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Recent fire regimes in the Etosha National Park and adjacent areas in northern Namibia

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ABSTRACT

We assessed the recent fire regimes in the semi-arid savannas of Etosha National Park and adjacent areas in northern Namibia using MODIS satellite imagery from 2001 – 2025 across gradients of vegetation types, mean annual rainfall (200 – 500 mm) and land ownership. Fires were highly seasonal, concentrated in the two driest months of the year (September and October). The average fire return period over 25 years was 6.9 years in Etosha National Park (where fires are often allowed to spread freely and where additional prescribed burns have been implemented), but more than four times greater (31.8 years) on adjacent freehold farms where prescribed fires were not conducted and aggressive fire suppression was practiced. The proportion of the Etosha National Park that burnt annually ranged from zero to 47.6% and the fire regime was characterised by relatively infrequent cases in which an exceptionally large area (44% on average) burnt in a single year. Some findings were counter-intuitive in that certain vegetation types in areas of low mean annual rainfall (< 300 mm) burnt frequently (fire return period 4.3 years) while others in higher rainfall areas (> 400 mm) experienced infrequent fires (fire return periods 34 – 206 years). Fire management has been adapted over time, with the most recent policy seeking to implement patch mosaic burns to break up continuous fuel loads, but implementation is challenging due to a lack of skilled capacity and funding.

1. Introduction

Fires are a common feature across a large proportion of sub-Saharan Africa (Archibald et al., 2010a). In African savannas, the area that burns is correlated with grass production, which in turn is correlated with the amount of rainfall over the preceding wet seasons (Archibald et al., 2010b; van Wilgen et al., 2004). Climate also influences fire regimes, with fire return periods increasing as mean annual rainfall declines (O'Connor et al., 2011). Fire is widely used in the management of savanna protected areas in Africa, where a recent review identified 15 distinct fire management practices that were used to achieve a range of ecological and social goals (Nieman et al., 2021). Nieman et al.'s review also concluded that there was still little consensus on a broad fire management approach that would adequately cater for a wide variety of ecological and social conditions, and that this frequently left managers without clear guidelines for implementing appropriate fire regimes.

Although indigenous people have used fire in Africa for millennia (Thompson et al., 2021), the fire regimes that characterised the arid savannas in northern Namibia in pre-colonial times have not been documented (Hall, 1984). Colonial governments in Africa were generally opposed to the use of fire to manage woodland ecosystems (Trapnell, 1959; Nieman et al., 2021), but there are no records of fire policies pertaining to northern Namibia prior to 1970. The earliest available source describes a policy that was adopted in 1970 for managing fires in the natural vegetation in the Etosha National Park in northern Namibia (Siegfried, 1981). This policy specifically forbade the use of prescribed burning, and called for any unplanned wildfires to be contained as far as possible. It was only in 1981 that fire was recognised as ecologically important, and the park adopted a burning strategy that attempted to simulate the incidence of lightning fires (Stander et al., 1993). The approach was based on seasonal rainfall and the post-fire age of the vegetation. Under this system, areas (termed 'blocks') were selected for prescribed burning in the late dry season (September to October) if the preceding wet season's rainfall was higher than the 20-year running mean, and priority was given to areas with the greatest post-fire age. Fires that were started by lightning were allowed to burn freely within priority blocks but were contained within the block. Because lightning fires were often halted by roads within the priority

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blocks their spread within the block was facilitated by deliberate ignitions. Wildfires initiated by other sources of ignition were contained as far as possible (Stander et al., 1993). The system was later refined to include additional criteria, in which areas initially identified as suitable for burning were excluded if grass fuel loads were low (< 2000 kg/ha) and if < 20% of the grass sward was moribund (du Plessis, 1987).

The occurrence of large wildfires in 2009 and 2011 precipitated an abrupt change in Etosha's fire management policy in 2012. This involved a shift in the season of prescribed burns to the early dry season (May to July) when fires would be less intense and more fragmented. Despite the intent to manage fires in this way, the implementation of this policy has been hampered by funding constraints, a lack of capacity, and a shortage of experienced staff (Anon., 2021). Turner et al. (2022) also noted that the implementation of early dry season burns in Etosha had proved difficult because of a lack of clear guidelines, varied fire suppression policies in the areas surrounding Etosha, and difficulties in changing negative perceptions of fire.

Almost all commercial framers and other landowners to the south of Etosha National Park do not use fire as a management practice (Le Roux, 2011). When unplanned wildfires occur, vigorous attempts are typically made to extinguish or contain them, and this is often achieved through the collaborative efforts of neighbours (Le Roux, 2011). In these areas, fires are not considered in terms of any ecological role that they may play, as the major goal is to prevent the widespread loss of grazing.

In this paper, we assess the recent fire regimes of the Etosha National Park and bordering farms to the south of the park using remotely sensed MODIS imagery for the period 2001 to 2025. We use this information to examine the variation in these regimes across broad vegetation types, a gradient of mean annual rainfall, and land use categories. We expected to find, as has been found elsewhere, that the annual extent of area burnt was correlated with variable annual rainfall, and that fire return periods were longer (i.e. fire would become less frequent) with decreasing mean annual rainfall (Archibald et al., 2010; van Wilgen et al., 2004). Based on our findings, we also discuss remaining gaps in understanding, and possible approaches to fire management that would be appropriate for achieving conservation goals.

2. Materials and methods

2.1. Study area

The Etosha National Park (approximately 22 800 km^2) is situated in northern Namibia (centred on 19°02′S and 16°28′E; Fig. 1). A large proportion of the park is occupied by the Etosha Pan, a dry salt pan that covers 4730 km^2. The climate is semi-arid with an average temperature of 22.1°C. Mean annual rainfall ranges from around 250 mm in the west to 450 mm in the east. Precipitation occurs mainly during the summer months, with > 98% falling in the months from November to April. The mean elevation of Etosha National Park is 1129 m above sea level.

