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# Opportunities and challenges for savanna burning emissions abatement in southern Africa

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ABSTRACT

Savanna fires occurring in sub-Saharan Africa account for over 60% of global fire extent, of which more than half occurs in the Southern Hemisphere contributing ~29% of global fire emissions. Building on experience in reducing savanna fire emissions in fire-prone north Australian savannas through implementation of an internationally accredited 'savanna burning' emissions abatement methodology, we explore opportunities and challenges associated with the application of a similar approach to incentivise emissions reduction in fire-prone southern African savannas. We first show that for a focal region covering seven contiguous countries, at least 80% of annual savanna large fire (>250 ha) extent and emissions occur under relatively severe late dry season (LDS) fire-weather conditions, predominantly in sparsely inhabited areas. We then assess the feasibility of adapting the Australian emissions abatement methodology through exploratory field studies at the Tsodilo Hills World Heritage site in north-west Botswana, and the Niassa Special Reserve in northern Mozambique. Our assessment demonstrates that application of a savanna burning emissions abatement method focused on the undertaking of strategically located early dry season (EDS) burning to reduce LDS wildfire extent and resultant emissions meets key technical criteria, including: LDS fine fuels tend to be markedly greater than EDS fuels given seasonal leaf litter inputs; LDS fires tend to be significantly more severe and combust more fuels; methane and nitrous oxide emission factors are essentially equivalent in EDS and LDS periods under cured fuel conditions. In discussion we consider associated key implementation challenges and caveats that need to be addressed for progressing development of savanna burning methods that incentivise sustainable fire management, reduce emissions, and support community livelihoods in wildfire-dominated southern African savannas.

## Author credits

**Jeremy Russell-Smith:** Supervision, Conceptualisation, Investigation, Formal analysis, Writing, review and editing. **Cameron Yates:** Conceptualisation, Investigation, Formal analysis, Writing, Review and editing. **Roland Vernooij:** Conceptualisation, Investigation, Formal analysis, Writing, Review and editing. **Tom Eames:** Conceptualisation, Investigation, Formal analysis, Writing, Review and editing. **Guido van der Werf:** Supervision, Funding acquisition, Conceptualisation, Investigation, Formal analysis, Review and editing. **Natasha Ribeiro**: Supervision, Investigation, Review and editing. **Andrew Edwards**: Investigation, Review and editing. **Robin Beatty**: Investigation, Review and editing. **Othusitse Lekoko**: Review and editing. **Jomo Mafoko**: Supervision. **Catherine Monagle**: Funding acquisition, Writing, Review and editing. **Sam Johnston:** Funding acquisition, Writing, Review and editing.

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## 1. Introduction

Tropical savannas constitute the most fire-prone of Earth's biomes, currently annually accounting for almost 90% of global burned area (Giglio et al., 2018) and 62% of global fire carbon emissions (van der Werf et al., 2017). Over the past two decades, savanna fires occurring in sub-Saharan Africa have accounted for over 60% of global fire extent, of which more than half has occurred in the Southern Hemisphere (Giglio et al., 2013b, 2018) contributing ~29% of global emissions (van der Werf et al., 2017). Most savanna fire extent is derived from human ignitions and occurs especially in the latter months of the annual dry season typically as wildfire under relatively severe fire weather (hot, windy, low humidity) conditions (e.g. Williams et al., 2002; Archibald et al., 2010; Pivello, 2011). It has been proposed that reducing savanna fire emissions, particularly through reduction in the extent and severity of late dry season (LDS) fires by undertaking strategic prescribed burning in the early dry season (EDS), potentially would contribute to significant GHG emissions reduction and, in specified circumstances, ecological and livelihood benefits (Russell-Smith et al., 2013a,b; Lipsett-Moore et al., 2018; Moura et al., 2019).

In their recent assessment, Lipsett-Moore et al. (2018) examined the global emissions reduction potential of undertaking early dry season (EDS)-focused fire management through implementation of the savanna burning emissions abatement methodology as currently applied in Australia (CoA 2015, 2018). They found that of 50 countries with eligible savanna-type vegetation and minimum mean annual rainfall (MAR) > 600 mm, 35 countries met criteria of mean (2000–2014) LDS emissions >50,000 t.$CO_2$-e $yr^{-1}$ (source: van der Werf et al., 2017), and relatively unpopulated protected areas >1000 $km^2$ suitable for EDS fire management. Of 30 African countries assessed with total mean emissions >50,000 t.$CO_2$-e $yr^{-1}$, LDS emissions exceeded EDS emissions by > 200,000 t.$CO_2$-e $yr^{-1}$ in 19 countries, and, in a further 7 countries where EDS emissions exceeded LDS emissions, LDS emissions were >200,000 t.$CO_2$-e $yr^{-1}$. The authors observed that 17 Least Developed Countries, all occurring in Africa, could potentially abate 37% (64.2 Mt$CO_2$-e $yr^{-1}$) of accountable global savanna fire emissions, albeit based on the unrealistic premise that all LDS anthropogenic fires should, and lightning ignitions could, be eliminated.

Since regulatory approval from 2012 for the undertaking of savanna burning emissions reduction projects in Australia, registered projects now operate across ~25% of the eligible 1.2 M $km^2$ tropical savannas region receiving >600 mm MAR (CoA, 2020; Edwards et al., 2020). Such projects have proven to be effective in significantly reducing emissions (Russell-Smith et al., 2013a; CoA, 2019) and substantially meeting Indigenous (Aboriginal) cultural and enterprise aspirations (Ansell et al., 2020; McKemey et al., 2020). However, application in broader pastoral land use (Cowley et al., 2014; Walsh et al., 2014) and ecological (Andersen et al., 2012; Corey et al., 2020) contexts can be contentious. Hence, while longer running (from the mid-2000s) commercial savanna burning projects have demonstrated significant ecological management outcomes at landscape scales (e.g. reductions in LDS wildfires, more patchy and less severe fire regimes, reduced burnt area overall; Evans and Russell-Smith, 2020; Edwards et al., 2020), they are not designed to meet a variety of ancillary requirements, for example: fine-scale conservation of fire-vulnerable habitats, fauna with small (hectare-scale) home ranges, poorly dispersed fire interval-sensitive (obligate seeder) plant taxa (Woinarski et al., 2005; Yates et al., 2008); controlling woody thickening (i.e. plant encroachment) in pastoral situations (Cowley et al., 2014; Walsh et al., 2014). Such specific management objectives require additional targeted investment; savanna burning projects are not a management panacea (Evans and Russell-Smith, 2020).

In broader global savanna management contexts similar considerations and caveats apply, albeit varying in type and magnitude under different local and regional land use settings, including: woody encroachment associated particularly with over-grazing and limited burning in southern Africa (e.g. Acocks, 1953; O'Connor et al., 2014; Archibald, 2016), fire suppression policies in South American cerrado (Durigan and Ratter, 2016), additionality of $CO_2$ fertilisation effects (Kgope et al., 2010; Bond and Midgley, 2012; Donohue et al., 2013); impacts of woody encroachment on floristic and wildlife species diversity associated with loss of open grazing (Parr et al., 2014) and avian (Sirami and Monadjem, 2012) habitat; and concerns regarding the misapplication of formal carbon sequestration schemes (e.g. Clean Development Mechanism [CDM], REDD+) promoting forest cover and forestry (Parr et al., 2014; Abreu et al., 2017). Savanna burning projects need to be responsive and adaptive to such issues, recognising that savanna woody cover is dynamic with respect to a variety of climatic, disturbance, and societal drivers (e.g. Scholes and Archer, 1997; Sankaran et al., 2005; Donohue et al., 2013; Poulter et al., 2014; Archibald, 2016).

In this paper we ask the question whether application of a savanna burning emissions reduction approach comparable to that implemented in Australia is technically feasible under fire-prone southern African conditions. To address this we take advantage of a pilot study in sparsely populated, frequently and severely burnt, woody savannas of Ngamiland, north-west Botswana, and complementary research in northern Mozambique. For regional context, we first consider contemporary seasonal fire patterning and associated emissions from southern African savannas. We then describe the essential components of the accounting methodology and its practical application, followed by technical assessment of key seasonal (EDS, LDS) savanna burning emissions accounting parameters (e.g. fire extent, fuel types and accumulation, fuel combustion, greenhouse gas [GHG] emission factors) derived from assembled remotely sensed data and field observations at our study sites. Finally, we consider critical implementation challenges and caveats associated with potential broader application of savanna burning emissions reduction projects in frequently and severely burnt southern African regional savanna landscapes.

## 2. Southern Africa regional context

For this assessment we consider seasonal fire and emissions patterning in savanna biomes focused on Botswana and surrounding southern African countries including Namibia, Angola, Zambia, Zimbabwe, Mozambique and South Africa (Fig. 1a–g). Mapping surfaces are derived from the following sources: the savanna biome, broadly defined at the Division level (Fig. 1a) following Sayre et al. (2013); mean seasonal rainfall distribution, 1979–2019, (Fig. 1b–d) (source: ERA5, total precipitation data—https://cds.climate.copernicus.eu/cdsapp#!/dataset/reanalysis-era5-single-levels?tab=overview); mean seasonal fire extent, 2001–19 (Fig. 1e), derived from Collection 6 MODIS 500m burned area data (after Giglio et al., 2018); mean annual fire emissions, 2003–19 (Fig. 1f; and Appendix S1 for country-level seasonal fire emissions), applying 0.25° GFED4s data (after van der Werf et al., 2017); and regional population density distribution, 2019 (Fig. 1g), after CIESIN (2018). With respect to fire extent data, of particular note is that the MODIS 500m burnt area product has been demonstrated to significantly underestimate fire patch sizes <250 ha (Roteta et al., 2019). As such, fire extent and emissions data presented below refer specifically to larger, typically more intense fires representative of wildfire conditions (van der Werf et al., 2017).

Mean annual rainfall (MAR) over the focal savanna region ranges from 300 to >1400 mm (Fig. 1b). For illustrative purposes, we have divided fire extent and emissions into early dry season (EDS—May-June) and late dry season (LDS—July-October) components based on equal division of the mean annual driest 6-month period describing the majority of the savanna region (Fig. 1c). Archibald et al. (2009) found that dry season length of at least six months was strongly associated with fire occurrence in southern African savannas. We acknowledge that this does not fully represent dry seasonal conditions for parts of the north-west and south-east where the driest 6-month period commences one month earlier. Of note, as defined here, the mean proportion of rain received in this 6-month dry season period (Fig. 1d) accords generally with equivalent conditions under which savanna burning projects are undertaken in north Australian savannas, where <10% MAR is received in the driest 6-month period, typically May–October (source: Australian Bureau of Meteorology: www.bom.gov.au).

While recognising that contemporary fire extent in southern Africa exhibits considerable variability with respect both to anthropogenic factors (Archibald, 2016) and especially rainfall dynamics (Wei et al., 2020), over the past two decades relatively high frequencies (>0.5 $yr^{-1}$) of large-sized fires (Fig. 1e) are shown to be significantly ($p < 0.01$) negatively associated with human population densities (Fig. 1g) across the focal southern African savanna region overall, especially under MAR conditions >1000 (Table 1). Higher GHG emissions (Fig. 1f) are notably concentrated in northern higher rainfall regions (Fig. 1b).

