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POLICY PERSPECTIVE

Better biodiversity accounting is needed to prevent bioperversity and maximize co-benefits from savanna burning

Ben Corey | Alan N. Andersen | Sarah Legge | John C. Z. Woinarski | Ian J. Radford | Justin J. Perry

Abstract
Strategies for mitigating climate change through altered land management practices can provide win–win outcomes for the environment and the economy. Emissions trading for greenhouse gas (GHG) abatement in Australia’s remote, fire-prone, and sparsely populated tropical savannas provides a financial incentive for intensive fire management that aims to reduce fire frequency, severity, and extent, and it supports important social, economic, and land management opportunities for remote communities, conservation agencies, and pastoralists. These programs now cover >20% of Australia’s 1.9 million km² tropical savanna biome, encompassing areas of globally significant biodiversity value. A common assertion is that by reducing the frequency, severity, and extent of fires for GHG abatement, these programs provide biodiversity co-benefits. However, such biodiversity benefits have been assumed rather than demonstrated. Much better accounting of how biodiversity is responding to changed fire management is required to ensure that there are no unintended outcomes for biodiversity (bioperversity), and that biodiversity co-benefits are maximized. Such accounting could underpin the earning of formal biodiversity credits from improved fire management, and will go a long way to understanding and improving the biodiversity outcomes of savanna fire management.

KEYWORDS
biodiversity offset, carbon farming Initiative, conservation, Emissions Reduction Fund, fire, greenhouse gas emissions, payment for ecosystem services

1 INTRODUCTION
The development of a carbon economy is driving land management changes that potentially provide win–win outcomes for environments and economies (Bradshaw et al., 2013). Tree plantings and avoided deforestation are two well-known examples (e.g., Lindenmayer et al., 2012; Phelps, Friess, & Webb, 2012). Another potential example is fire management in tropical savannas, the most fire-prone landscapes on Earth (Lipsett-Moore, Wolff, & Game, 2018). Savanna fires emit substantial quantities of the greenhouse gases (GHG) methane and nitrous oxide, and “savanna burning,”
an accountable activity under the Kyoto Protocol, has been identified as a major GHG abatement opportunity globally (Lipsett-Moore et al., 2018).

Australia is the only country to recognize savanna burning in its national emissions accounts, and savanna fires contribute 4% of Australia’s annual GHG emissions (Cook & Meyer, 2009). The Australian Government’s Emissions Reduction Fund (ERF) provides a carbon-crediting mechanism for engaging industry to offset GHG emissions and provides the financial basis for savanna fire management projects. Such projects represent important livelihood and land management opportunities, especially for remote Aboriginal communities, which own and manage much of Australia’s tropical savanna (Cook, Jackson, & Williams, 2012; Russell-Smith et al., 2013), but also pastoralists (Skoeblin, Legge, Webb, & Hunt, 2014) and conservation agencies (Russell-Smith, Evans, Edwards, & Simms, 2017).

Landscape fire management has been critical to the management of natural resources, and access to them, by Aboriginal people in northern Australia for millennia, and fire management continue to play a fundamental role in the connection to and stewardship of land by Aboriginal communities (Cook et al., 2012). However, European colonization resulted in the dispossession and movement of Aboriginal people off their traditional lands, and fire became largely unmanaged across the vast tropical savannas of northern Australia. As a result, in the second half of last century, these landscapes became dominated by frequent, large, and high-intensity wildfires, mostly occurring in the late dry season (LDS; Yates, Edwards, & Russell-Smith, 2008; Figure 1). These fires threaten fire-sensitive habitats embedded in the savanna matrix, as well as a range of relatively fire-sensitive taxa (Table 1). A widespread aim of conservation management is therefore to re-establish strategic fire management in order to reduce the extent of higher intensity fires occurring late in the dry season (Andersen et al., 2005).