Mendelsohn et al. (2002) included eight vegetation types in the study area. These were (in order of area occupied, see Table 1): karstveld mixed woodlands on mollic leptosols; pans, which are extensive, but essentially are unvegetated and almost never burn, and were excluded from our analysis; western Kalahari broadleaved woodlands on ferralic arenosols; mopane shrublands on ferralic arenosols, dominated by *Colophospermum mopane* trees, Fig. 2; Etosha grass and dwarf shrublands on calcaric regosols; north-eastern Kalahari broadleaved woodlands on ferralic arenosols; the western highlands which support grasslands and scattered trees; and Cuvelai drainage floodplain grasslands and woodlands.

Large herbivorous mammals are found throughout the area, and include browsers (e.g. south-western black rhinoceros, *Diceros bicornis bicornis*; Angolan giraffes, *Giraffa camelopardalis angolensis*; and kudu, *Tragelaphus strepciceros*), grazers (e.g. white rhinoceros, *Ceratotherium*

Fig. 2. Woodlands south of central Etosha National Park, dominated by mopani trees (*Colophospermum mopane*). Photograph B.W. van Wilgen.

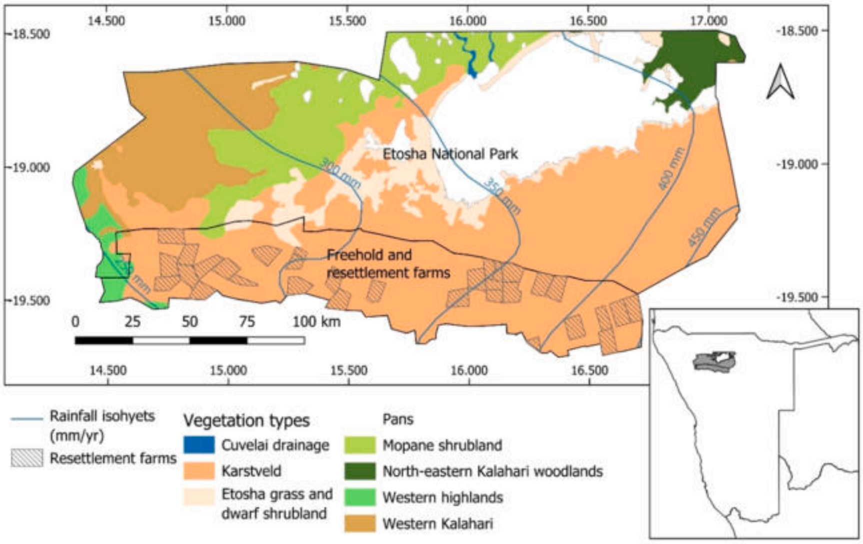

Fig. 1. Geographic location of Etosha National Park and adjacent freehold and resettlement farms in northern Namibia, showing vegetation types and isohyets of mean annual rainfall used in the analysis of fire regimes. The inset shows the location of the study area in northern Namibia.

simum; Burchell's zebras, *Equus quagga burchellii*; Hartmann's mountain zebras, *Equus zebra hartmannae*; blue wildebeest, *Connochaetes taurinus*; and gemsbok *Oryx gazella*); and mixed feeders (e.g. African elephants, *Loxodonta africana*; black-faced impala, *Aepyceros melampus petersi*; and springbuck, *Antidorcas marsupialis*). Termites, notably in the genus *Macrotermes*, are important consumers of above-ground plant material (Boutton et al., 1983). The Etosha National Park is a popular destination for international tourists, making an important contribution to the Namibian economy (Kimero et al., 2025).

Our study included a 30 km buffer to the south of the Etosha National Park. A buffer to the south, rather than one surrounding the park, was chosen because land to the north of the park (north-central Namibia, historically referred to as Ovamboland) differs markedly from the Etosha National Park and freehold land to the south, reflecting fundamentally different land tenure systems, population densities, and management objectives. Land to the south of Etosha National Park is divided into farms of between 5000 – 10 000 ha, with both private and communal ownership. Land in the buffer area is utilised for various purposes, including raising commercial livestock, game farming and hunting, tourism and conservation, and human population density is very low. Charcoal production using *Colophospermum mopane* and other tree species in the genera *Terminalia, Vachellia* and *Senegalia* is a common form of land use, and charcoal kilns often provide a source of ignition for bushfires. Land parcels are typically fenced, often restricting the movement of wildlife across the landscape. North-central Namibia is characterised by a very high rural population density, with dense networks of homesteads (often clustered into villages). There is very little unoccupied land, which is made up of a mosaic of fields, grazing, and settlements.

For the purposes of assessing fire regimes, we excluded the Etosha Pan and other large pans from all analyses. For this analysis, we recognised two land use categories in the buffer area. The first was land in private ownership, utilised for livestock farming, hunting, tourism or conservation purposes. The second was resettlement areas, which provide land access to marginalized indigenous groups to build livelihoods and address historical land inequalities, and allow for communal management for a range of purposes (Odendaal and Werner, 2020).

2.2. Baseline data

Our study was based on remotely sensed satellite records of fire occurrence, gridded rainfall data that combines satellite observations with in-situ station observations, and a vegetation map (described above). The area burnt between 1 January 2001 to 31 October 2025 was obtained from the MODIS MCD64A1 Version 6 burnt area product. This product differentiates burnt and unburnt surfaces at a resolution of ± 500 × 500 m (e.g. Smit et al., 2024) at an acceptable level of accuracy (Campagnolo et al., 2021) and records the dates on which the relevant pixels burnt. The area burnt was extracted for each month individually, and for each calendar year (a "fire year", Fig. 3).

Rainfall data were sourced from the Climate Hazards Center Infrared Precipitation (CHIRPS v3) product available on Google Earth Engine (Gorelick et al., 2017). CHIRPS incorporates 0.05° (~5 km) resolution satellite imagery merged with in-situ station data to create a gridded rainfall time series (Funk et al., 2015). We used these data to divide the study area into bands spanning 50 mm of mean annual rainfall from 200 to 500 mm (Fig. 1). Rainfall data were summed for each year from 1 July to end of June of the following year so that the rainfall totals included the full wet season which extends across two calendar years, i.e. from November in one year to April the next year (a "rainfall year", Fig. 3). This allowed us to examine the relationship between rainfall in a rainfall year, and area burnt in the following fire year. We examined fire frequency and seasonality for the entire study area, as well as for different categories of land. These categories included the area within the Etosha National Park, privately-owned land and resettlement areas in the buffer zone, the area within each vegetation type, and the area in different rainfall categories separately.