Savannas comprise >75% of the vegetation of all focal southern African countries save Namibia (59.8%) and South Africa (24.1%) (Table 2). The mean annual extent of burning of savannas by large fires (2000–19) ranges between 23 and 27% in three countries (Zambia, Mozambique, Angola), between 7 and 9% in a further three countries (Botswana, Zimbabwe, Namibia), and <3% in South Africa (Table 2). Most savanna fire extent occurs under high rainfall conditions characteristic of respective countries save Angola and South Africa (Table 2). In all countries at least 80% of savanna large fire extent occurs in the LDS, and >94% in four (Zimbabwe, Mozambique, Botswana, Namibia; Table 2). Unsurprisingly, the proportion of LDS emissions from savanna fires in respective countries follows closely the same patterns as exhibited by large-sized fire extent (Table 2).

## 3. Savanna burning emissions abatement methodology

### 3.1. Background

Following Seiler and Crutzen (1980) and IPCC (1997), in its most basic form, savanna burning emissions (*E*) may be calculated as the product of the mass of fuel pyrolised (*M*) and the emission factor (*EF*) of respective accountable GHG (*g*) species:

$$E = M * EF(g) \tag{Eq 1}$$

where, *M* is the product of the area exposed to fire (*A*) taking into account spatial unburnt patchiness (*P*), the fuel load (*FL*) in respective fuel type classes (*FLc*—e.g. grass, tree litter, woody debris) of defined vegetation fuel types (*VFTs*), and the combustion completeness (*C*) defined as the mass of fuel exposed to fire that is pyrolised. *EF(g)* is defined relative to fuel elemental content where, for carbon species, *EF (g)* is expressed relative to fuel carbon, and nitrogen species are expressed relative to fuel nitrogen. Fuel carbon mass is determined from fuel mass by the fuel carbon content, while fuel nitrogen is derived from the fuel mass by the product of carbon content and the fuel nitrogen to carbon ratio. GHG emissions accounting from savanna fires typically considers emissions of methane ($CH_4$) and nitrous oxide ($N_2O$).

Iterations of Australia's savanna burning methodology (e.g. CoA , 2015, 2018) include substantial enhancements to this basic emissions calculation framework (Russell-Smith et al., 2009; Meyer et al., 2012; Murphy et al., 2015). Although recent enhancements include transitioning to a methodological approach combining accounting both of emissions and sequestration components (e.g. CoA , 2018), our focus here solely addresses emissions accounting. A generalised description of the application of the emissions methodology at a project scale, including associated emissions accounting and operational procedures, is given in Russell-Smith et al. (2013a).

A particular feature of the Australian approach is recognition of contrasting emissions profiles produced under different seasonal fire-weather conditions—accounting for less fuel consumption under relatively mild EDS conditions associated with enhanced spatial unburnt patchiness (*P*) and less fuel pyrolisation (CC), *vs*. extensive, intense wildfires under LDS (high temperature, low humidity, gusty) conditions (Russell-Smith et al., 2009; Yates et al., 2015). Fine fuel loads (grass, tree litter < 6 mm diameter) typically are greater in the LDS given progressive dry season litterfall (Cook, 2003; Yates et al., 2020).

Under north Australian seasonal conditions, the EDS burning period is defined as pre-August, and the LDS thereafter. Although this fixed temporal division is evidently somewhat arbitrary (Perry et al., 2020), especially for a 1.2 M $km^2$ region spanning ~3000 km of longitude, it reflects generally the time of year after which fires begin to burn through the night (Maier and Russell-Smith, 2012) and accords well with time-tested Aboriginal seasonal calendars regarding the onset of hot, highly flammable LDS conditions (Russell-Smith et al., 2003; Garde et al., 2009; Vigilante et al., 2009). Notably, however, work is underway to replace these seasonal defaults with remote sensing methods that independently measure fire intensity and severity characteristics (e.g. Edwards et al., 2018).

EDS fire management focuses on progressively burning ground fuels as they cure—e.g. commencing with elevated sites then progressively burning downslope onto moister, less flammable fuels—to strategically reinforce natural and built barriers, enhance fuel load discontinuity at landscapes scales through imposition of a mosaic of burnt and unburnt patches, and thereby reduce the risk of unwanted LDS wildfires. Under such Australian cured fuel conditions, *EF($CH_4$)* and *EF($N_2O$)* have been shown to be equivalent in EDS and LDS periods (Hurst et al., 1994, 1996; Meyer et al., 2012), albeit exhibiting substantial variation associated with different vegetation (*VFT*) and fuel types (*FTc*) (Meyer et al., 2012). Conversely, various African savanna studies have demonstrated that *EF ($CH_4$)* from Miombo woodlands and seasonally inundated dambo grasslands are strongly influenced by fuel moisture content (Hoffa et al., 1999; Korontzi et al., 2003a, 2003b, 2004; Korontzi, 2005). Undertaking prescribed EDS fire management under such enhanced moist fuel conditions would necessarily confound the putative emissions abatement benefits of the proposed method.

### 3.2. Study sites

We provide a case study assessment of the application of the Australian savanna burning emissions accounting approach focused on the World Heritage Tsodilo Hills Enclave region, Ngamiland, north-west Botswana. Additionally, we also report results from a complementary *EF (g)* assessment undertaken in Niassa Special Reserve, Mozambique. Specifically, our assessment is based on exploratory field studies undertaken in 2019, both in EDS and LDS periods, addressing key seasonal accounting parameters that might be expected to support application of a prescribed EDS fire management approach for mitigating emissions from LDS wildfires.

The 5726 $km^2$ Tsodilo study site encompasses mostly sparsely tree-covered, shrub-dominated Dry Savanna (after Sayre et al., 2013), occupying strongly east-west oriented infertile sandy Kalahari dunefields, interspersed with typically linear interdune depressions and omuramba, relatively fertile ancient riverbeds. Despite being seasonally dry, omuramba support large numbers of 'cattle posts'—sites typically located around a water bore which provide water for livestock and sparse settlement through the 7–8 month (Apr–Nov) annual dry season. It follows that dry season grazing by domestic stock (cattle, donkeys, goats) is intense in association with, and especially in vicinity of, omuramba cattle posts, with resultant reduced fuel availability and fire frequency, and woody plant encroachment. Vegetation in the western sector intergrades with a structurally better developed, floristically attenuated form of Miombo Broadleaf Savanna (after Sayre et al., 2013), referred to here as Woodland Savanna. Regionally, both Dry Savanna and Woodland Savanna types comprise Kalahari Woodlands as described by Mendelsohn and Roberts (1997). MAR of the study area is ~600 mm, highly seasonally and annually variable.

The Mozambican study site is situated adjacent to the Chiuwexi (or R4) block in the 42,000 $km^2$ Niassa Special Reserve (NSR) (Fig. 1a;

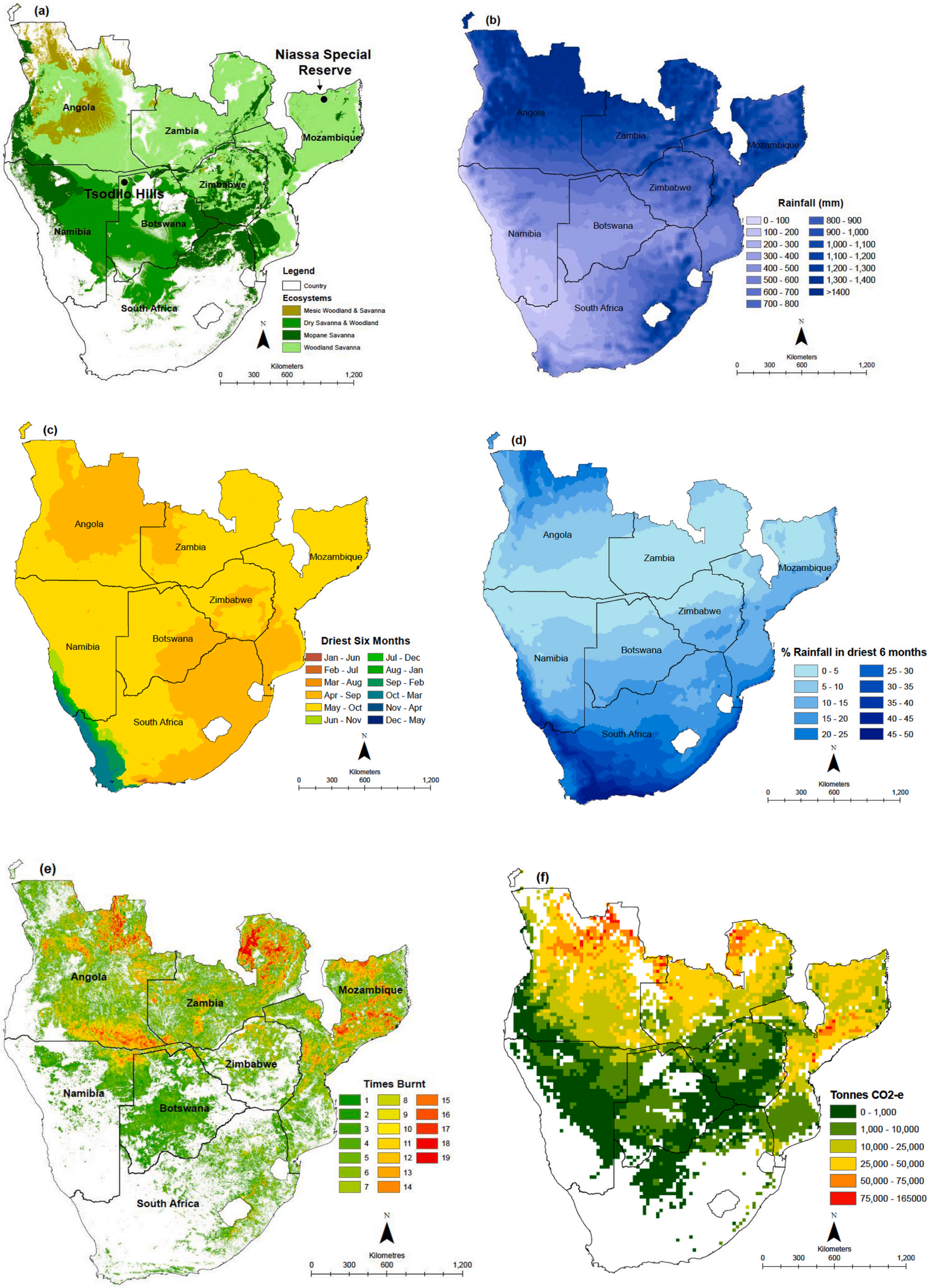
(a)
Niassa Special Reserve
Angola
Zambia
Mozambique
Tsodilo Hills
Zimbabwe
Namibia
Botswana
South Africa
Legend
Country
Ecosystems
Mesic Woodland & Savanna
Dry Savanna & Woodland
Mopane Savanna
Woodland Savanna
N
Kilometers
0 300 600 1,200
(b)
Angola
Zambia
Mozambique
Zimbabwe
Namibia
Botswana
South Africa
Rainfall (mm)
0 - 100
100 - 200
200 - 300
300 - 400
400 - 500
500 - 600
600 - 700
700 - 800
800 - 900
900 - 1,000
1,000 - 1,100
1,100 - 1,200
1,200 - 1,300
1,300 - 1,400
>1400
N
Kilometers
0 300 600 1,200
(c)
Angola
Zambia
Mozambique
Zimbabwe
Namibia
Botswana
South Africa
Driest Six Months
Jan - Jun
Feb - Jul
Mar - Aug
Apr - Sep
May - Oct
Jun - Nov
Jul - Dec
Aug - Jan
Sep - Feb
Oct - Mar
Nov - Apr
Dec - May
N
Kilometers
0 300 600 1,200
(d)
Angola
Zambia
Mozambique
Zimbabwe
Namibia
Botswana
South Africa
% Rainfall in driest 6 months
0 - 5
5 - 10
10 - 15
15 - 20
20 - 25
25 - 30
30 - 35
35 - 40
40 - 45
45 - 50
N
Kilometers
0 300 600 1,200
(e)
Angola
Zambia
Mozambique
Zimbabwe
Namibia
Botswana
South Africa
Times Burnt
1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
19
N
Kilometres
0 300 600 1,200
(f)
Tonnes CO2-e
0 - 1,000
1,000 - 10,000
10,000 - 25,000
25,000 - 50,000
50,000 - 75,000
75,000 - 165000
N
Kilometers
0 300 600 1,200

**Fig. 1.** Key characteristics of southern African study region: (a) savanna vegetation Ecosystem Divisions (Sayre et al., 2013), and locations of Tsodilo Hills Enclave (Botswana) and Niassa Special Reserve (Mozambique) study locales; (b) mean annual rainfall (MAR); (c) mean rainfall of consecutive driest six months; (d) mean rainfall of consecutive driest 6 months as proportion of MAR; (e) fire frequency, 2001–2019; (f) fire emissions ($CO_2$-e), 2003–2019 (g) human population density. Refer Section 2 for methods and data sources.