Prescribed fire management early in the dry season (EDS) is the basis of savanna burning for GHG abatement, and so it is often argued that GHG abatement is strongly aligned with biodiversity conservation (Russell-Smith et al., 2013, 2015). However, the notion that EDS burning benefits biodiversity is based on assumption rather than robust evidence (Bowman & Legge, 2016), and it has received little scrutiny. Such scrutiny is important, because elsewhere potential trade-offs between emissions offset schemes and biodiversity are explicitly recognized (e.g., Lindenmayer et al., 2012), and frameworks for addressing trade-offs developed (e.g., Phelps et al., 2012). Given that savanna burning projects are rewarded only for annual GHG abatement, the potential exists for perverse biodiversity outcomes (bioperversity) if there is a lack of congruence between the outcomes of fire management used to reduce GHG emissions and the requirements of fire-sensitive biota (Abreu et al., 2017; Andersen, Woinarski, & Parr, 2012; Archibald, 2011; Richards et al., 2012).

Here, we examine the asserted benefits of savanna burning projects for biodiversity in northern Australia, and the potential for bioperversity. Our aims are two-fold: (a) to provide a critical analysis of the assumption of biodiversity benefit and (b) to offer policy makers and fire managers recommendations for maximizing biodiversity co-benefits (or minimizing potential biodiversity detriment). We focus on the direct effects of savanna burning on biodiversity, rather than long-term, indirect amelioration of climate change that may accrue from managed reductions of emissions.

FIGURE 1 Australia’s northern tropical savannas showing the frequency of (a) late dry season fires (wildfires), (b) all fires from 2000 to 2018, and (c) current savanna burning emissions abatement projects. Note. Indigenous, Indigenous-owned and Indigenous-managed projects (including Indigenous Protected Areas); Government, Government conservation agency; non-Government organization, non-Government conservation organization; pastoral, pastoral properties. Fire history data were obtained from http://www.firenorth.org.au/nafi3/ and savanna burning project shapefiles were obtained from http://www.cleanenergyregulator.gov.au/ERF/project-and-contracts-registers/project-register/project-mapping-files.
<table>
<thead>
<tr>
<th>Taxon and/or community</th>
<th>Description</th>
<th>Fire regime vulnerability</th>
<th>Evidence</th>
<th>Fire management objectives</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Fauna</strong></td>
<td></td>
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<tr>
<td>Small-medium sized mammals</td>
<td>Marsupials and rodents &lt;5.5 kg in body size with restricted home ranges, limited dispersal capacity, and specific habitat requirements[^1-2]</td>
<td>Susceptible to spatially extensive, and frequent fires that remove ground layer, hollow logs, and understorey vegetation,[^1-15] which provide cover from predators[^6-19] and resources such as tree hollows and fleshy fruits[^18-24]</td>
<td>Widespread declines in species richness and abundance[^1, 2, 14, 25] linked with frequent and spatially extensive fire regimes[^8]; modeling predicts decline under pattern of both frequent early and late fires[^3-5, 10, 26]</td>
<td>Provision of small patchy fires and mosaic of burnt and unburnt vegetation (particularly &gt;3 years of age) in woodland communities[^2, 3, 9, 11, 15, 21, 27] and protection of fire refuges (e.g., rainforest patches) through careful early burning</td>
</tr>
<tr>
<td>Bird communities</td>
<td>A range of seed-eating, grass-dwelling, frugivorous, insectivorous, and hollow-using species[^28]</td>
<td>Susceptible to spatially extensive, and frequent fires that remove ground layer[^28-37] and overstory vegetation,[^38, 39] which provide cover from predators or resources such as tree hollows[^40-42], fleshy fruits[^43] and grass seeds[^44]</td>
<td>Widespread declines in abundance and nesting success[^28, 30, 45, 46], linked with spatially extensive and frequent fire regimes[^28]</td>
<td>Provision of small patchy fires and mosaic of burnt and unburnt vegetation, particularly maintenance or increase in area of vegetation &gt;3 years of age[^28-31, 37, 44] in woodlands, and protection of key resources such as older-aged spinifex (Triodia sp.) for nesting and food through careful early burning</td>
</tr>
<tr>
<td><strong>Flora</strong></td>
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<tr>
<td>Cypress pine</td>
<td>Long-lived obligate seeding tree that occurs mostly as small (&lt;0.5 ha) groves embedded within broader savanna landscapes[^47]</td>
<td>Susceptible to frequent and severe fires[^48]; dispersal limited and no persistent seed bank[^49, 50]; juveniles require &gt;10 years before they can survive even low intensity fires[^50-53]; adult trees are quite fecund and recruit profusely in the absence of fire[^54]</td>
<td>Widespread declines[^48, 49, 55, 56] linked with disruption of fine-scale Indigenous fire management[^57-59] and instigation of frequent and spatially extensive fire regime[^60]</td>
<td>Provision of 3–5 year fire-free intervals[^50] and small, low-intensity fires[^58, 59]</td>
</tr>
<tr>
<td>Monsoon rainforest patches</td>
<td>Small (&lt;5 ha) spatially isolated patches of rainforest (supporting plant and animal assemblages that contrast markedly with the extensive savanna) embedded within broader savanna landscape (“dry” forest; semi-deciduous species) or in gullies with perennial moisture (wet forest; evergreen species)[^61, 62]</td>
<td>Composed of mostly re-sprouting species with some tolerance to mild fire[^61-64]; patches not protected by rugged terrain are susceptible to frequent and severe fires[^6-70]; non re-sprouters are very vulnerable to fires[^69]</td>
<td>Despite expansion of some patches linked with wetting trends and possibly atmospheric CO₂ enrichment[^71-75] other patches have declined[^68, 69, 70] and this has been linked to fire frequency and intensity[^70, 76-78]</td>
<td>Provision of small, low-intensity fires, particularly around margins of dry rainforest patches[^48, 69, 77, 79]</td>
</tr>
</tbody>
</table>