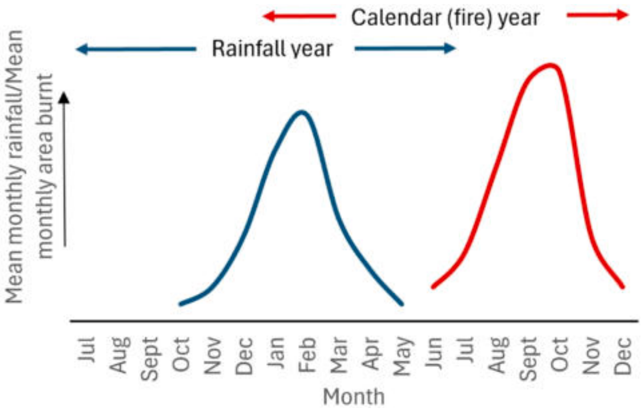

Fig. 3. Schematic illustration of the rainfall year (1 July to 30 June) and calendar or fire year (1 January to 31 December). Data are hypothetical means for monthly rainfall (blue) and monthly area burnt (red) for illustrative purposes.

2.3. Fire regime analysis

The mean fire return period for each vegetation type, land use category, and mean annual rainfall category was estimated as $RP = y/(b/a)$, where RP is the return period in years, y is the number of years over which fires were recorded, b is the extent of all fires recorded over y years, and a is the area over which fires were recorded (van Wilgen et al., 2000). The area of each land parcel examined, and the extent of area burnt in that land parcel was obtained from the baseline data described in section 2.2 above (areas for land parcels and extent of fires are provided in Table 1). The return period estimated using this formula assumes that fires are randomly distributed in space, that burning is uniform over time, and that there is no strong spatial autocorrelation. As such, the return period should be interpreted as a landscape-average return interval. The return periods between actual fires can be highly skewed, and to account for this we constructed fire interval distribution curves (i.e. the likelihood that a given pixel would burn again within a given time after a previous fire) for different vegetation, land use or rainfall categories, using Kaplan–Meier survival analysis (Kaplan et al., 1958; D'Arrigo et al., 2021) implemented in the R package *survival* (Therneau, 2024). Annual burnt-area raster layers were overlaid on a systematic 1 km^2 grid of 24 860 sampling points to construct fire histories for each point. For every observed fire at a point, the time-to-event (duration) was calculated as the number of years from that fire until either the next fire (uncensored event = 1) or the end of the observation period in 2025 (right-censored event = 0). Points that never experienced fire during the 25-year period contributed no episodes and were excluded from the fire interval distribution analysis. Kaplan–Meier curves were produced for (i) the land tenure classes (freehold and resettlement areas combined), (ii) each vegetation type and (iii) rainfall class, respectively, showing the cumulative probability of fire reoccurrence as a function of years since the previous fire.

Fire seasonality was estimated from the total area burnt each month, for all fires on record. The relationship between the area burnt each year and the rainfall in the preceding wet season for the Etosha National Park was estimated by comparing the area burnt in each calendar year with the rainfall year over the preceding season (e.g. the area burnt in the 2001 calendar year was compared to rainfall between 1 July 2000 and 30 June 2001, Fig. 3). Burnt area or CHIRPS rainfall pixels were assigned to different vegetation types, land tenure classes, or rainfall zones when the pixel centroids were located within the boundaries of the overlaying mapped polygons.

To examine the relationship between seasonal rainfall and annual area burnt in Etosha National Park and adjacent farmland, we conducted separate Pearson correlation analyses for each tenure class using rainfall data in each rainfall year and burnt area in the following fire year (Fig. 3). To account for potential carry-over effects of fuel from preceding rainfall, we correlated burnt area with rainfall from the two earlier successive rainfall years separately, and as a mean over two years. Significant correlations ($P < 0.05$) were further examined by simple linear regressions. A multiple linear regression was also used to assess the independent influences of two successive rainfall years on burnt area.

To assess whether fire seasonality differed between tenure classes, monthly burnt area was standardised by expressing each month as a proportion of the annual burnt area within each zone and year. This removed absolute differences in burnt area and allowed comparison of seasonal distribution patterns. Monthly proportional distributions between Etosha National Park and adjacent farmland were compared using the non-parametric two-sample Kolmogorov–Smirnov tests. Tests were conducted separately for each month, and p-values were adjusted for multiple comparisons using the Holm correction.

All statistical analyses were conducted in R (version 4.5.0; R Core Team).

3. Results

3.1. Fire return intervals

The mean fire return interval in the Etosha National Park, excluding pans, was 6.9 years but was almost 2.5 times greater in the resettlement areas and more than four times greater on freehold farms and reserves south of the park (Table 1; Fig. 4). The fire return periods for most vegetation types ranged between 4.3 and 13.5 years but were substantially longer for Cuvelai drainage floodplains (34 years), western highlands (37.1 years) and north-eastern Kalahari broadleaved woodlands (206.9 years). Where mean annual rainfall was between 250 and 500 mm, fire return periods ranged from 7.5 to 10.2 years. In areas of very low rainfall (<250 mm) the mean fire return period was substantially longer (48.2 years).

Fire interval distribution curves indicate that 50% of the area covered by three vegetation types (Karstveld, Mopane shrublands, and western Kalahari woodlands) would experience a successive fire within five years of a previous fire (Fig. 5), while in Etosha grass and dwarf shrublands it took more than 10 years on average for half of the area to burn again. In the north-eastern Kalahari woodlands, and western highlands, fires were rare, and it took more than 15 years for around 20% of the area to burn again, while most of the area that burnt once did not experience a subsequent fire in the 25 years examined here. Land tenure also had a marked effect, with 50% of the land in Etosha National Park burning again within 4 years of a previous fire, compared to 10 years in freehold and resettlement farms (Fig. 5). Mean annual rainfall had less effect on the shape of fire interval distribution curves, with 50% of the land in most rainfall classes experiencing a subsequent fire within 4 to 7 years. This increased markedly to 15 years when the mean annual rainfall was below 250 mm (Fig. 6).