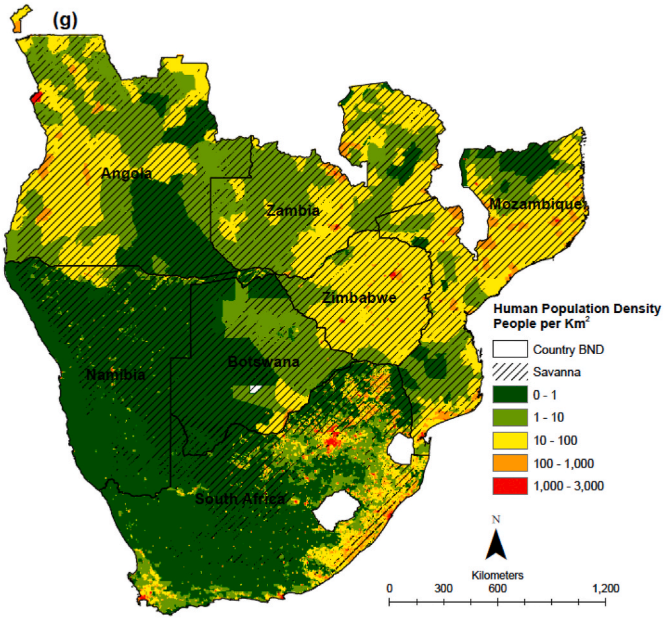


**Fig. 1.** (*continued*).

**Table 1**
Statistical tests (two-tailed *t*-test with unequal variance) comparing mean population density (persons km$^{-2}$) at sites representative of different fire frequencies (number of times burnt 2001–19, after Giglio et al., 2018) per rainfall class (source: ERA5, refer Section 2), for 0.25$^0$ cells, in the southern Africa focal savanna region (refer Fig. 1a). Significant differences ($p < 0.01$) between rainfall class means denoted by different superscript letters. Number of observations ($n$) given in parentheses.

| Rainfall class (mm) | Population density (persons km$^{-2}$) | | |
|---|---|---|---|
| | Fire frequency | | |
| | 0 ($n$ = 2025) | 1-9 ($n$ = 2452) | 10-19 ($n$ = 816) |
| <500 ($n$ = 640) | 7.7$^a$ | 8.0$^a$ | – |
| 501–1000 ($n$ = 2231) | 69.0$^a$ | 25.3$^a$ | 14.6$^a$ |
| 1001–1500 ($n$ = 1971) | 76.4$^a$ | 24.1$^a$ | 17.4$^b$ |
| >1500 ($n$ = 451) | 17.9$^a$ | 16.2$^a$ | 7.6$^b$ |
| *Total* ($n$ = 5293) | 52.7$^a$ | 23.1$^a$ | 15.2$^b$ |

Mbanze et al., 2019). The vegetation is predominantly Woodland Savanna (i.e. Miombo and Associated Broadleaf Savanna, *sensu* Sayre et al., 2013; Dry Miombo Woodland *sensu* Timberlake and Chidumayo, 2011; Dry Zambezian Miombo Woodland *sensu* Ribeiro et al., 2008), interspersed by seasonally flooded grasslands (dambo) in lower lying more fertile areas. MAR (900 mm) is substantially higher and with a typically longer (Oct–Apr) wet season (Mbanze et al., 2015) than at Tsodilo. The NSR is sparsely populated, and domestic livestock grazing is limited given tsetse fly impacts and predation, although native herbivore grazing likely affects fuel load particularly in dambo. Frequent anthropogenic LDS fires are a major ecological factor in the area (Mbanze et al., 2015). With reference to the Collection 6 MODIS 500m burned area archive, mean annual fire frequency of the entire NSR for the period 2001–19 was 0.55 yr$^{-1}$, comprising predominantly LDS fires (0.51 yr-$^1$).

### 3.3. *Sampling methodology and fire treatments*

Remote sensing and field methods were adapted from equivalent Australian studies for assessment at Tsodilo, and novel drone-based assessments were utilised to assess *P* at Tsodilo and *EF(g)* both at Tsodilo and Niassa.

#### 3.3.1. *Tsodilo assessment*

*3.3.1.1. Burnt area and VFT mapping.* Burnt area mapping was derived for the Tsodillo region using all available cloud-free Landsat satellite imagery (30 m pixels; acquired every 16 days), 2014–2019, for path-row 173–76. Following Evans and Russell-Smith (2020), a difference image of the red and near infra-red bands for sequential image date pairs was derived with a 1–99% image stretch applied to the difference image. A density slice was then performed to the difference image to classify burnt and unburnt area. A month value was assigned to each mapped image and then compiled for each year. The mapping for the six-year period was then used to create fire frequency and time-since-fire layers. Validation of the mapping method was undertaken in September 2019 by helicopter survey, covering all major Vegetation Fuel Types (*VFTs*; see below). To compare our relatively fine-resolution Landsat-derived fire mapping with mapping derived from more commonly used, but relatively coarse resolution, MODIS imagery (500 m pixels; acquired daily), we collected 342 validation points sampled at 30 s intervals, and intersected these with 2019 Landsat and MODIS satellite-derived fire mapping products current at time of survey. Overall accuracy for Landsat mapping was 95%, and 93% for MODIS mapping products.

*VFT* mapping followed procedures outlined in Lynch et al. (2015), derived from Sentinel 2 satellite imagery. Using Google Earth Engine (Gorelick et al., 2017), a median image was derived from cloud-free images, January 2015–April 2020. Object-based image segmentation

**Table 2**
Mean large-fire (>250 ha) burnt area and fire emissions per country, savanna biome, and MAR classes, southern Africa, where the Late Dry Season (LDS) is defined here as commencing July 1. Refer Section 2 for fire burnt area and emissions data sources.

| Country | Area | Savanna | Mean savanna area burnt[a] 2000–19 | | Mean savanna emissions[a] 2003–19 | | |
|---|---|---|---|---|---|---|---|
| | ($km^2$) | (%) | Proportion of savanna area burnt (%) | *Proportion burnt in LDS (%)* | *EDS (Mt $CO_2$-e)* | *LDS (Mt $CO_2$-e)* | % LDS |
| ***Angola*** | 1,228,569 | 77.5 | 23.0 | 79.5 | 7.628 | 26.949 | 77.9 |
| <500 mm | 43662 | 3.6 | 0.8 | 75.1 | 0.004 | 0.010 | 69.1 |
| 500–1000 mm | 339270 | 27.6 | 31.4 | 88.3 | 1.051 | 5.882 | 84.8 |
| >1000 mm | 569530 | 46.4 | 21.9 | 73.9 | 6.573 | 21.057 | 76.2 |
| ***Botswana*** | 599,389 | 75.6 | 9.0 | 95.0 | 0.069 | 1.127 | 94.2 |
| <500 mm | 361573 | 60.3 | 7.3 | 93.4 | 0.047 | 0.562 | 92.3 |
| 500–1000 mm | 91356 | 15.2 | 16.2 | 98.0 | 0.022 | 0.565 | 96.2 |
| >1000 mm | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| ***Mozambique*** | 792,385 | 85.4 | 26.5 | 95.8 | 0.868 | 19.137 | 95.7 |
| <500 mm | 36439 | 4.6 | 6.5 | 84.5 | 0.037 | 0.176 | 82.5 |
| 500–1000 mm | 413094 | 52.1 | 24.5 | 93.7 | 0.699 | 9.631 | 93.2 |
| >1000 mm | 227536 | 28.7 | 33.4 | 99.0 | 0.132 | 9.331 | 98.6 |
| ***Namibia*** | 855,281 | 59.8 | 7.0 | 94.3 | 0.077 | 1.187 | 93.9 |
| <500 mm | 329346 | 38.5 | 2.8 | 93.6 | 0.015 | 0.205 | 93.1 |
| 500–1000 mm | 181829 | 21.3 | 14.8 | 94.5 | 0.062 | 0.982 | 94.1 |
| >1000 mm | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| ***South Africa*** | 1,339,141 | 24.1 | 2.8 | 84.3 | 0.114 | 0.440 | 79.5 |
| <500 mm | 172836 | 12.9 | 1.3 | 84.4 | 0.017 | 0.085 | 83.7 |
| 500–1000 mm | 142861 | 10.7 | 4.5 | 84.5 | 0.086 | 0.317 | 78.7 |
| >1000 mm | 6449 | 0.5 | 3.8 | 79.2 | 0.012 | 0.039 | 76.9 |
| ***Zambia*** | 742,764 | 84.5 | 27.3 | 86.6 | 2.807 | 19.496 | 87.4 |
| <500 mm | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 500–1000 mm | 220916 | 29.7 | 20.7 | 73.6 | 1.143 | 3.711 | 76.5 |
| >1000 mm | 406507 | 54.7 | 30.9 | 92.5 | 1.665 | 15.784 | 90.5 |
| ***Zimbabwe*** | 396,792 | 91.4 | 8.2 | 95.9 | 0.098 | 1.569 | 94.14 |
| <500 mm | 28,772 | 7.3 | 0.3 | 89.1 | 0.001 | 0.006 | 91.70 |
| 500–1000 mm | 315,979 | 79.6 | 8.4 | 96.2 | 0.084 | 1.287 | 93.88 |
| >1000 mm | 17,747 | 4.5 | 17.4 | 93.6 | 0.013 | 0.276 | 95.39 |

[a] Note that difference in burned area mapping derived from MCD64A1 product, and more refined GFED4s fire mapping incorporating small fires, is18%.

was undertaken in SAGA (http://www.saga-gis.org/en/index.html), yielding an initial un-supervised classification of 10 classes which, upon inspection, were consolidated into five major *VFTs*—of which two classes represented relatively dense, and sparse Dry Savanna, respectively.

*3.3.1.2. Pre-fire transect assessment.* Based on methods modified from Russell-Smith et al. (2009) and Yates et al. (2015), transects (50 × 10 m) were established in groups of three, where each replicate was located in structurally similar vegetation of equivalent fuel age based on Landsat-derived time-since-fire mapping, separated by at least 100 m. Groups of transects were established in as broad a range of time-since-burnt fuel age conditions as practicable given access constraints. In the 2019 EDS sampling period, 21 May - 5 June, pre-fire measurements were undertaken at 39 transects, and at 31 transects in the LDS period, 6–12 September, of the same year.