(Continues)
TABLE 1 (Continued)

<table>
<thead>
<tr>
<th>Taxon and/or community</th>
<th>Description</th>
<th>Fire regime vulnerability</th>
<th>Evidence</th>
<th>Fire management objectives</th>
</tr>
</thead>
<tbody>
<tr>
<td>Obligate seeding shrubs</td>
<td>Communities of obligate seedling shrubs, for example, “sandstone heath,” that occur in rugged uplands, but also other long-lived shrubs; some communities have EPBC* listing</td>
<td>Many species have juvenile periods of 3–5 years, with some up to 10 years; vulnerable to fires at high frequencies; contemporary patterns of burning deemed too frequent for long-term persistence; modeling predicts further declines under regime of frequent annual fires.</td>
<td>Provision of minimum 3–5 year fire-free intervals to ensure effective replenishment of seedbanks; fires should be small and of low intensity to ensure that enough individuals survive to produce seed.</td>
<td></td>
</tr>
<tr>
<td>Large hollow-bearing savanna trees</td>
<td>Trees containing hollows that provide nesting and roosting sites for mammals and birds (see above)</td>
<td>Frequent fires can lead to top kill of plants (fire trap) preventing trees from reaching maturity; intense fires reduce hollow availability by destroying tree or stag.</td>
<td>Declines in basal area of trees with frequent fire.</td>
<td>Low intensity fires that do not lead to collapse and consumption of hollow trees.</td>
</tr>
<tr>
<td>Other savanna noneucalypt plants</td>
<td>Species that provide resources such as food and shelter and habitats with distinctive faunal assemblages (e.g., riparian areas)</td>
<td>Frequent fires simplify grass communities, decreasing perennial seed availability, resulting in an annual Sorghum sp. dominated understory; dominance of annual Sorghum sp. enhances fuel loads, promoting extensive fires; grass–fire cycle can facilitate spread of invasive species; frequent fires degrade riparian zones reducing tree and shrub cover.</td>
<td>Landscapes burnt less frequently have less annual grasses; fire reintroduced into long unburnt savanna leads to reappearance of annual Sorghum and forb species; riparian zones burnt more frequently have more grass cover and less woody richness and abundance.</td>
<td>Spatially and temporally diverse low-intensity fires (including some wet season burning prior to onset of seed), which do not top kill shrubs, trees, and perennial grass tussocks and hummocks, and maintain ground layer vegetation competition and thereby reduce biomass of flammable annual grasses; riparian areas should be treated the same as monsoon rainforest.</td>
</tr>
</tbody>
</table>

Note. Superscript numbers indicate reference source (reference list is provided in Supporting Information).

*Environment Protection and Biodiversity Conservation Act 1999, which provides a legal framework to protect and manage nationally and internationally important flora, fauna, and ecological communities in Australia.