3.2. Seasonal distribution of fires

Fires occurred predominantly in the late dry season in September and October. Fires in these two months accounted for 71.2% of all fires recorded between 2001 and 2025 in Etosha National Park and 66.3% in the adjacent farming land; fires were virtually absent between January and May inclusive (Fig. 7). Monthly burnt area during the 25-year period was strongly influenced by large areas burnt in a small number of years, resulting in a right-skewed distribution where monthly means greatly exceeded medians. When monthly burnt area was expressed as a proportion of the annual total, the seasonal distribution of fires did not

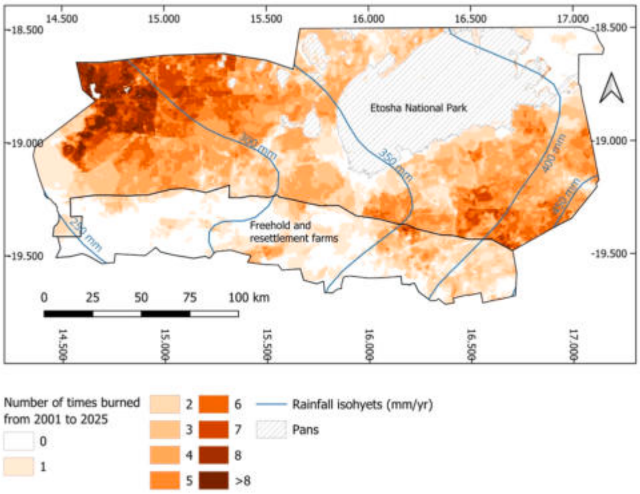

Fig. 4. The fire frequency (number of times burnt over 25 years) in the Etosha National Park and adjacent areas.

Table 1
Areas burnt, estimated fire return intervals and proportion of area remaining unburnt, between 2001 and 2025 in different categories of land use, vegetation types, and mean annual rainfall for the Etosha National Park and adjacent areas.

Aspect examined	Category	Total area (km^2)	Area burnt over 25 years (km^2)	Mean fire return interval (years)	Proportion remaining unburnt (%)
Study area	Entire area	24866	71264	8.7	20.6
Land use	Etosha National Park (excluding pans)	17717	64497	6.9	8.8
	Resettlement farms south of Etosha National Park	1701	2479	17.1	36.2
	Freehold farms and reserves south of Etosha National Park	5449	4287	31.8	53.9
Vegetation type	Cuvelai drainage	52	38	34.0	48.1
	Etosha grass and dwarf shrubland	2045	3784	13.5	24.6
	Karstveld	14895	34757	10.7	23.9
	Mopane shrubland	3240	12462	6.5	3.5
	North-eastern Kalahari woodlands	771	93	206.9	88.7
	Western highlands	496	334	37.1	44.6
	Western Kalahari	3368	19795	4.3	0.5
Mean annual rainfall category (mm)	200-250	212	110	48.2	66.8
	250-300	7661	24271	7.9	26.5
	300-350	7291	21076	8.6	18.4
	350-400	6013	16639	9.0	12.0
	400-450	3517	8601	10.2	24.7
	450-500	171	567	7.5	9.7

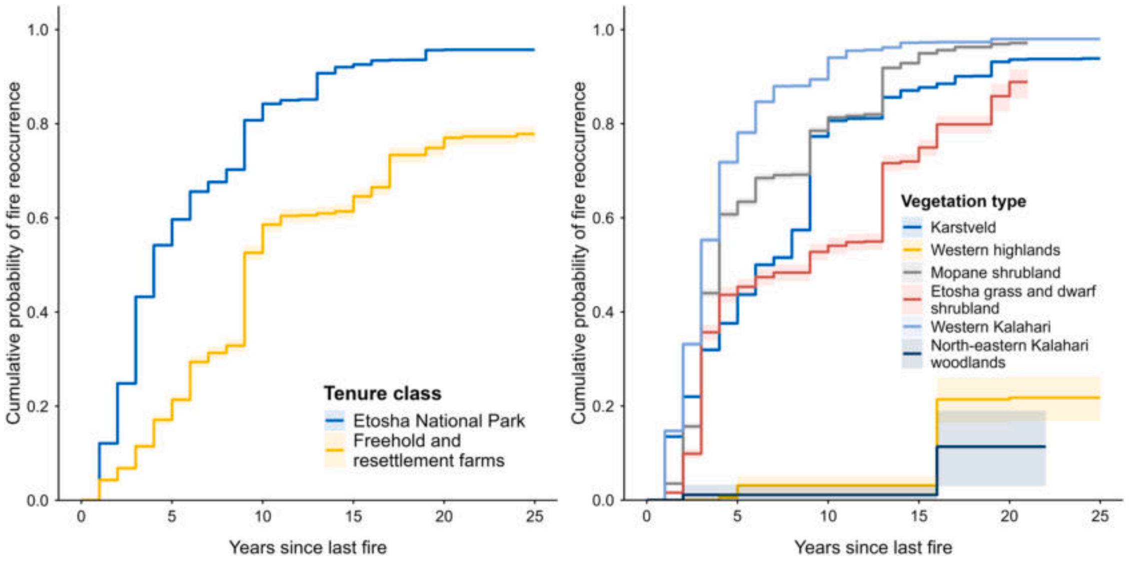

Fig. 5. Fire interval distribution curves for the Etosha National Park and freehold and resettlement farms south of the park (left panel) and six vegetation types (right panel) in the Etosha National Park and surrounding areas derived from 25 years of burnt area data. The fire interval distribution curves were calculated for areas that burnt at least once during the 25 years. Shaded areas represent pointwise 95% confidence intervals.