At each transect, sampling of different fuel components was undertaken as follows. Grass and leaf litter debris (<0.6 mm diameter) fractions were sampled separately from five 1 × 1 m quadrats at 10 m intervals along a central tape, weighed with digital scales, and sub-samples taken for subsequent determination of oven dry weight (ODW). All coarse woody debris (CWD; 6–50 mm diameter) materials were collected from three 5 × 5 m quadrats equally spaced along the transect, weighed, and sub-samples taken for ODW determinations. Using a prepared pro forma, the volume of heavy woody fuels (>50 mm diameter) along a 10 × 50 m belt transect was estimated recording the length, diameter, and hollowness of all materials. In absence of wood density data we express heavy fuel results volumetrically. Diameter at breast height (DBH, 1.3 m) of tree stems (≥5 cm DBH) was measured in the same 10 × 50 m belt transect, for subsequent calculation of stem basal area ($m^2.ha^{-1}$).

Live and dead shrubs/tree juveniles (<50 mm DBH) were counted separately in five 1 × 10 m quadrats along the central tape. The number of individuals per species of live shrubs was recorded in four height classes <50 cm, 50 cm – 1m, 1 – 2m, and >2m—where individuals represent single stems or, most commonly, clusters of stems arising from a common rootstock. Dead shrubs were recorded, but not to species. Although representative individuals of major shrub species were cut and weighed in respective height classes to estimate shrub dry biomass, here we only report shrub densities (individuals $ha^{-1}$) given incomplete representative sampling.

*3.3.1.3. Post-fire transect assessment.* Following pre-fire plot establishment in the EDS period, perimeter prescribed burns (incorporating both fronting fires and backburns) ranging in extent from ~5 to 100 ha were undertaken under late afternoon, relatively benign fire-climate conditions in accordance with the Tsodilo fire management strategy (Magole et al., 2017). Post-fire assessments were undertaken at 21 transects (although two transects in 2-year old fuels would not ignite). In the LDS sampling period, advantage was taken of an extensive wildfire burning in close proximity to Tsodilo under hot, often gusty conditions, mostly in dunefield Dry Savanna, and on occasion in omuramba where sparse fuels were available. Of the 31 transects established in anticipated advance of the fire-front, LDS post-fire assessments were undertaken at 24 which burnt.

Following Russell-Smith et al. (2009), fire severity at respective burned transects was assessed as follows: Low severity—leaf scorch <2 m; Moderate severity—scorch >2 m but < tree canopy top height; High severity—tree canopies scorched. Post-fire patchiness, comprising the proportion of unburnt fuel, vegetation, and bare ground, was assessed in five 1 × 10 m sections along the centre of each transect.

Post-fire measurements of fuel components, including calculation of sample ODW, followed generally the same procedure as for pre-fire assessments. Post-fire remnant litter and grass fractions were recorded in five 1 × 1 m quadrats at 10 m intervals along the transect on the opposite side of the tape from unburnt measures. Consumed CWD was recorded in five 10 × 1 m quadrats on the opposite side of the central tape from pre-fire sampling, where the proportion consumed was estimated by observing the amount of CWD left unburnt, including remnants of partly burnt twigs—for example, where the consumed component leaves a readily observed 'ash trail'. Heavy woody fuels were recorded in the same 10 × 50 m belt transect as used for the pre-fire assessment. Counts of remnant shrub individuals in respective height classes were recorded in the same five 1 × 10 m quadrats as used for pre-fire assessments, additionally estimating the proportional loss of stems and leaves per individual.

*3.3.1.4. Drone-based assessment of fire patchiness.* Using a multi-spectral camera (MicaSense RedEdge) mounted on a UAV (DJI Matrice 100), we flew over plots covering two or three transects per flight and an area of between 1.5 and 3.5 ha. Multiple images were taken and stitched together using MicaSense's provided software library (https://github.com/micasense/imageprocessing) and OpenDroneMap (https://www.opendronemap.org) to produce orthophotos of the plot. This was done before and after the plots were burned, and each pre-fire orthophoto was classified into five categories (bare soil, foliage, grass, woody material or shadow) and post-fire into two categories (burned or unburned) using open-source object-based image analysis (Clewley et al., 2014). Any pixels (5 × 5 cm) which had been classified as both 'vegetation' (i.e. either grass, foliage or woody material) and 'unburned' within the area the fire passed over were considered to be pixels in which vegetation remained after the fire. Patches of pixels designated as remnant vegetation <1 $m^2$ were filtered out in an effort to reduce errors resulting from misalignment between pre- and post-fire images. As such, the percentage of remnant vegetation calculated using this method can be considered a minimum. The total area covered by the UAV maps was 12.2ha in the LDS, and 13.8ha in the EDS.

### 3.3.2. Niassa assessment

In the 2019 assessment period, prescribed fire treatments were undertaken in the context of a large-scale EDS prescribed burning trial in an area that in recent years had been affected by severe annual LDS fires (Ribeiro et al., 2017). The aim was to create an effective barrier in order to prevent the entrance of fires from adjoining areas. In the LDS, we targeted patches in the same area that were confined by EDS burn scars or physical barriers (e.g. roads and rock outcrops). While our assessments covered an area roughly 50 × 100 m, some fires were much larger.

In the EDS sampling period, 21 June - 7 July, we report assessments of seven prescribed fires of which four were in dambo grasslands and three in Woodland Savanna. In the LDS period, 6–19 October, 20 prescribed fires were measured of which 14 were in Woodland Savanna and six in dambo grasslands. Although sampling effort was stratified by year-since-fire with reference to the Collection 6 MODIS 500m burned area product, we do not report those data given the coarse mapping resolution; all transects were mapped as being either one or two years unburnt.

Pre- and post-fire fuel measurements were undertaken in two parallel transects (roughly 50 m apart), following the methodology as described in *3.3.1.2 and 3.3.1.3*. In the laboratory, fuel samples were oven-dried for 48 h at 70 °C, weighed again to derive the moisture content, and ground to a coarse powder (Cyclotec 1093, Foss A/S). Subsequently, at the Vrije Universiteit Amsterdam, we pulverized the powder in a second milling phase using a high energy vibrational mill (MM 400, Retsch). After drying the sample again for 24 h, ± 4 mg of powder was analysed for nitrogen and carbon content (Flash EA 1112 series, Thermo electron corporation). Weighted average *VFT* carbon contents represent the average of the different fine fuel classes, multiplied by their respective contributions to the consumed fuel mixture.

### 3.3.3. Drone-based assessment of emission factors at Tsodilo and Niassa

To determine the seasonal and *VFT* dependence of *EF(g)*s at both Tsodilo and Niassa, we measured smoke from fires in different vegetation fuel types. At Tsodilo, sampling of prescribed EDS fires and LDS wildfires was undertaken at the same plots as for the drone-based assessment of fire patchiness. At Niassa, sampling was undertaken at prescribed EDS and LDS fires typically lit around 2 p.m. Smoke samples were collected from the mixture of flaming and smouldering phases, at an altitude of 15 m. While we predominantly sampled headfires, we found no statistical difference between *EF(g)* from headfires or backfires. Sampling continued until smouldering had ceased, with the exception of point sources like logs, trees and dung.

We sampled fresh smoke in 1L Tedlar bags using an unmanned aerial system (UAS) following the methodology described in Vernooij et al. (2020). Within 12 h of sampling, samples which were transported under dark conditions were measured for $CO_2$ and $CH_4$ (Micro portable, Los Gatos research) as well as CO and $N_2O$ (Pico analyser, Aeris technologies). In respective samplings, 1 bag of standard gas (concentrations listed in Vernooij et al., 2020) was included to allow for analyser drift correction. Prior to the fire, four background samples were collected at 15-m altitude, which were subtracted from the smoke samples to obtain the excess mixing ratio (EMR). EFs were calculated using the carbon mass balance method, following Yokelson et al. (1999):

$$EFg = \mathrm{Fc} \times \mathrm{MW(g)AMc} \times \mathrm{C\ gCtotal} \times 1000\ \mathrm{g\ kg}\ -1 \quad \text{(Eq 2)}$$

where, *EFg* is the emission factor of species g and Fc is the fractional carbon content by weight of the vegetation mixture. This value is the average of the carbon content of the fuel subclasses, weighted by their respective contribution to the total pyrolyzed carbon.

The respective weighted mean carbon contents for Woodland Savanna and dambo grassland vegetation, were 45% and 43%. Since no carbon content measurements were available for the Tsodilo plots, we assumed a carbon content of 45%. $MM_i/AM_c$MMi/AMcMMi/AMc *MW*(*g*) is the molecular weight of species *g* which is divided by the atomic mass of carbon *AMc*. C (g)is the moles of carbon emitted in species g. $C_{total}$ is the total moles of emitted carbon. Carbon emitted as non-methane hydrocarbons (NMHC) and particulate matter ($PM_{2.5}$) was estimated based on literature values. The carbon in NMHC was assumed to represent 3.5 times the carbon in $CH_4$ and the total particulate fraction was estimated to be 7% of *EF(CO)* (Andreae, 2019) with carbon accounting for 68% of the $PM_{2.5}$-mass (Reid et al., 2015). Using these values, $PM_{2.5}$ and NMHC contribute 0.4–1.3% and 0.4–2.9% to the total carbon, respectively, depending on the *EF(CO)* and *EF($CH_4$)* of the sample.

Weighted averages for individual fires or seasonal *VFT*-averages were calculated following Eq. (2), based on the cumulative EMR of the combined samples in the class. Particularly for $N_2O$, weighting by EMR reduces the noise caused by measurement errors which become significant in bags with very low or even negative EMR (Vernooij et al., 2020).

## 3.4. Results

### 3.4.1. Tsodilo transect-based assessments

*Seasonal fire frequency*—based on interpretation of fire mapping derived from Landsat imagery (30 m pixels; Fig. 2a), and the automated MODIS (500 m pixels) product, 2014–19, nearly all fire extent occurred in the LDS period (post-June), especially in Dry Savanna and Woodland

Savanna fuel types (Fig. 2b; Table 3).

*Pre-fire measures*—Dry Savanna occupying Kalahari dunefield substrates comprised a sparse tree cover (Basal area = 1.3 [range: 0–9.2] $m^2.ha^{-1}$) often overtopping a relatively dense multi-stemmed shrub layer typically <2 m height, and sparse heavy woody fuel remnants (Fig. 3). At time of EDS sampling, many shrubs and trees had already begun to shed their leaves. This is reflected also in the observed tendency for seasonal fuel loads, dominated by leaf litter components, to be markedly higher in the LDS than EDS with increasing time since fire (Fig. 4). Significant differences between EDS and LDS fuel components (t-tests with unequal variance) were observed only for two-year old litter ($p = 0004$), total fine fuels (litter + grass; $p = 0.0019$), and CWD ($p = 0.049$).

*Post-fire measures*—All 19 prescribed EDS fires were assessed as being of Low severity whereas, of 24 LDS measurements, 14 were of High severity, two of Moderate severity, and eight of Low severity. Under prescribed EDS conditions we had great difficulty with getting fires to carry, especially in plots with 1–2 years of fuels. Resultant unburnt area (*P*) was significantly different ($p = 0.003$) between fires of Low (mean = 50%) and High (mean = 24%) severity (Table 4). There were also significant differences in consumption (*CC*) of different fuel components under fires of Low vs High severity: Fine fuels—Low 65% vs High 89% ($p < 0.0001$); Coarse woody debris—Low 17% vs High 35% ($p = 0.005$); Heavy woody fuels—Low 6% vs High 30% ($p < 0.05$) (Table 4).