2 | SAVANNA BURNING IN NORTHERN AUSTRALIA

The key aims of savanna burning are to shift the timing of fire from LDS to EDS, and/or to reduce overall fire frequency and extent (Cook & Meyer, 2009; Russell-Smith et al., 2013; Yates et al., 2015). This is achieved by prescribed burning during the EDS, when relatively moist vegetation, low winds, and lower temperatures produce fires of lower intensity and size (Figures 2 and 3). Such fires result in lower GHG emissions by reducing biomass consumed within the fire footprint (including dead organic matter such as hollow logs and trees; Cook, Meyer, Muepu, & Liedloff, 2016), and by limiting the spread of high intensity fires (Figure 3) that will inevitably occur later in the dry season.

The ERF is a legislated offset scheme that allows land managers to earn Australian Carbon Credit Units (ACCUs) by reducing GHG emissions. These ACCUs are a financial commodity representing 1 ton of CO$_2$ or carbon dioxide equivalent (CO$_2$e) prevented from release into the atmosphere and can be sold to third parties wishing to offset their emissions. For the purposes of the ERF, fires occurring after August 1 (the mid-point of dry season, according to emission reduction criteria) are considered “late,” whereas those before this are “early” (Figure 2). The first project to use this approach was the West Arnhem Land Fire Abatement project (Figure 1c), which, under contract to a multinational energy corporation, has offset over 100,000 tons CO$_2$e per year since 2006 (Russell-Smith et al., 2013; Ansell & Evans, in press).
Rainfall in northern Australia is highly seasonal with >90% of the average annual rainfall (dashed white line) falling between November/December and March/April, during the monsoonal wet season. This rainfall drives vegetation growth, including that of grasses. The intervening period is characterized by little to no rain, and grasses cure and become more flammable as the dry season progresses. Current savanna burning methodology directs that prescribed burning takes place during the early dry season (before August 1). The end of this dry period is characterized by hot temperatures and frequent “dry” storm activity (lightning with little rain) that are conducive for uncontrolled fire conditions until the wet season rains begin. Fire size and intensity is indicated by the size of the flame. Rainfall data were obtained from the Bureau of Meteorology and are long-term averages of major towns in northern Australia.

Across northern Australia, there are now more than 80 savanna burning projects, which include Indigenous, private, and Government-managed protected areas, as well as pastoral leases. Collectively they cover 380,000 km² (ca. 20%) of the savanna (Figures 1c and 4), and include some of Australia’s most structurally intact and biodiverse landscapes (Russell-Smith et al., 2015).

3.1 Fire timing

Savanna burning considers seasonality in terms of fire intensity and fuel consumption but does not consider impacts of timing of fire on biodiversity. The effects of fire on photosynthesis rates and export of carbohydrates to roots or lignotuber and thus plant growth, flowering, and seed production vary markedly with the timing of fire in savannas (Beringer et al., 2015; Prior, Eamus, & Bowman, 2004). Savanna fires have the greatest impacts on plants during the early dry season, because the region is home to many threatened species currently undergoing marked declines (Table 1). Although the extent to which fire is implicated in these declines remains poorly resolved, current evidence suggests that biodiversity responses are more nuanced than the “early fire is good” versus “late fire is bad” dichotomy (Table 1). Below we identify a range of potential conflicts between savanna burning for GHG emissions abatement and biodiversity conservation (Table 2).
## Table 2 Summary of potential negative outcomes for biodiversity (bioperversity) from savanna burning greenhouse gas emissions abatement programs with potential solutions