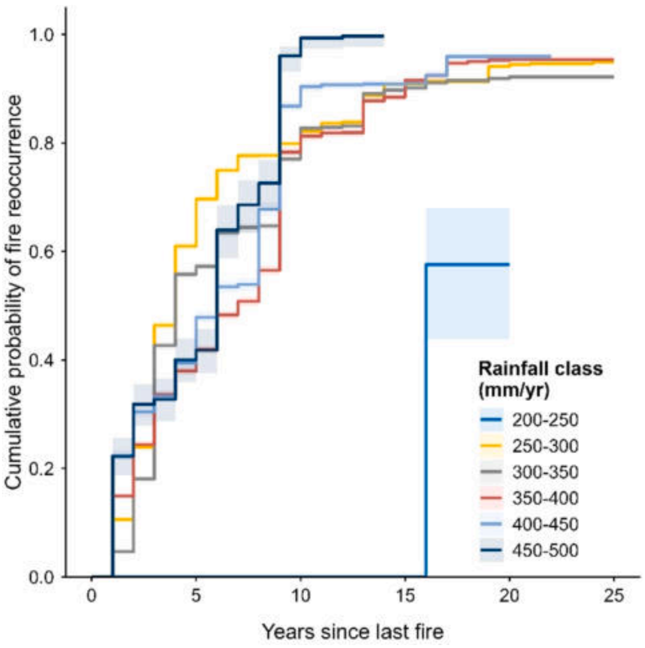

Fig. 6. A comparison of fire interval distribution curves for six classes of mean annual rainfall in the Etosha National Park and freehold and resettlement farms south of the park derived from 25 years of burnt area data. The fire interval distribution curves were calculated for areas that burnt at least once during the 25 years. Shaded areas represent pointwise 95% confidence intervals.

differ between the Etosha National Park and the buffer area to the south of the park (two-sample Kolmogorov–Smirnov tests with Holm correction, $p > 0.05$), although a smaller proportion of the area burnt annually outside of the park (on average 14.6% in Etosha National Park vs 3.8% outside of the park).

3.3. *The effect of antecedent rainfall on area burnt*

The total rainfall in individual rainfall years between 2001 and 2025 varied markedly and ranged between 119 and 711 mm in Etosha and 122 and 747 mm in the adjacent areas. As expected, burnt area was positively correlated with mean rainfall in the preceding rainfall year in Etosha National Park (Pearson's $r = 0.67$, $p < 0.001$, Fig. 8). A linear regression indicated that approximately 44% of the variation in annual burnt area was explained by rainfall in the preceding rainfall year, while rainfall in the rainfall year preceding that showed no correlation with burnt area in Etosha ($r = 0.24$, $p > 0.05$, Fig. 8). In the multiple regression model (Adjusted $R^2 = 0.39$, $F_{2,22} = 8.8$, $P < 0.01$), preceding rainfall over two years was also not a significant predictor ($\beta = 0.11$, $P > 0.05$), whereas most of the variation in burnt area extent was explained by current rainfall year ($\beta = 13.7$, $P < 0.001$). Burnt area was minimal when rainfall in the preceding rainfall year was below 200 mm.

On the other hand, on the freehold and resettlement land, neither current rainfall, preceding rainfall nor the mean of these two rainfall years were correlated with burnt area year ($P > 0.05$, Fig. 9). This suggests that management inputs, in the form of effective fire suppression and containment, has negated any effect that inter-annual variation in rainfall may have had on the area burnt annually.

3.4. *Variation in area burnt annually*

The area burnt annually during the 25-year period was highly variable, with a few years in which an exceptionally large area burnt. The average area burnt per year in the Etosha National Park was 2583 km^2 (Fig. 10), but in 11 of the 25 years the area burnt was below 3% of the park (average 216 km^2, or 1.2% of the area of the park excluding pans). On the other hand, large fires exceeding 7000 km^2 occurred in only six years, and averaged 7817 km^2, or 44% of the park (Fig. 11). Years in which exceptionally large areas burn are therefore relatively infrequent, but because they burn such large areas, they have a disproportionate impact on fire regimes. Most large fires occurred after above average rainfall in the antecedent wet season (range 497 – 711 mm), although antecedent rainfall before the 2021 fires was only 360 mm (Table 2). Antecedent rainfall explains around 44% of the variation in annual area burnt (Fig. 8), so other factors must have affected the large area burnt in 2021. These could include more severe fire weather, or a buildup of fuel over the past 8 years when only 3.4% of the area burnt annually on average (Fig. 10).

Large fires are newsworthy, especially if large mammals are killed, and they have acted as triggers to reassess fire management policies (Table 2). Following the fires in 2011, when numerous animals were killed, a policy of patch mosaic burning was introduced to reduce fuel

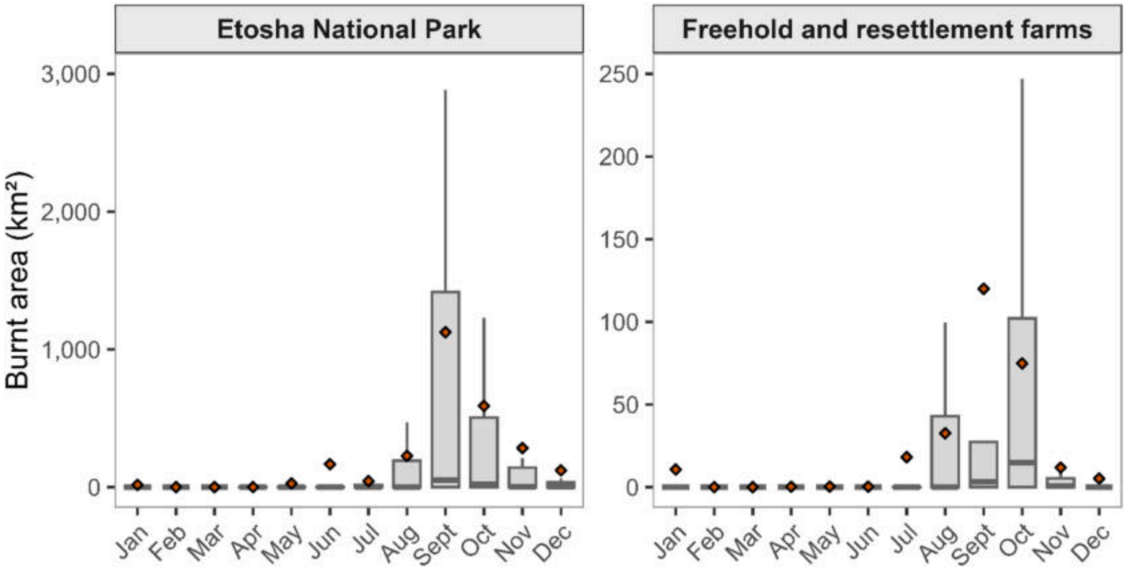

Fig. 7. The area burnt per month in Etosha National Park and in freehold and resettlement areas in a 30-km buffer south of the park between 2001 and 2025. In the boxplots the horizontal line represents the median. The box spans the 25th to 75th percentiles, and whiskers extend to 1.5 × the interquartile range. The diamonds indicate the mean burnt area for each month.