### 3.4.2. *Tsodilo post-fire drone-based patchiness*

In drone classification images, EDS fires also resulted in more vegetation-classed pixels remaining unburned than in LDS fires. Adjusting for relative abundance of fuel in each plot, the percentage of surface area covered by unburned vegetation pixels was 14% in Low

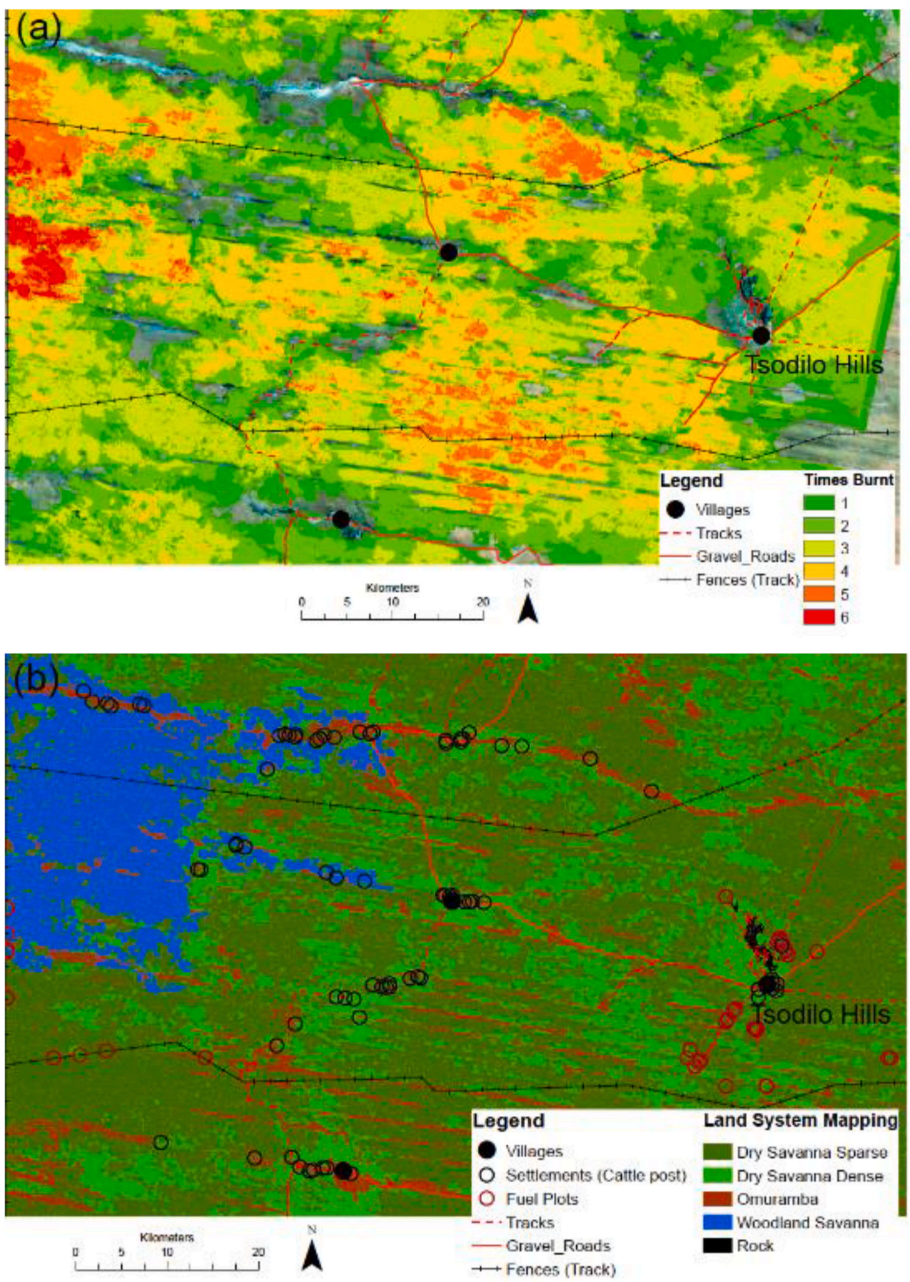


**Fig. 2.** Tsodilo Hills Enclave study site, Ngamiland, Botswana, illustrating sampling plot locations, 'cattle posts', access roads and tracks: (a) Landsat-derived fire frequency, 2014-19—unburnt areas given in white/grey; (b) Vegetation Fuel Types. Unpublished cattle post data provided by Arthur Albertson, Kalahari Wildlands Trust.

**Table 3**
Mean area burnt 2014–19 of major landscape units, Tsodilo, comparing fire mapping derived from manual Landsat (30 m pixels) with automated MODIS (500 m pixels) imagery.

| Landscape unit | Area ($km^2$) | Landsat mapping | | | MODIS mapping | | |
|---|---|---|---|---|---|---|---|
| | | Mean burnt area (%) | | Annual range (%) | Mean burnt area (%) | | Annual range (%) |
| | | EDS | LDS | | EDS | LDS | |
| Dry Savanna | 4766 | 0.1 | 34.5 | 20–56 | 0.1 | 39.7 | 21–59 |
| Woodland Savanna | 564 | 0 | 47.3 | 21–68 | <0.1 | 45 | 19–66 |
| Omuramba | 386 | 0.1 | 18.5 | 6–36 | 0.1 | 26.3 | 11–43 |
| Rock outcrop | 9 | 0 | 1.5 | 0–4 | 0 | 10.1 | 0–36 |

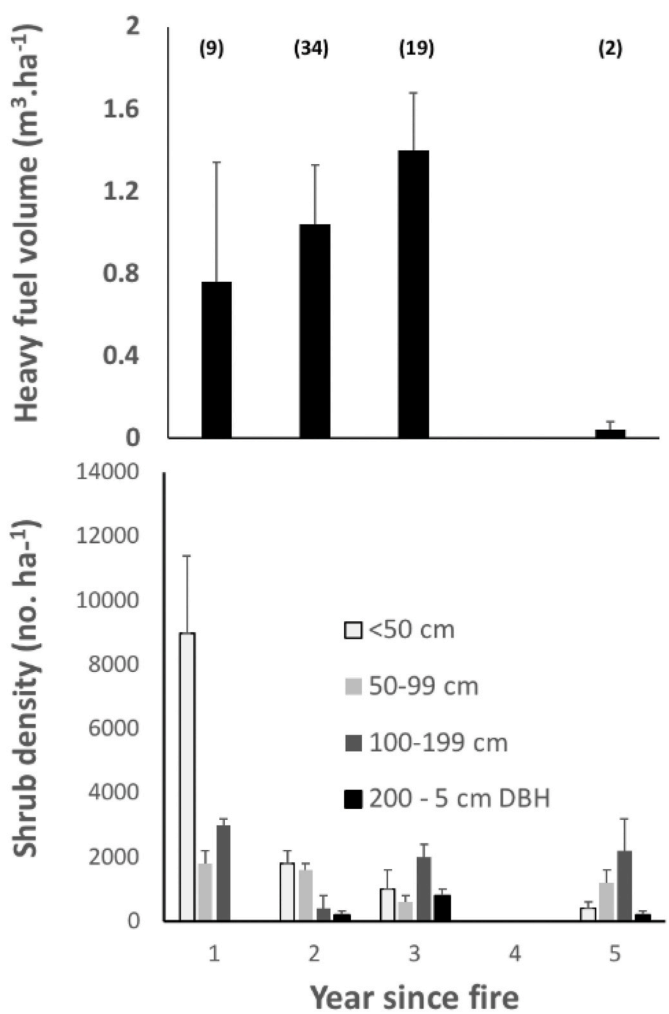


**Fig. 3.** Pre-fire assessment of heavy woody fuel (upper panel) and shrubby fuels (lower panel) components sampled at transects either in EDS or LDS. Note no 4 years since fire sampled. Number of transects sampled (*n*) given in parentheses; error bars given as Standard Error of the Mean (S.E.M.).

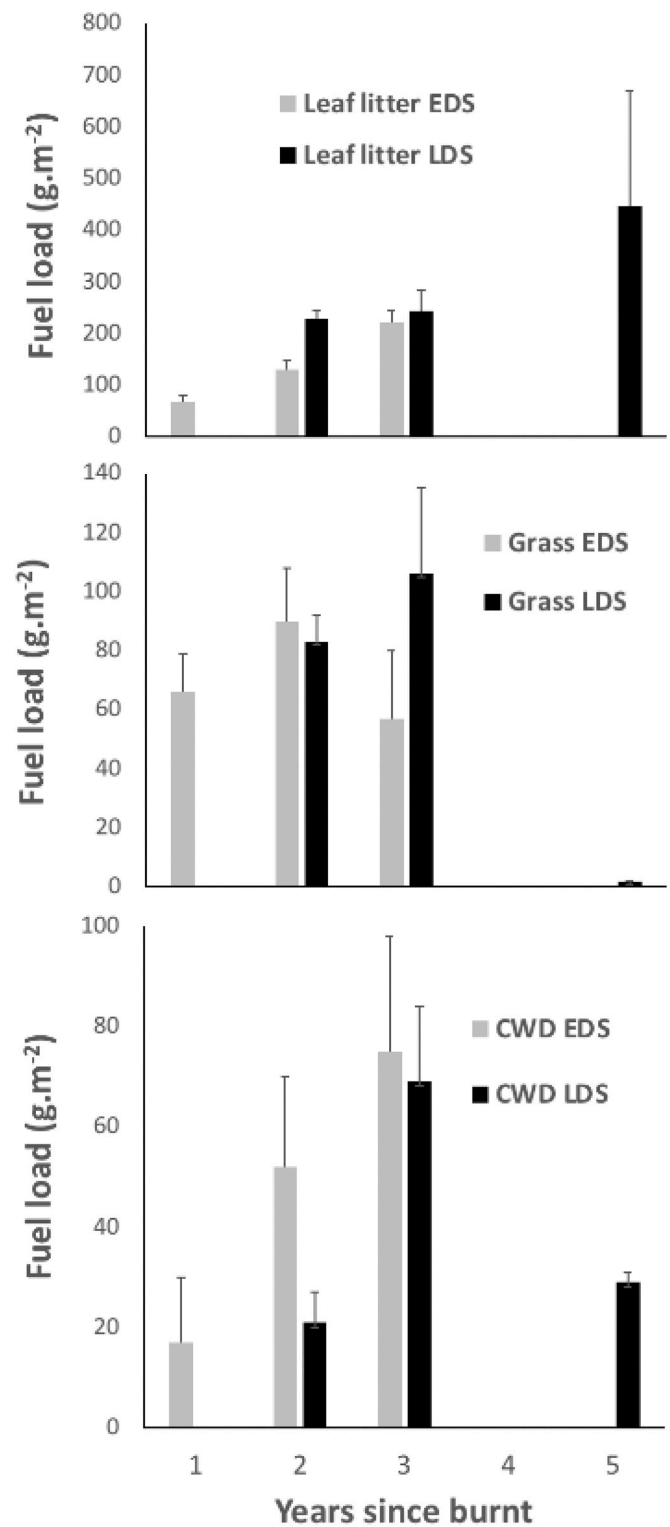


**Fig. 4.** Pre-fire seasonal (EDS, LDS) assessment of leaf litter, grass, coarse woody debris (CWD) fuel components. Note different scales on y-axes and no 4 years since fire sampled; error bars given as S.E.M.

severity EDS fires and 9% in the more severe LDS fires, weighted by image area (Table 5). Deliberately burning into the wind also gave noticeably different *P* outcomes, where the difference between less severe back-burn and more severe fronting fires was obvious. In some cases, this contributed up to a 19% difference in drone-calculated patchiness within a few metres either side of the back-burn line (Fig. 5).