<table>
<thead>
<tr>
<th>Issue</th>
<th>Description</th>
<th>Potential for bio-perversity</th>
<th>Potential solution</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fire timing</td>
<td>Focus on fire seasonality (EDS vs. LDS) does not consider impact of fire timing on biological responses or life history attributes of fire-sensitive species</td>
<td>Plant growth, flowering, and seed production vary with fire timing. Fires during periods of active growth may be detrimental and have implications for species that are reliant on these resources for food or shelter. Potential reduction in reproductive success for animal species breeding in the EDS (e.g., the ground-nesting partridge pigeon, <em>Geophaps smithii</em>). EDS fires may prolong period of susceptibility to predation for ground-dwelling animals</td>
<td>Instigate adaptive on-ground monitoring of species likely to be impacted by fire timing</td>
</tr>
<tr>
<td>Limited evidence of biodiversity</td>
<td>Studies linking faunal responses to fire regimes are correlative and vary markedly among regions</td>
<td>Generalizations about how biodiversity responds to fire regimes in one region may not apply in other regions</td>
<td>Instigate adaptive on-ground monitoring of species likely to be impacted by altered fire regimes</td>
</tr>
<tr>
<td>responses to fire</td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Older-aged vegetation as a liability</td>
<td>Areas of longer unburnt vegetation seen as risk to projects due to their increased flammability and risk of burning during LDS when emissions are higher. But many resources (e.g., hollows and fruit) used by savanna animals are most abundant in longer unburnt areas</td>
<td>Deliberate fire management to reduce the number and extent of longer unburnt areas (because their fuel load is seen as risk) will diminish habitat for many fire-sensitive species and diminish resources for many other species</td>
<td>Seek to maintain or enhance the extent of longer unburnt vegetation and include an index of this attribute in monitoring. Identify important refugial areas that naturally support long-unburnt vegetation and avoid actively targeting these areas</td>
</tr>
<tr>
<td>Unrealistic distinction</td>
<td>Projects only report on, and are awarded for, annual fire patterns (i.e., what is burnt EDS vs. LDS), and are not based on longer term attributes</td>
<td>Using annual fire patterns as a measure of success may obscure other important fire attributes such as fire sizes, pyro diversity and extent, and distribution of longer unburnt vegetation that are important considerations for fire-sensitive species</td>
<td>Project proponents should also identify and report on targets that relate to spatial and temporal biodiversity-related fire patterns including quantifying patchiness of unburnt vegetation, size of individual fires, and amount of longer unburnt vegetation in woodland communities; nonwoodland communities may require other metrics</td>
</tr>
<tr>
<td>between “early” and “late” fires</td>
<td></td>
<td></td>
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</tr>
<tr>
<td>Conflict between market economy</td>
<td>Fire management for explicit biodiversity outcomes can be expensive because burning needs to be done when fires self-extinguish and may only travel a short distance; repeated visits are required, which adds to operating costs</td>
<td>Burning solely to abate emissions can be achieved by burning when fires will travel further, creating larger burnt areas to protect against LDS fires; these larger fires (while still “early”) can be detrimental to relatively immobile species (e.g., small mammals)</td>
<td>Support the development of co-benefit metrics for biodiversity that are linked to higher ACCU values Note: The 2018 savanna burning methodology now includes carbon sequestered in dead organic matter (hollow logs and dead trees), placing more emphasis on longer unburnt vegetation</td>
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<td>and biodiversity outcomes</td>
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(Continues)
3.2 Limited evidence of biodiversity responses to fire

Most studies in Australian savannas linking faunal responses to fire regimes have design limitations (Andersen et al., 2012; Griffiths & Brook, 2014). For instance, the same variables driving fire mosaics (e.g., rockiness, soil type, local topography, and proximity to water) also influence species’ occurrence and populations. Any association with fire regimes therefore cannot be assumed to be causal. There are few studies where fire can be confidently identified as a causal factor in population trends of threatened fauna (Williams, Woinarski, & Andersen, 2003), and these are often inconsistent. For instance, modeling of mark-recapture data from the Kapalga fire experiment in the Northern Territory (NT) indicated that an increase in fire frequency above once every 5 years increased extinction risk for several small mammals (*Dasyurus hallucatus*, *Trichosurus vulpecula*, *Isoodon macrourus*, and *Melomys burtoni*; Griffiths, Garnett, & Brook, 2015). In contrast, a fire experiment on Melville Island (elsewhere in the NT) indicated that fire frequency did not influence the abundance of *T. vulpecula*, and fire had no effect on mammal diversity (Davies et al., 2018). These differences presumably reflect geographic variation among study systems, and caution against generalizations about how small mammals respond to fire.