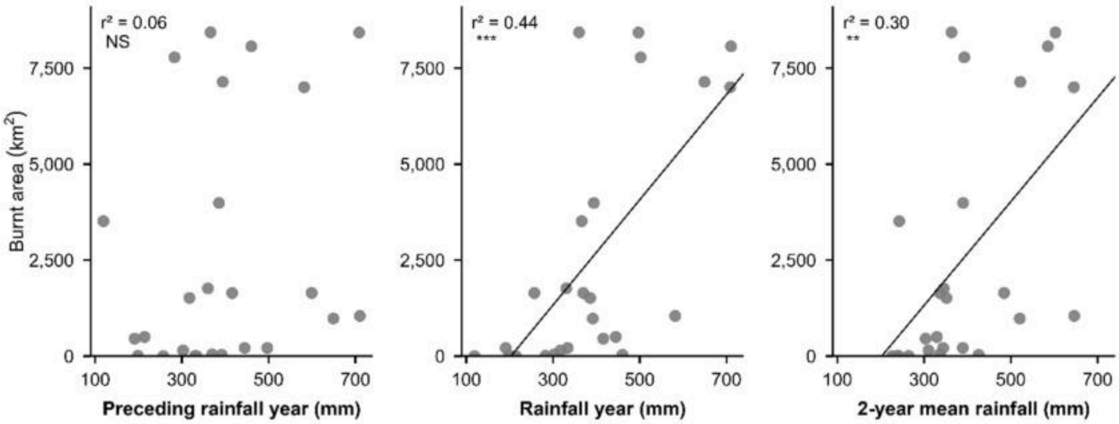

Fig. 8. Relationships between area burnt in a fire year in Etosha National Park and rainfall two years prior to the fire year (left panel); rainfall over the immediate past rainfall year (centre panel); and mean rainfall over the past two years (right panel) for 2001 through 2025. The regression equation for the immediate past rainfall year is y = 13.72x – 2787.33 (r^2= 0.44). NS = not significant; * = P<0.05, ** = P<0.01; *** = P<0.001.

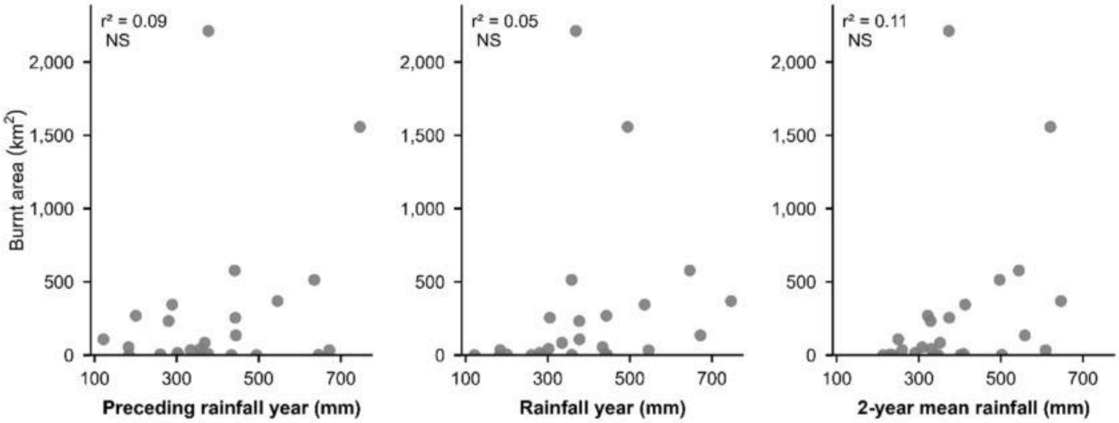

Fig. 9. Relationships between area burnt in a fire year in land south of Etosha National Park, and rainfall two years prior to the fire year (left panel); rainfall over the immediate past rainfall year (centre panel); and mean rainfall over the past two years (right panel) for 2001 through 2025. None of the relationships were significant (NS, P > 0.05).

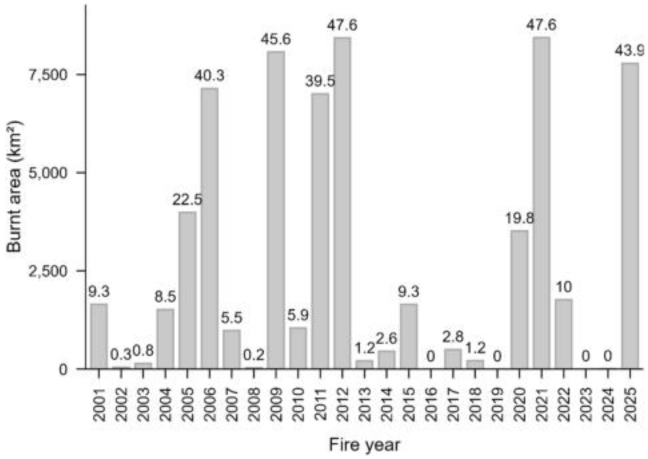

Fig. 10. The annual area burnt from 2001 to 2025 in the Etosha National Park. The percentage of Etosha that burnt per year is shown above the bars.

loads (Anon., 2021). No animals were reported as killed following extensive fires in 2021, and this was attributed by officials to reduced fuel loads following the implementation of the policy of patch mosaic burning (Anon., 2021; Smit, 2021). This seems unlikely as only 3.4% of the area burnt annually on average over the previous 8 years, so prescribed burning would have had a minimal effect. Lower fuel loads may also explain the lack of mortality following the fires in 2012, which were preceded by relatively low (497 mm) rainfall and extensive fires in the previous year (Table 2). These explanations are speculative at this stage.