### 3.4.3. *Niassa transect-based assessments*

*Pre-fire measures*—Sampled Woodland Savanna transects comprised a sparse tree cover (Basal area = 1.4 [range: 0.4–3.7] $m^2.ha^{-1}$), relatively dense shrub layer and sparse heavy woody fuels similar to that of Dry Savanna at Tsodilo (Fig. 3; Table 6). Despite obvious limitations of time-for-space sampling, it would appear that after 1–2 years following

**Table 4**
Post-fire assessment of unburnt fire patchiness, and fuel consumption, under different fire severity conditions, where: Low severity = leaf scorch <2m height; Mod severity = leaf scorch >2 m but < canopy height; High severity = leaf scorch of full canopy. Note: all 19 prescribed EDS fires were of Low severity whereas, of 24 LDS measurements, 14 were of High severity, two of Moderate severity, and eight of Low severity. Statistical tests (two-tailed t-tests with unequal variance) comparing mean effects between fires of Low and High Severity only).

| Criterion | Fire severity class | | | *p* |
|---|---|---|---|---|
| | *Low* | *Mod* | *High* | (Low v High) |
| Patchiness | | | | |
| % unburnt | 50 | 25 | 24 | 0.003 |
| *N* | 27 | 2 | 12 | |
| S.E.M. | 5 | 9 | 6 | |
| **Percent fuel consumption** | | | | |
| *Fine fuels (<6 mm diameter)* | | | | |
| % burnt | 65 | 73 | 89 | <0.0001 |
| *N* | 27 | 2 | 14 | |
| S.E.M. | 2 | 8 | 2 | |
| *Coarse fuels (6 – 500 mm diameter)* | | | | |
| % burnt | 17 | 11 | 35 | 0.005 |
| *N* | 27 | 2 | 12 | |
| S.E.M. | 3 | 3 | 5 | |
| *Heavy fuels (>500 mm diameter)* | | | | |
| % burnt | 6 | 7 | 30 | 0.049 |
| *n* | 20 | 2 | 11 | |
| S.E.M. | 3 | 7 | 10 | |

fire in Woodland Savanna at Niassa, accumulation of CWD was equivalent, and grass and litter fuels up to double that, of Dry Savanna at Tsodilo (Fig. 4; Table 6). Conversely, dambo fuels were characterised almost exclusively by a substantial grass component (Table 6).

*Post-fire measures*—Based on scorch height assessments, prescribed fires at 18 of 20 Woodland Savanna transects were of Low severity, with two transects experiencing partly Moderate - Low severity EDS fires. All seven dambo transects were burnt continuously throughout by prescribed fires (Table 6). Resultant consumption rates of total fine fuels, CWD, and heavy woody fuels in Woodland Savanna transects were consistent with equivalent results under Low – Moderate severe fires at Tsodilo (Tables 4 and 6). In dambo, grass fuel consumption was markedly greater in LDS treatments (Table 6).

**Table 5**
Post-fire unburnt patchiness in individual drone flights. An unburned vegetation pixel is defined as a pixel which in the pre-fire flight was classified as either grass, foliage or woody material, and in the post-fire flight as 'unburned'.

| Flight ID (Transect numbers) | Unburned vegetation pixels (%) | FL-adjusted unburned vegetation pixels (%) | Season | Fire severity[a] | Area (ha) |
|---|---|---|---|---|---|
| 5_6 | 22.2 | 22.2 | EDS | Low | 2.1 |
| 7_8 | 15.0 | 13.9 | EDS | Low | 1.6 |
| 9_10 | 14.1 | 11.7 | EDS | Low | 2.2 |
| 23_24 | 17.5 | 11.3 | EDS | Low | 1.7 |
| 34_35 | 22.4 | 17.6 | EDS | Low | 2.5 |
| 37_39 | 12.4 | 11.0 | EDS | Low | 3.7 |
| 63_64 | 31.9 | 17.0 | LDS | Low – Mod[b] | 3.2 |
| 66_67 | 6.2 | 4.1 | LDS | Mod – High[b] | 4.0 |
| 74_76 | 10.1 | 8.3 | LDS | High | 5.0 |

[a] Fire severity: Low = scorch height <2 m; Mod = scorch height >2 m but < tree canopy height; High severity = tree canopy scorched.
[b] Transect 63 was burnt at Low severity, Transects 64 and 66 were burnt at Moderate Severity, and Transect 67 was burnt at High severity.

### 3.4.4. Tsodilo and Niassa post-fire emission factors

Based on studies both at Tsodilo and Niassa, *EF(g)*s for $CH_4$ were stable or even slightly elevated in the LDS for the Dry Savanna and Woodland Savanna, whereas we found a large seasonal decline of *EF* $(CH_4)$ in dambo. The EMR-weighted mean $EF(CH_4)$ and Modified Combustion Efficiency (MCE) for EDS and LDS fires sampled in this study are given in Table 7, where MCE represents the completeness of the oxidation process and is defined as the EMR of $CO_2$ divided by the combined EMRs of $CO_2$ and CO. $EF(CH_4)$ was 6.12 g $kg^{-1}$ vs 1.45 g $kg^{-1}$ for dambo, 1.51 vs 2.22 g $kg^{-1}$ for Woodland Savanna, and 1.31 g $kg^{-1}$ vs 1.34 g $kg^{-1}$ for Dry Savanna, in the EDS vs LDS respectively (Fig. 6a). While we found a positive correlation of $EF(CH_4)$ with Fuel Moisture Content (FMC; Pearson R = 0.84, $p = 0.017$) in dambo, the opposite was observed for woodland fires (Pearson R = −0.64, $p = 0.024$). With an average MCE of 0.97, and $EF(CH_4)$ of 0.60 g $kg^{-1}$, combustion in open grassland at Tsodilo ('Kalahari grassland') was especially more efficient compared with other woody savanna types (Table 7).

The EMR-weighted averaged $EF(N_2O)$ were stable at 0.12 vs 0.11 g $kg^{-1}$ for dambo, 0.10 vs 0.12 g $kg^{-1}$ for Woodland Savanna, and 0.19 vs 0.17 g $kg^{-1}$ for Dry Savanna, in the EDS vs LDS respectively. During the LDS, $EF(N_2O)$ was more variable with standard deviations exceeding absolute *EF(g)* values (Fig. 6b). With the exception of the seasonal *EF* $(N_2O)$ differences in Dry savannas and Dambo grasslands, all seasonal differences (though small) were significant (two tailed t-tests with unequal variance: $p < 0.05$). In dambo, the $EF(CH_4)$ were much lower and MCE much higher in the LDS whereas we found a small opposite trend in Woodland Savanna.

## 4. Discussion

The contribution of annual savanna fires in southern African biomes to global emissions budgets is well recognised (van der Werf et al.,

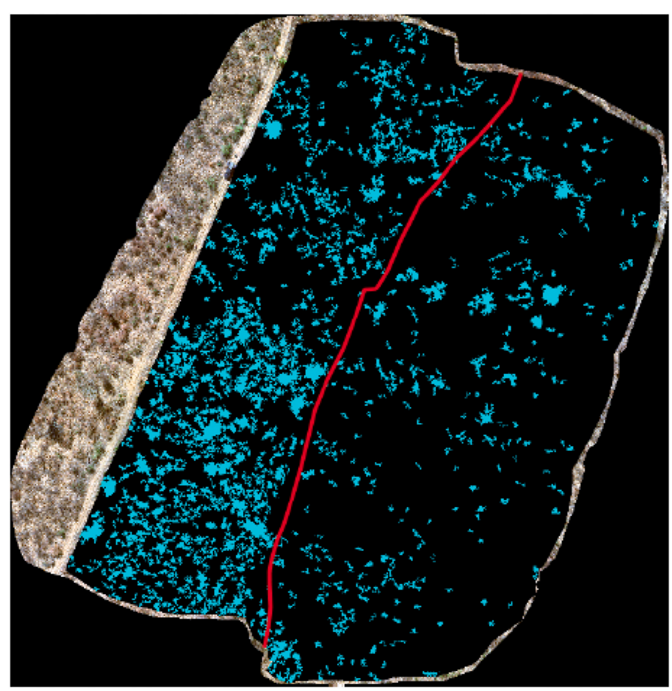

**Fig. 5.** Drone-based patchiness map for an example plot at Tsodilo contrasting greater patchiness in back-burning (to left of red line) vs frontal fire, on right. Blue patches indicate pixels classed as both 'vegetation' and 'unburned' with an area greater than 1 $m^2$.

**Table 6**
Pre- and post-fire assessment characteristics at 27 Niassa study transects. Note all Woodland Savanna prescribed fires were of Low Severity (leaf scorch <2m height) whereas severity was not determined for dambo prescribed fires. Error given as S.E.M.; DBH = diameter at breast height, 1.3 m

| Parameter | Woodland Savanna | | Dambo | |
|---|---|---|---|---|
| **(a) Pre-fire** | ($n$ = 17) | | ($n$ = 10) | |
| ***Shrub density*** (no. $ha^{-1}$) | | | | |
| <50 cm tall | 2506 (±383) | | 40 (±52) | |
| 50 cm - <5 cm DBH | 2259 (±278) | | 80 (±57) | |
| ***Heavy fuel volume*** ($m^3$. $ha^{-1}$) | 0.90 (±0.19) | | 0.01 (0.01±) | |
| | ***EDS*** ($n$ = 3) | ***LDS*** ($n$ = 14) | ***EDS*** ($n$ = 4) | ***LDS*** ($n$ = 6) |
| ***Leaf litter*** (g. $m^{-2}$) | 280 (±69.9) | 300 (±20.8) | 16.8 (±13.9) | 1.7 (±0.7) |
| ***Grass*** (g. $m^{-2}$) | 221.9 (±69.5) | 154.3 (±12.0) | 371.5 (±38.6) | 459.6 (±56.3) |
| ***Coarse woody debris*** (g. $m^{-2}$) | 53.4 (±4) | 47.4 (±7.8) | 6.9 (±6.9) | 0 |
| **(b) Post-fire** | ***EDS*** ($n$ = 3) | ***LDS*** ($n$ = 14) | ***EDS*** ($n$ = 4) | ***LDS*** ($n$ = 6) |
| ***Patchiness*** (% burnt) | 100 | 91.3 (±2.1) | 100 | 100 |
| ***Fuel consumption*** (%) | | | | |
| *Total fine fuels* (<6 mm diameter) | 60.3 (±11.5) | 70.5 (±2.1) | 69.8 (±6.5) | 90.8 (±1.0) |
| *Coarse fuels* (6–500 mm diameter) | 1.7 (±0.9) | 18.5 (±2.7) | 0 | 1.0 (±0.7) |
| *Heavy fuels* (>500 mm diameter) | 2.0 (±1.4) | 6.5 (±1.5) | 0 | 1.3 (±1.3) |