3.3 Older-aged vegetation as a liability

Australian savannas are characterized by frequent fire, mostly recurring every 1–4 years (Figures 1 and 5), and consequently only a small proportion of the landscape remains long-unburnt (Figure 5). Long-term and large-scale exclusion of fire in Australian savannas is almost impossible (Russell-Smith et al., 2013). However, exclusion of fire for more modest periods (e.g., 3–10 years) is realistic in some areas and may result in notable improvements in quality of savanna habitats for threatened species (Woinarski & Legge, 2013). Intensive management around fire-sensitive habitats embedded in the dominant savanna matrix (e.g., rainforest patches; Table 1) may provide long-term protection of these distinctive non-savanna environments from fire. However, in the dominant savanna communities longer unburnt areas are often viewed as risks in GHG-abatement programs, because fuel loads typically increase for 3–10 years postfire (Cook & Meyer, 2009; Murphy & Russell-Smith, 2010). Accordingly, some...
GHG-abatement programs deliberately burn such areas to reduce their extent. Such targeted burning is directly at odds with the widely recognized need to “increase” the area of longer unburnt savanna to conserve fire-sensitive biota (Table 1).

3.4 Focus on annual rather than long-term fire patterns

GHG abatement projects report on, and are funded according to, performance on annual fire patterns, rather than longer term fire histories. Therefore, taxa that require older aged vegetation or longer fire-free intervals (Table 1) may be poorly served by schemes that focus on annual fire extent and timing (Perry, Vanderduys, & Kutt, 2016). Evidence that repeated LDS fires are detrimental to obligate seeding plants and sedentary fauna with small home ranges (e.g., Yates et al., 2008) is not evidence that a high incidence of EDS fires is favorable. For fire sensitive taxa, fires (whether EDS or LDS) could still be harmful if they occur frequently or are too extensive (Table 1).

Increasing the area of unburnt country is widely seen as a key goal for conservation management (Andersen et al., 2005), but to date savanna burning projects have not achieved this because the economic imperative is to shift the season of burning within an annual fire cycle (Figure 5; see also Evans & Russell-Smith, 2019; Russell-Smith et al., 2015). This
is not a directly negative consequence of savanna burning, but rather an example of how savanna burning could be suboptimal for biodiversity that requires longer fire-free intervals (see below).

The emissions calculation methodology was updated in 2018 to include carbon sequestered in dead organic matter in addition to avoided GHG emissions (Cook et al., 2016). This changes the focus to longer term fire management, as it places more emphasis on reducing fire frequency and increasing the proportion of the landscape with longer unburnt patches. This is a welcome advance that has potential benefits for biodiversity, such as greater survival of old hollow trees and logs. Nonetheless, it is important that such benefits be demonstrated rather than assumed. Moreover, there is no requirement for existing projects to switch to this new methodology.

### 3.5 Unrealistic distinction between “early” and “late” fires

The contrast between “low-intensity and patchy” EDS fires and “high-intensity and extensive” LDS fires is a generalization; all fire types can occur at any time during the dry season. Furthermore, the “hard” dry season midpoint (August 1) delineating EDS and LDS fires is insensitive to regional and inter-annual variation in the timing and extent of wet season rainfall (Perry et al., 2017). This is particularly the case in low rainfall areas where temporal rainfall patterns vary greatly and there is no relationship between weather patterns and GHG cutoff dates (Perry et al., 2017). Regional and inter-annual variation in rainfall affects grass growth, and hence fuel loads, and curing rates then determine when fires will take, their intensity, rate of spread, and therefore their extent. Consequently, burning to abate GHG emissions can adhere to the methodology yet fail to produce small, patchy, low-intensity fires required by fire-sensitive species (Table 1).

### 3.6 Conflict between market economy and biodiversity outcomes

Fire management (relying heavily on prescribed burning with ignitions dropped from aircraft) for biodiversity conservation is expensive (Russell-Smith et al., 2017). For example, if ignited fires only travel short distances before self-extinguishing, then multiple and repeated ignitions are required. The cheapest option within a market economy is to ignite fires within the regulated EDS period when fires will burn extensively with the fewest ignitions and minimized flight times. Thus, ACCUs can be maximized by extensive EDS burning, rather than creating many small, low-intensity fires that are favored by some taxa (Table 1).