4. Discussion

4.1. Differences in fire return intervals

We found a greater than four-fold difference in fire return intervals between Etosha National Park (6.9 years) and farmed land to the south of the park (31.8 years). This is most likely due to a reluctance to use fire as a management practice on these farms, as well as to effective collaborative efforts by landowners to contain or extinguish any fires that occurred (Le Roux, 2011). In essence, the large difference was the result of effective containment on farms, which was not the case in Etosha National Park where many fires in the park were not suppressed (see discussion in Section 1). Research into the effects of fire at an arid location in central Namibia (361 mm mean annual rainfall) found that woody seedling and sapling mortality was higher in burnt areas compared to unburnt areas (Joubert et al., 2012). The authors concluded from this that fire would interrupt the transition from open grassy savanna to thicket in arid savannas and suggested that farmers who prevent fires are likely to experience encroachment by woody shrubs and trees in future. This suggests that farmers are trading short-term gains (preventing the loss of grazing to fire) for long-term losses (loss of grazing to woody encroachment). Although multiple factors are responsible for woody thickening in southern Africa (O'Connor et al., 2014), human suppression of fires that would have prevailed under a pre-colonial burning regime is likely to be playing an important role in the woody thickening process.

We expected that vegetation types in areas of lower mean annual rainfall would burn less frequently due to lower fuel loads. The western highland grass and shrublands areas are situated in the far west of Etosha National Park where mean annual rainfall is low (200 – 300 mm), and fire return intervals were longer (37.1 years) as expected. The western Kalahari, on the other hand, experienced regular fires (fire return period 4.3 years) despite low mean annual rainfall (< 300 mm) which was unexpected. The Cuvelai drainage floodplains (fire return period 34 years) occupy a very small proportion of the Etosha National Park (0.22%) north of the Etosha Pan (Fig. 1). These areas would presumably be more heavily grazed due to their location along drainage lines, reducing above-ground biomass (fuels) and preventing frequent fires despite relatively higher mean annual rainfall. The north-eastern Kalahari broadleaved woodlands experienced a remarkably long average fire return interval (206.9 years). This vegetation type occurs at the eastern end of the Etosha National Park, which receives a relatively high mean annual rainfall (400 mm). The reasons for the paucity of fires in the north-eastern Kalahari broadleaved woodlands are not fully understood but could be related to high levels of grazing which removes fuels. Research in African savannas elsewhere has demonstrated that a high biomass of grazing mammals led to a significantly lower extent of fires, and that this effect was most marked in arid or semi-arid areas (~ 450 mm/yr) (Smit and Archibald, 2019).

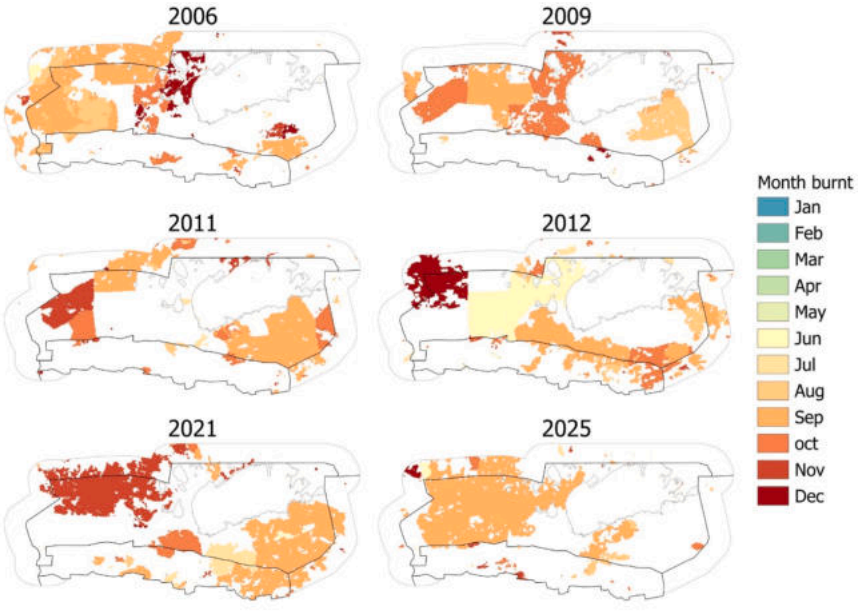

Fig. 11. The distribution of fires in six years between 2001 and 2025 when more than 40% of the Etosha National Park (large pans excluded) was burnt.

Table 2
Salient features of fires during six years between 2001 and 2025 in which the area burnt exceeded 40% of the Etosha National Park (excluding large pans). The area burnt was recorded for the calendar year. Total antecedent rainfall prior to dry season was recorded between 1 July in the previous calendar year to 30 June in the current year (see Fig. 3). Other information has been gleaned mainly from reports in the media.

Year	Area burnt (km^2) and proportion of Park burnt (%)	Total antecedent rainfall (mm)	Cause of fire	Wildlife mortality	Notes
2006	7282 (40.3)	649	Lightning	No animals known to have been killed (Shigwedha, 2006).	Unserviceable equipment contributed to inability to combat the fire (Briweidlich, 2006).
2009	8240 (45.6)	711	No information	No information	No information
2011	7138 (39.5)	709	Charcoal kiln adjacent to park (Nasa, 2012; Smith, 2011a, 2011b)	25 black rhinos, five white rhinos, 11 elephants, three lions, 60 giraffes and 30 kudus were killed (Smith, 2011b), many in backburns set to contain the fires.	Government responses included an intention to introduce prescribed burns to reduce fuel loads, and to fund the purchase of equipment and training of staff (Smith, 2011b).
2012	8601 (47.6)	497	Escaped prescribed fire	There were no reports of animal deaths (Nasa, 2012)	Although a large area was burnt, the effects were not regarded as severe.
2021	8601 (47.6)	360	No information	There were no reports of animal deaths (Smit, 2021)	The absence of wildlife deaths was attributed by government to the earlier introduction of prescribed burning (Smit, 2021).
2025	7933 (43.9)	502	Charcoal kiln adjacent to park (Links, 2025).	One elephant, six springbuck and three grey duikers killed (Calitz, 2025)	Wind contributing to spot fires was reported as a contributing factor to the spread of fires (Links, 2025)

4.2. Priorities for research

A robust understanding of fire regimes and their determinants, and the effects of manipulating elements of the fire regime on the recipient ecosystems, is needed to underpin evidence-based fire management. Managers of fire-prone ecosystems need to know what actions they should take regarding where, when and how often to set prescribed burns, and whether to supress wildfires or allow them to burn unhindered. Our study has provided baseline information on the fire regimes of the Etosha ecosystem, which can be used to set initial targets for management, but several important questions remain. All fire management policies that have been adopted in the Etosha region over the past decades (outlined in section 1 above) have not had a basis in ecological understanding but have rather sought to introduce fire patterns that were assumed to be natural and therefore desirable, or to reduce fuel loads. Whether the desired outcomes of these policies are realistically achievable and/or ecologically desirable remains to be tested.