**Table 7**
Excess Mixing Ratio (EMR)-weighted mean *EF(g)*s and Modified Combustion Efficiency (MCE) results from assessments at Tsodilo and Niassa. Error given as Standard Deviation. The EMR is the concentration measured in the sample minus the background air, sampled prior to the fire. The Modified Combustion Efficiency (MCE) is defined as the EMR of $CO_2$ divided by the combined EMRs of $CO_2$ and CO, as applied commonly as a proxy for the completeness of the oxidation process.

| Vegetation | Season | MCE | *EF($CH_4$)* (g $kg^{-1}$) | *EF($N_2O$)* (g $kg^{-1}$) |
|---|---|---|---|---|
| Kalahari grassland (Tsodilo) | EDS | **0.97** (±0.12) | **0.60** (±0.40) | **0.11** (±0.13) |
| Dry Savanna (Tsodilo) | EDS | **0.93** (±0.15) | **1.34** (±1.11) | **0.19** (±0.15) |
| | LDS | **0.94** (±0.24) | **1.31** (±0.88) | **0.17** (±0.24) |
| Woodland Savanna (Niassa) | EDS | **0.93** (±0.17) | **1.51** (±1.03) | **0.10** (±0.16) |
| | LDS | **0.92** (±0.18) | **2.22** (±1.21) | **0.12** (±0.18) |
| Dambo grassland (Niassa) | EDS | **0.87** (±0.05) | **6.12** (±2.22) | **0.12** (±0.05) |
| | LDS | **0.95** (±0.23) | **1.45** (±0.71) | **0.11** (±0.24) |

2017). Perhaps less well appreciated is that the vast majority of extensive fires and associated emissions occur in the latter part of dry season (Table 1; Appendix S1). As noted by Roteta et al. (2019), both regional fire extent and emissions are likely to be significantly underestimated given that the currently available burnt area mapping archive from 2001 is derived from MODIS imagery at 500 m pixel resolution. Comparing burnt area mapping derived from MODIS with Sentinel 2 imagery (20 m pixels) for sub-Saharan Africa in 2016, those authors observed that whereas burnt area mapping derived from both sensors yielded equivalent monthly results for fires >250 m, the coarser resolution MODIS product detected substantially less smaller-sized fires. Although many small detected fires presumably reflect burning for agricultural purposes (e.g. field preparation, crop residues), the present study is concerned principally with addressing solutions for mitigating emissions from extensive LDS wildfires.

For illustrative purposes we have defined the LDS period for the entire study region as commencing July 1 as determined by the mid-month of the annual driest six months (Fig. 1c and d), but acknowledge that considerable regional variability exists both spatially (e.g. Fig. 1c) and temporally (Frost, 1996; Archibald et al., 2009, 2010). Similar fire seasonality considerations apply in other southern hemisphere savanna contexts, with the end of July often being considered a useful demarcation both in Brazilian cerrado (Pivello, 2011; Fidelis et al., 2018) and north Australian eucalypt-dominated savanna (Williams et al., 2002; Maier and Russell-Smith, 2012).

At the Tsodilo study site the impacts of the contemporary LDS-dominated fire regime on former woody cover were often stark as illustrated in (Fig. 7a and b), but typically not so evident in heavily grazed, hence relatively fire-protected areas, including omuramba (Fig. 7c), and some areas of structurally well-developed Woodland Savanna (Fig. 7d). Such impacts may be anticipated more generally in southern African savannas under frequent, relatively severe LDS fire regimes, given globally observed relationships between severe fires, tree stem death, and associated development of multi-stemmed shrubby understoreys (e.g. Frost and Robertson, 1987; Bond and van Wilgen, 1996; Hoffman and Solbrig, 2003; Prior et al., 2010). However, substantial local spatial variability can be expected given historical and recent interactions with densities of both domestic and native grazing and browsing fauna, and local fire management practices (van Langevelde et al., 2003; Staver et al., 2009; Archibald, 2016). At Tsodilo, for example, although not formally investigated in this study, it was apparent that various drivers have contributed to significant landscape-scale variation in localised tree and complementary shrub density dynamics, including: ongoing establishment of 'cattle-post' settlements in omuramba over recent decades; spatially differentiated impacts of domestic stock (cattle, goats, donkeys) densities and accompanying regional losses of native grazers and browsers; regulated suppression of landscape-scale fire management practiced traditionally by Ju/oasi San (or 'Bushman') and Hambushuku Bantuan (cattle herding) peoples.

By contrast with these LDS fire impacts, much contemporary southern African literature focuses on woody thickening and encroachment issues, especially in South Africa (e.g. Acocks, 1953; Trollope, 1974; O'Connor et al., 2014; Ward et al., 2014; Stevens et al., 2017), but also from heavily grazed Botswana sites (van Vegten, 1984; Moleele et al., 2002). Acknowledging that the drivers of woody thickening are often complex, including the exacerbating effects of $CO_2$ fertilisation (O'Connell et al., 2014; Ward et al., 2014), it is perhaps telling that, of all southern African countries assessed here, annual large fire occurrence is exceptionally restricted in South African savannas (Table 2). Where sufficient ground fuels are available, repeated application of high-intensity fires has been demonstrated to effectively reverse woody encroachment in South African savannas (Smit et al., 2016).

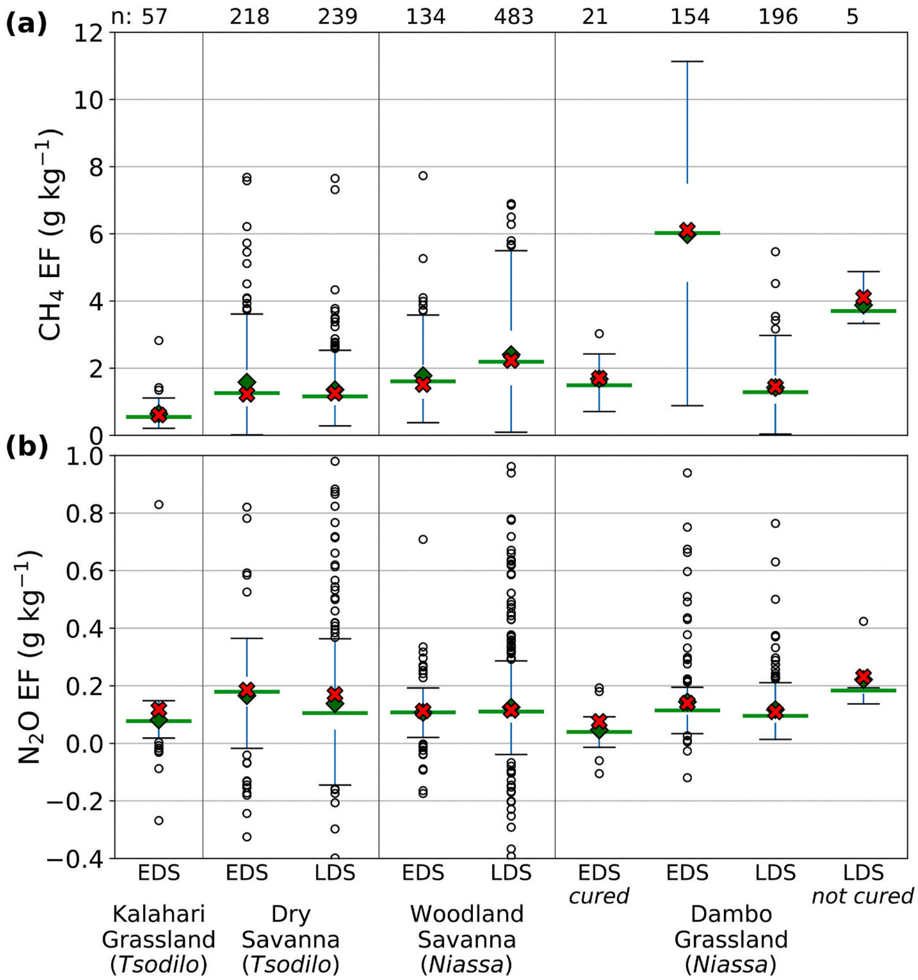


**Fig. 6.** Variability of Emission Factors of $CH_4$ (a) and $N_2O$ (b), measured from fires in the studied savanna vegetation types at Tsodilo and Niassa in the early dry season (EDS) and the late dry season (LDS), respectively. The number of samples (*n*) in each class is given at the top of each boxplot column. The green diamond represents the arithmetic mean and the red cross represents the Excess Mixing Ratio weighted mean. Measurements more than 1.5 times the interquartile range (IQR) above the upper, or below the lower quartile, are presented as outliers (open circles). Whiskers represent the outermost values within 1.5 times the IQR of the respective quartiles.

### 4.1. Savanna emissions abatement technical feasibility

Our assessment demonstrates that the application of a savanna burning GHG emissions abatement method in the Tsodilo region focused on the undertaking of prescribed EDS burning in strategic locations to reduce LDS wildfire extent and resultant emissions is technically feasible, notably with respect of: extensive fire-prone savanna vegetation cover; low human population density, albeit concentrated in scattered sites; LDS fine fuels tending to be markedly greater than EDS fuels given seasonal leaf litter inputs; LDS wildfires tending to be significantly more intense and combusting more fuels; methane and nitrous oxide emission factors being essentially equivalent in EDS and LDS periods under cured fuel conditions.

Although it is not our purpose here to quantify the net GHG emissions abatement generally achievable through the implementation of prescribed EDS fire management—especially since this requires the development of a methodological approach (e.g. CoA , 2015, 2018) which integrates unique spatial (e.g. distribution of vegetation fuel types) and dependent temporal (e.g. fuel type accumulation) parameters for defined project areas—fuel combustion and related data presented here for respective Tsodilo Hills and Niassa project sites indicates that EDS fire treatments reduced accountable GHG emissions relative to LDS treatments by 2.58 times, and 1.8 times, respectively. Under Australian conditions, Russell-Smith et al. (2009) showed that, for a 24,000 km$^2$ site supporting a diverse variety of savanna vegetation fuel types and associated fire histories, net emissions from LDS fires typically were 2.08 times greater per unit area than from EDS fires.

Additionally, while not currently accounted for in savanna emissions inventories given substantial measurement uncertainties, greater fuel consumption under more intense and currently much more regionally extensive LDS conditions results in very significant emissions of aerosol black carbon (ACB) with relatively short-term (days to weeks) atmospheric warming and air quality effects (Bond et al., 2013; Chiloane et al., 2017), and long-term (centuries to millennia) stocks of relatively inert soil pyrogenic carbon (Jones et al., 2019). Implementing a prescribed fire management program which substantially reduces the extent of LDS wildfires affords obvious major ancillary benefits for mitigating ACB emissions and ecologically unsustainable conversion of living biomass to charcoal.