### 3.7 Interactions between prescribed fire and other threatening processes

Fire can interact with other threatening processes in a way that is not considered by savanna burning. For instance, EDS burning can exacerbate grazing pressure by introduced herbivores (Table 1; see also Legge et al., 2019). If this is repeated, it can lead to decreases in landscape condition, loss of understory vegetation, and loss of structural cover for threatened taxa (Table 1), creating conditions favorable to invasive predators due to increased hunting success (Table 1). Failure to consider the interactions of fire with other threatening processes could amplify negative impacts of these threats (Table 1).

### 3.8 Sacrificial areas and fire breaks

A commonly used method in prescribed burning is to repeatedly burn topographic features such as water courses and valley systems to create barriers to protect other areas (e.g., Murphy, Cochrane, & Russell-Smith, 2015; Price, Edwards, & Russell-Smith, 2007). This approach may help to reduce the risk of LDS fires in nearby habitats, but may be detrimental to intrinsic biodiversity values, such as riparian vegetation and its distinctive fauna assemblages (Table 1). Furthermore, such repeated burning can exacerbate the effects of grazing and trampling by large herbivores.

### 3.9 Maximizing biodiversity co-benefits from savanna burning

Although savanna burning may lead to some beneficial biodiversity outcomes, it could be perceived as obviating the need for dedicated fire management for specific biodiversity benefit, and thus have suboptimal outcomes for species requiring a finer-scale burning approach (Table 1). Monitoring of threatened small mammals in the Northern Kimberley (nearly entirely managed for GHG abatement, Figure 1c) shows a correlation between species richness and abundance, and longer unburnt vegetation (Figures 6a and 6b). Here, savanna burning has increased EDS fires and reduced the area burnt by LDS fires (Figure 6c), but the total fire extent has slightly increased (Figure 6c) and there has been no increase in the extent of longer unburnt vegetation (Figures 5e and 5f), suggesting that outcomes that most benefit biodiversity are not being achieved.

If prescribed burning interacts negatively with other threatening processes, this could also produce suboptimal outcomes. For instance, EDS fires can be detrimental if the resulting burnt areas are too large, because they expose small mammals to enhanced predation by feral cats (Table 1). Similarly, frequent EDS burning may lead to a grass–fire cycle of flammable native annual and exotic perennial grasses (Table 1). If frequent EDS burning promotes the spread of invasive grasses (e.g., Andropogon gayanus; Table 1), then
Monitoring in the Northern Kimberley demonstrates the relationship between vegetation age (years since last burnt) and small mammal richness (a) and abundance (b). Monitoring plots \((n=91)\) have been classed into rocky (rugged sandstone) and non-rocky (open woodland) habitat types. Fire scar imagery at the time of trapping (see http://www.firenorth.org.au/nafi3/) was used to determine vegetation age at each plot, which is 0.25 ha and contains a standard set of 24 traps (various types) and is operated for 120 trap-nights per year. Average area burnt in two assessment periods: baseline (no fire management, 2000–2008) and the last 10 years (fire-managed, 2009–2018) shows that fire management has increased the extent of early dry season fires and reduced the extent of late dry season fires but has not increased the extent of unburnt area (c). Bars are standard errors.

3.10 | Traditional and contemporary approaches to savanna fire management

Although savanna burning (and EDS fire management generally) is intended to partly redress the loss of pre-European Aboriginal fire management (Russell-Smith et al., 2013), there are differences between the pre-European fire regimes and contemporary fire management for GHG abatement. Under current savanna burning methodology, there is an economic incentive and contractual obligations to apply fire in a particular way (Fache & Mozio, 2015; Perry et al., 2018), and financial penalties for fires that occur after August 1 across the entire savanna (Figure 2).

Traditional burning was not motivated by money, western conservation ethic, or a binary two-season approach (Petty, deKoninck, & Orlove, 2015). Rather it occurred throughout the entire year to fulfil cultural obligations, to facilitate passage through the landscape, and to attract game animals, and implementation varied from region to region (Preece, 2002). The application of fire was gradual as people moved around the landscape on foot, lighting many small fires as vegetation cured and became flammable. However, as the number of Aboriginal people living on and managing the landscape has markedly decreased, and most people reside in regional centers or larger towns, aerial prescribed burning has largely replaced walking, and fire is now applied to the landscape in a very short period of time (Figure 2). Contemporary fires applied for GHG abatement therefore do not replicate past fire regimes (Petty et al., 2015).