Some vegetation and land tenure types examined here experienced unexpectedly low fire frequencies, while others had unexpectedly high fire frequencies. The dynamics of rainfall cycles, levels of herbivory, and their effects on fuels are not well understood at present, especially in arid African savannas. There are very few African savanna areas where a reasonably sound research-based understanding of fire ecology and management has been developed, and these all receive higher rainfall than the arid and semi-arid vegetation of Etosha (250 – 450 mm), e.g. the Kruger (400 – 800 mm) and Hluhluwe-iMfolozi (550 – 1000 mm) parks in South Africa, and the Serengeti in Tanzania (800 – 1200 mm) (van Wilgen et al., 2014; Archibald et al., 2017; Eby et al., 2015).

Several issues deserve attention if our understanding of the determinants and ecological effects of fire in arid and semi-arid areas is to improve. Rainfall and drought both affect fuel dynamics and the occurrence of fires. The fire record in Etosha was characterised by relatively infrequent cases in which an exceptionally large area burnt in a single year. In Etosha, over 25 years, almost half of the park burnt every 4th year on average, while less than 3% burnt in almost half of the years. An ability to predict high fire years would require a better understanding of their causes and would be useful in preparing to deal with them.

Herbivory is also important, for two reasons. First, fires and herbivores are competitors for the same resource, grass fuels (Bond and van Wilgen, 1996). Numerous herbivores can prevent fires, or reduce their intensity, by consuming most available fuel (Smit and Archibald, 2019), while above-average rainfall generates abundant forage that can satiate herbivores, leaving sufficient fuels to support fires. Direct estimates of fuel loads, using remote sensing for example, could help to distinguish between these factors. Secondly, interactions between elephants and fires can have dramatic effects on trees in savannas (Vanak et al., 2012; Stevens et al., 2016) and the relative importance of this issue in drylands requires further investigation. Finally, a shift in burning from the late to the early dry season, as currently proposed (Anon., 2021; Turner et al., 2022), may have ecological implications that are currently not fully understood (Knowles et al., 2025). Turner et al. (2022) noted that studies on the “interactions between vegetation dynamics, herbivores and fire in the greater Etosha landscape are highly important for future research”. Research aimed at improving our understanding of these dynamics should be structured around regular monitoring and recording of inputs (rainfall, fuel loads, herbivore numbers, and the occurrence of fires) and outputs (plant species composition, biomass and structure). This could provide a broad basis for improving understanding over the longer term.

4.3. Implications for management

The Etosha National Park’s management policy recognises that fire plays an important ecological role and that fires must “be applied primarily to reduce fuel loads to limit the size and scale of fires, especially wildfires” (Anon., 2021). The intention is to shift the seasonality of fires from the late to the early dry season, by initiating patch burns early in the dry season to break up continuous fuel beds, as has been done elsewhere (e.g. Brockett et al., 2001; Mulqueeny et al., 2010). Managers in Etosha seek to ignite fires in the early dry season (May to July) when fires are characterised by low intensity, a high degree of patchiness, and a tendency to extinguish spontaneously overnight (Anon, 2021).

Implementation of this policy has commenced, but it has proven difficult to carry it out consistently because of a lack of skills and experience in the safe application of fires, and ongoing negative perceptions of fire (Anon., 2021; Turner et al., 2022). As a result, most of the park still appears to burn in unplanned fires, but data on the cause of fires are not routinely recorded. Occasional large wildfires cause considerable negative publicity, especially if large and charismatic animals are killed or injured in such fires, and this highlights the need for the effective implementation of a practical and defensible fire policy.

The current fire policy has proved difficult to implement because of the size of the area, limited access roads, and the lack of experienced managers to implement an effective patch mosaic burning strategy. Several studies in South Africa (Archibald et al., 2017; Brockett et al., 2001; Mulqueeny et al., 2010; Balfour et al., 2001) have found that patch mosaic burning was effective in increasing the heterogeneity of fire patterns in savanna protected areas. Many of these areas were relatively small (<1 000 km^2 in size) compared to the Etosha National

Park, but the practice was also shown to be effective in larger areas, e.g. the Kruger National Park in South Africa (van Wilgen et al., 2014), which is similar in size to Etosha National Park (20 000 km^2), but better resourced. A policy of regular prescribed burning was implemented for over half a century across the entire Kruger National Park (van Wilgen et al., 2004). The outcomes of the Kruger National Park's policy were later examined using satellite remote sensing, which revealed that some areas burnt very infrequently regardless of the prevailing fire management policy. A decision was taken not to deliberately burn these areas, and they were excluded from any future fire management interventions (van Wilgen et al., 2014), freeing up scarce capacity and resources to manage fires elsewhere. In Etosha National Park, the Cuvelai drainage, northeastern Kalahari woodlands, and western highlands also burn very infrequently (Table 1), and arguably need not be subjected to patch burning practices. This may relieve some pressure on managers, although the area excluded from active fire management would be less than 10%. The effective implementation of patch mosaic burning across the remaining fire-prone areas in Etosha National Park would require a larger investment into management capacity, training and fire control equipment if the occurrence of occasional extensive wildfires is to be effectively reduced.

CRediT authorship contribution statement

Brian W. van Wilgen: Writing – review & editing, Writing – original draft, Validation, Methodology, Investigation, Formal analysis, Conceptualization. **Cornelis van der Waal:** Writing – review & editing, Visualization, Validation, Software, Methodology, Investigation, Formal analysis, Data curation, Conceptualization.

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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