Although grassy fuels are considered the archetypic fuel-type of savannas (Scholes and Archer, 1997), and that it is widely observed that grass production is inversely related to woody cover in more mesic savannas (Cook, 2003; Scholes, 2003; Dohn et al., 2013; Donohue et al., 2013), fine fuel loads were dominated by leaf litter both in Dry Savanna at Tsodilo and Woodland Savanna at Niassa, likely reflecting substantial leaf litter inputs from shrubs especially. At Tsodilo, leaf litter inputs exhibited strong seasonality peaking in the LDS associated with progressive woody plant leaf fall, as has been described generally for the extensive southern African Miombo woodland savannas dominated by seasonally leafless woody taxa (Malaisse et al., 1975; Frost, 1996; Ribeiro et al., 2013). By contrast, grassy fuel load components, once cured, may be anticipated to be aseasonal except where extensively grazed or consumed by termites (Scholes et al., 1996; Hély et al., 2003). CWD inputs are most likely associated with stochastic events such as severe wet season storms, dry season fires (Yates et al., 2020) and, in African contexts, destructive impacts of large animals such as elephants


**Fig. 7.** Landscapes at respective study sites: (a) low severity burn in Kalahari dunefield Dry Savanna, Tsodilo—noting extent of unburnt sandy patches, and fallen and standing woody stem remnants of former more dense woody vegetation; (b) emissions plume associated with LDS wildfire, Tsodilo; (c) heavily grazed *omuramba* associated with relatively fertile dune swale, Tsodilo; (d) Woodland Savanna unburnt for two preceding years, dominated by *Burkea africana*, Tsodilo—photo taken in EDS before significant leaf drop; (e) moist smoky EDS fire in dambo vegetation, Niassa Reserve; (f) LDS fire in dambo vegetation, Niassa Reserve; (g) Woodland Savanna vegetation, Niassa Reserve. Photos: Jeremy Russell-Smith (a–d); Roland Vernooij (e–g).

(Mosugele et al., 2002; Ribeiro et al., 2008).

As expected, fuel combustion was observed to be strongly related to fire severity, and thereby seasonal conditions, as widely reported in the savanna literature (e.g. Frost and Robertson, 1987; Shea et al., 1996; Russell-Smith et al., 2009). Likewise, fire patchiness, describing the post-fire spatial extent of unburnt fuel, was observed to be greater under prescribed low fire severity, EDS conditions, as observed widely in Australian savanna studies (Oliveira et al., 2015). Although greater unburnt patchiness was observed in metre-scale transect-based studies *vs*. plot-scale drone-based assessments at Tsodilo, especially from EDS fires, this is likely attributable in part to the filtering out of unburnt patches <1 $m^2$ in the drone-based assessment process. Drone-based assessments of patchiness, and above-ground biomass (e.g. Cunliffe et al., 2020; Eames et al., 2021), show great promise in providing more cost-effective and spatially comprehensive pre- and post-fire parameter assessments than have been available hitherto.

The limited seasonal fluctuation of *EF*($CH_4$) and *EF*($N_2O$) observed in Botswanan Dry Savanna and Mozambican Woodland Savanna is consistent with previous studies from northern Australian savanna (Meyer et al., 2012), Zambian Miombo Woodland (Hoffa et al., 1999), and Brazilian cerrado (Vernooij et al., 2020), which found either no significant or limited seasonality differences in *EF*($CH_4$) and *EF*($N_2O$) under cured fuel conditions. The larger EF-spread, as well as the slight increase of *EF*($CH_4$) in LDS Woodland Savanna, is consistent with findings from Brazilian cerrado (Vernooij et al., 2020). This may be related to a larger contribution of reduced smouldering combustion-prone fuels (e.g. leaf litter) in the LDS, combined with more efficient combustion of litter and grass. Although *EF(CH4)* was strongly influenced by fires under different seasonal moisture conditions in dambo (Fig. 6a— as illustrated in Fig. 7e and f), the opposite was observed in Woodland Savanna exhibiting marked seasonal leaf fall (Fig. 6a—as illustrated in Fig. 7g) and typically greater heavy woody and CWD fuel consumption under LDS conditions (Table 6).

We found a large seasonal difference of *EF*($CH_4$) in dambo that was similar to, and in some instances exceeded, effects observed by Hoffa et al. (1999) and Korontzi et al. (2003). Dambos are seasonally flooded grasslands and therefore strongly geomorphologically bound (Bullock, 1992). The large variability in grass moisture content is driven by local water table depth and soil type rather than precipitation. Hence, we found fully cured dambo in the latter EDS and uncured dambos in the LDS, in which both the Modified Combustion Efficiency (MCE) and *EF* ($CH_4$) did not reflect the overall seasonal trend (Fig. 6a). At the time of experimental EDS fires (May–June 2019), most dambo grasses sampled in this study were still much more moist compared to woodland fuels with an average dry weight FMC of 54 ± 17% *vs* 17 ± 7% for woodlands. The absolute FMC-difference compared to LDS fires (October 2019) is therefore much larger, where the dry weight FMC range of dambo grasses declined to 21 ± 11% whereas the FMC of assessed Woodland vegetation declined to 7 ± 4%. Their geomorphologically confined nature and extreme seasonality make them ill-suited as a reference for grassland savanna, since typical savanna grasslands on freely draining substrates will cure earlier in the dry season. Fires in uncured dambo only consumed part of the biomass, stimulating a burst of post-burn regrowth. Prescribed fires under these conditions create inadequate fire breaks, allowing for passage of a second fire in the LDS.

The seasonal pattern of emission characteristics found in dambo indicates the need to individually assess specific VFTs for a tailored fire management approach.

As demonstrated in this study, and also by Vernooij et al. (2020), drone-based sampling can provide large numbers of high-accuracy measurements. Such application can potentially target different parts of the fresh smoke plume separately to provide unique insights into the intricacies of different phases of the combustion process, and understanding variability in EFs.

### 4.2. Implementation challenges

Despite this positive technical feasibility assessment, considerable implementation challenges are involved with the development of emissions abatement savanna fire management projects in southern Africa including, but not limited to, legal and policy issues, equity and rights concerns, governance arrangements, building capacity, research and evidentiary data needs, market-based instruments. These matters have been canvassed previously by Russell-Smith et al. (2013b) and are updated and summarised in Appendix S2. Here we address a few outstanding issues raised above.

In their assessment focusing on savanna burning emissions abatement opportunities especially in Africa, Lipsett-Moore et al. (2018) concluded that such projects could most feasibly be undertaken in designated protected areas, including in four fire-prone countries considered here (Angola, Mozambique, Zambia, Zimbabwe) with extensive regions receiving >600 mm MAR. Although the Australian methodology also applies to regions receiving >600 mm MAR, there is no *a priori* reason that fire-prone regions receiving less than this quantum should be excluded, for example in Namibia and Botswana. Assessment of the climatic-fire envelope underpinning establishment of the current Australian methodology found that effective savanna burning projects might feasibly extend to 500 mm MAR in situations supporting frequent wildfires (Whitehead et al., 2014). As demonstrated here for the Tsodilo Hills Enclave, and previously for Chobe (Botswana) and Bwabwata (Namibia) National Parks (Pricope and Binford, 2012; Russell-Smith et al., 2013b), significant opportunity exists for the undertaking of savanna burning projects under relatively low rainfall and high fire frequency conditions. In many lower rainfall situations however, fire frequency is often too low and variable (van Wilgen et al., 2007; Trollope, 2014) to support economically viable emissions reduction projects (Whitehead et al., 2014).

The Tsodilo example illustrates the potential for implementation of savanna burning projects beyond protected area boundaries particularly in sparsely populated regions (Fig. 2a,b), and a variety of land use, community governance, and regulatory constraints. With dependence on rain-fed crops, pastoralism and natural product harvesting on customary land tenure, the Tsodillo Hills community represents the vast majority of southern African savanna livelihoods (UN-Habitat, 2005). In more remote settings, despite varying levels of regulatory deterrents, traditional burning is frequently used to support contemporary rural livelihoods, for example: improving grazing and controlling pests; harvesting of building material, medicine and food; reducing wildfire threats (Sheuyange et al., 2005; Shaffer, 2010). Savanna burning projects have the potential to not only complement these traditional practices (e.g. UN-FAO, 2011), but also to legitimise, strengthen and incentivise them toward sustainable development in impoverished communities throughout the region. Operationally, the Australian methodology allows for co-occurrence of commercial savanna burning and beef-cattle enterprise activities (CoA , 2018), highlighting that savanna burning projects do not require exclusive land use rights to function.

Botswana, like several southern African countries, has a well-developed formal system of Community-Based Natural Resource Management (CBNRM) that aims to serve the interests of local communities for developing commercial enterprises and benefits derived from natural resources (e.g. ecotourism, wildlife trophy hunting, forest products). Such community-based organisations, including that serving the Tsodilo Hills community, could provide an appropriate governance structure for implementing commercial community-based fire management, but often lack institutional commitment and support to deliver this (see Mmegionline, 2020). Governance, resource availability and technical capacity are enduring challenges in CBNRM throughout the region (Chevallier, 2016), requiring considerable long-term support to implement effective savanna burning projects.

Further, despite the availability at Tsodilo of a rich local record of traditional fire knowledge describing prescribed EDS burning practices (Xuma et al., 2020), an appropriate regional fire management plan (Magole et al., 2017), and an active fire team supported by the national fire management authority (Dept Forestry & Range Resources), community inclusivity in addressing local fire management needs is effectively discouraged under current national fire suppression policy (Dube, 2013). Similar fundamental regulatory and local governance capacity issues affecting community engagement in fire management activities, and questions concerning the legal rights of local communities to commercial carbon benefits from savanna burning projects, are common to all southern African settings (Appendix S2).

Finally, our focus has been on assessing the potential of implementing prescribed fire management for emissions reduction purposes. As noted by Lipsett-Moore et al. (2018), potential exists for extending the base method to include carbon sequestration in living biomass and soils, but both present challenges. A fire and tree biomass dynamics sequestration methodology, based on a gap-phase modelling approach (Ryan and Williams, 2011), but requiring onerous annual field verification, is available for eastern Miombo woodlands covering parts of Malawi, Mozambique, Tanzania (VCS, 2015a). In Australia, a similar sequestration methodology, but based on modelling of fire regime effects on a substantial empirical spatio-temporal dataset, is under advanced development for inclusion in the National Greenhouse Gas Inventory. For soils, whereas a global-scale methodology addressing carbon sequestration in grasslands associated with grazing and fire impacts is available (VCS, 2015b), in African and Australian studies relationships between fire regime (and grazing) impacts and soil organic carbon stocks in typically infertile savanna soils to date have been inconsistent (Frost and Robertson, 1987; Allen et al., 2014).

Ecologically contentious is the appropriateness of sequestration schemes which potentially incentivise woody encroachment in open savanna and grassland habitats (Parr et al., 2014; Abreu et al., 2017), with associated undesirable consequences for various wildlife (Sirami and Monadjem, 2012; Parr et al., 2014; Archibald, 2016). Evidently, the design of savanna burning projects incorporating either emissions abatement or especially living biomass sequestration approaches needs to be responsive and adaptive to such issues.

## 5. Conclusion

It is widely recognised that LDS savanna wildfires in substantial areas of southern Africa annually contribute globally significant greenhouse gas emissions as well as having associated major negative impacts on regional biodiversity conservation values and people's livelihoods. In this paper we have posed the question whether application of an incentivised savanna burning emissions reduction approach comparable to that implemented in Australia, involving the undertaking of strategic prescribed burning under mild EDS fire-weather conditions, is technically feasible in southern African fire-prone savanna landscapes. Our assessment, based on exploratory field trials in regionally representative, relatively sparsely populated, savannas in fire-prone north-west Botswana and northern Mozambique, illustrate that the approach is technically feasible particularly with respect of: LDS fine fuels tend to be markedly greater than EDS fuels given seasonal leaf litter inputs; LDS wildfires tend to be significantly more intense and combust more fuels; methane and nitrous oxide emission factors are essentially equivalent in EDS and LDS periods under cured fuel conditions. Despite this positive technical assessment, we note that there are considerable medium-to long-term institutional, capacity building, and ecological challenges involved with developing effective emissions abatement savanna fire management projects in southern Africa.

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## Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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## Appendix A. Supplementary data

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