinarski and Legge 2013, Woinarski and Winderlich 2014).</td>
<td>For UWL habitats: &lt;40% remains at least three years unburnt, and; &lt;10% remains at least ten years unburnt. Refer Figs 8, 9 (e) and (f), 'UWL 3yo' and 'UWL 10yo'. For LWL habitats: &lt;25% remains at least three years unburnt, and; &lt;5% remains at least ten years unburnt. Refer Figs 8, 9 (g) and (h)</td>
</tr>
<tr>
<td>Frequent large fires impacting fauna with restricted home ranges and obligate seeder flora (UPL, LWL)</td>
<td>Invertebrates, birds and small mammals with restricted home ranges, and obligate seeder plant taxa with limited dispersal capacity and non-dormant seed banks, have both been shown to be impacted by extensive fires—including fires ranging in size from &lt;1 - &lt;10 ha (Lowe 1995; Kerle 1998; Franklin 1999; Oakwood 2002; Fraser et al. 2003; Woinarski et al. 2005; Firth et al. 2006; Russell-Smith 2006; Yates et al. 2008; Barrow 2009; Radford 2012; Woinarski and Legge 2013; Hohenet et al. 2015; Lawes et al. 2015; Radford et al. 2015).</td>
<td>&gt;10% of major landscape units are impacted by any extensive (&gt;1 km²) fire over five years. Refer Figs 8, 9 (i) and (j)</td>
</tr>
</tbody>
</table>
Figure 1: Location of the West Arnhem Land Fire Abatement project.
Figure 2: Distribution of landform units (a) uplands, lowlands, (b) closed canopy monsoon forests (UCF), (c) sandstone heaths (SSH), where (b) and (c) mapped as proportion of 1 x 1km cells.
Fig. 3: Fire mapping for the WALFA project area, illustrating (a) first five years (1990-1994) representative of pre-project period, and (b) last five years (2013-2017) representative of project period, where early dry season (EDS) fires are given in green, and late dry season (LDS) fires are given in red.
Figure 4: Average area burnt distributions, as proportion of WALFA region, in four assessment periods, where: pre-project periods = 1990 – 1997, 1998 – 2005; with-project periods = 2006 – 2011, 2012 – 2017; error bars = standard error of mean; * indicates significant (t-test: P<0.05) change from previous pre-project period.
Figure 5: Fire size-class (km$^2$) distributions, expressed as proportion of WALFA region, in four assessment periods, where: pre-project periods = 1990 - 1997, 1998 – 2005; with-project periods = 2006 – 2011, 2012 – 2017; error bars = standard error of mean; * indicates significant (t-test: $P<0.05$) change from previous pre-project period.
Figure 6: Seasonal (EDS, LDS) fire patch-size distributions, 1990 – 2017, (a) number of contiguous burnt patches (X 1,000), (b) proportion of WALFA project area burnt, and dispersion/disaggregation of burnt area indicated by (c) Clumpiness Index, and (d) Landscape Shape Index (refer text for details).
Figure 7: Annual and late dry season (LDS) mean time-since-burnt (years) distributions, over four six-year periods expressed as proportion of WALFA region. Note: pre-project periods = 1994 - 1999, 2000 – 2005; with-project periods = 2006 – 2011, 2012 – 2017; * indicates significant (t-test: $P<0.05$) change from previous pre-project period.
Figure 8: Twenty four-year trends for respective ecological performance threshold metrics, for (a) severe fires affecting closed canopy monsoon forests (UCF), (b) frequent fires affecting sandstone heath (SSH), (c) severe fires affecting upland woodlands (UWL), (d) severe fires affecting lowland woodlands (LWL), (e) UWL unburnt for at least 3 yr, (f) UWL unburnt for at least 10 yr, (g) LWL unburnt for at least 3 yr, (h) LWL unburnt for at least 10 yr, (i) upland habitats burnt by fires >1 km², (j) lowland habitats burnt by fires >1 km². Dashed line is threshold. Note that the calculated threshold metric is based on fire mapping data for the five preceding years except for (f), (h) based on preceding ten years (refer Methods for details).
Figure 9: Spatial distribution of the status of ecological thresholds represented as a proportion of habitat affected in 1 x 1 km cells for 2005 and 2017: (a) UWL unburnt for at least 3 yr, (b) severe fires affecting LWL. Colour ramp indicates proportion of habitat affected with green as zero, yellow being near-threshold levels, and red as all habitat within cell affected.
(b) LWL severe

2005

2017
Supplementary material

Spatial distribution of the status of ecological thresholds for (a) severe fires affecting closed canopy monsoon forests (UCF), (b) frequent fires affecting sandstone heath (SSH), (c) severe fires affecting upland woodlands (UWL), (d) severe fires affecting lowland woodlands (LWL), (e) UWL unburnt for at least 3 yr, (f) UWL unburnt for at least 10 yr, (g) LWL unburnt for at least 3 yr, (h) LWL unburnt for at least 10 yr, (i) upland habitats burnt by fires >1 km$^2$, (j) lowland habitats burnt by fires >1 km$^2$, represented as proportion of habitat affected in 1 x 1 km cells for 2005, representative of pre-project period, and 2017, representative of project period. Colour ramp indicates proportion of habitat affected with green as zero, red as all habitat within cell affected.