4 | THE NEED FOR BETTER BIODIVERSITY ACCOUNTING IN SAVANNA BURNING

We are not suggesting that savanna burning projects are having a detrimental impact on biodiversity. Rather, we are arguing that there is “potential” for this to be occurring, and that it is not possible to know if this is being realized without a better understanding of how the spatial and temporal arrangement of fires influences biodiversity, and interactions with other threatening processes (Driscoll et al., 2010; Parr & Andersen, 2006), or directly monitoring biodiversity outcomes. Such monitoring, however, is not a requirement of savanna burning projects. Prescribed burning is no different to any other management intervention; its effectiveness and impacts across a range of values need to be adequately reviewed (Legge, 2015).

Currently, monitoring that evaluates savanna burning is simple and inexpensive: freely available fire scar imagery...
is analyzed to determine the annual extent of EDS and LDS burning relative to a target based on a nominated level of improvement from the fire regime existing before the establishment of the program (e.g., Figures 5a, 5b, 6c, and 6d). Biodiversity monitoring is more complex because biodiversity responses may take years to be realized, will require on-ground biological surveys, and biodiversity may have varied and nuanced responses among different species, habitats, and regions. Assessment of the impacts of savanna burning on biodiversity will also have stronger inference if based on a Before-After-Control-Impact design.

Partnerships among organizations can assist in the design, implementation, and interpretation of biodiversity monitoring programs (e.g., Austin et al., 2017; Gillespie, Stevens, Mahney, Legge, & Low Choy, 2015). Such programs can be designed to report on trends in key biodiversity features (e.g., Table 1) or biodiversity surrogates or indicators (e.g., extent of longer unburnt savanna (Figures 5e and 5f) and fire-sensitive nonsavanna habitats). However, there is currently no ready funding sources for such monitoring. Given the increasing number of projects (Figure 4) with multiple outcomes, those intending to incorporate biodiversity monitoring may need further support; or existing programs may need to be modified to ensure adequate provision for biodiversity monitoring.

It is important to recognize that there is no “one-size-fits-all” approach (Perry et al., 2016). Bioperversity may be avoided by incorporating the requirements of threatened and fire-sensitive taxa (Table 1) into the design and implementation of fire management programs. If potential trade-offs are identified, their broader implications can be transparently measured. For instance, if frequent EDS burning is good for abating GHG emissions and community livelihoods (e.g., financial rewards), but detrimental to threatened taxa (Table 1), then decisions about which interests hold precedence, or options for compromises, can be developed.

## 5 | CONCLUSION

Savanna burning has proven to be a successful tool for incentivizing improved fire management in northern Australia, while providing important social, economic, and broader environmental benefits (Ansell & Evans, in press). It also has the potential to provide biodiversity benefits (Evans & Russell-Smith, 2019). However, this potential has not been demonstrated, and has received little scrutiny. As a result, proponents contracted to these schemes may inadvertently subvert biodiversity values, or at least fail to optimize biodiversity benefits. The management of fire for biodiversity conservation needs to explicitly consider the responses of taxa that may be disadvantaged by any or all fire regimes (Table 1). Here, we have identified a range of potential conflicts (and solutions; Table 2) between savanna burning for GHG abatement and biodiversity outcomes. If these are addressed during project planning stages, then biodiversity co-benefits can be maximized (Table 2).

Finally, proponents should not make claims of biodiversity co-benefits unless these are demonstrable, and where they are, proponents should be able to seek premium funding. For a market-based economy to affect change in GHG emissions while supporting biodiversity, a more flexible framework is required; and, given the typically slow rate of biodiversity recovery, market payments should relate to long-term performance. The rapid uptake of savanna burning (Figure 4) indicates that if similar incentives were introduced for fire management that maximizes biodiversity benefits, then transformational change could be achieved. A successful biodiversity accounting system will require funding for biodiversity monitoring that complements the accounting of GHG emissions, and also payments that respond in the short- and long-term to indication of biodiversity benefits.

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## REFERENCES


**SUPPORTING INFORMATION**

Additional supporting information may be found online in the Supporting Information section at the end of the article.

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