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ORIGINAL RESEARCH

What shapes fire size and spread in African savannahs?

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Abstract

Fires play an important role in savannah ecosystems, shaping among other things vegetation structure and altering species composition. As direct and indirect anthropogenic pressures on these ecosystems increase, fire dynamics in savannahs are expected to change in the coming decades, with potential impacts on ecosystem functioning. Although the ecological impacts of fires are relatively well-known, the factors that shape fire dynamics in these ecosystems have received less research attention. Using Pendjari National Park (Benin) as a case study, we assessed the importance of different biotic and abiotic factors in shaping fire size and spread in the region. Our results show that fires spread faster (1) in the middle of the dry season compared to the early or late dry season, (2) in areas that are far away from natural and anthropogenic firebreaks, and (3) in areas that are covered with highly flammable vegetation. By contrast, most vegetation types had little influence on fire size, which seems to depend instead on rainfall. Our approach and results highlight new avenues for satellite data to improve our understanding of fire dynamics in large, remote savannah ecosystems and to improve our ability to predict how fires spread, a key variable for wildlife management in the face of rapidly changing environmental conditions.

Introduction

Natural and human-ignited fires have a defining impact on Earth's ecosystems. Fires are a major source of global greenhouse gases and aerosols (Maraseni et al., 2016; Van Der Werf et al., 2010), and modify nutrient and water fluxes that shape ecosystem productivity (Beringer et al., 2004; Frost & Robertson, 1987). They moreover alter vegetation biomass and structure (Van Langevelde et al., 2003), transforming ecological communities through direct killing, and through their impacts on growth and reproduction rates as well as on species interactions (Frost & Robertson, 1987). Temporal and spatial shifts in fire dynamics are expected to trigger major domino effects on ecosystem composition and functioning (Furley et al., 2008; Shackleton & Scholes, 2000). As a result, scientists are increasingly paying attention to the factors shaping fire dynamics and impacts.

Of particular concern are the rapid observed changes in both land use and climatic conditions in savannahs (defined as grassland ecosystems characterized by the coexistence of grasses and trees; Bond & Parr, 2010), a type of ecosystem where fires exert particularly important control on ecological interactions and processes. In these ecosystems, fires shape vegetation structure by reducing woody vegetation height (Smit et al., 2010), stem density and biomass (Shackleton & Scholes, 2000), and, to a lesser extent, by altering vegetation composition of the herbaceous layer (Furley et al., 2008; Van Wilgen et al., 2007). As a result, changes in fire dynamics can have a profound effect on savannah biodiversity (Abreu et al., 2017). In recent decades, land use changes have significantly contributed to the depletion and fragmentation of savannah landscapes, such as through encroachment and overgrazing, which have been linked to reduced fire size and frequency (Andela et al., 2017; Van Langevelde et al., 2003). For instance, cropland expansion was shown to directly modify fire dynamics in many areas, reducing the sizes of individual fires and the overall area burned (Archibald et al., 2013; Clerici et al., 2005). In addition, changes in temperatures and precipitation can impact fuel load, combustibility, and ignition rates, potentially leading to fires becoming more severe and/or more common in many places (Furley et al., 2008; Jolly et al., 2015). Rising dioxide carbon levels have also been implicated in encroachment of savannahs by woody vegetation, which reduces the impact of fires on vegetation structure (Stevens et al., 2015). Land use change and climate change can moreover interact, with land use change potentially reducing the resistance and/or resilience of ecosystems to shifts in disturbance regimes caused by climate change (Schulte to Bühne et al., 2020).

Although fire ecology is a well-established research field, much remains to be known about the impacts of climate change and land use change on fire dynamics and subsequent changes in vegetation structure and composition. This is particularly true when looking at parameters related to fire spread (Andela et al., 2019; Frantz et al., 2016), with almost all large-scale studies to date having limited themselves to researching the factors influencing fire frequency or burned area size (see e.g., Abdi et al., 2018; Turner & Romme, 1994). However, the rate of fire spread is an important variable for fire management (Belval et al., 2015): knowing where existing or future fires may spread faster can for example enable managers to prioritize areas for the deployment of effective preventative (e.g. fire breaks creation, including by controlled burns) and acute (i.e., preventing active fires from spreading) measures. To help address this knowledge gap, we investigate how climatic conditions and landscape features shape fire size and spread in the savannah of Pendjari National Park, which is part of the W-Arly-Pendjari transboundary protected area complex. Covering some 36 000 km^2 and straddling three countries (Benin, Burkina Faso and Niger), the W-Arly-Pendjari complex is the largest remaining stronghold of West Africa's iconic flora and fauna, including cheetahs *Acinonyx jubatus* (Durant et al., 2017) and half of all West African lions *Panthera leo* (Henschel et al., 2014).

Since savannah fires are primarily fuelled by grasses and shrubs (Cheney & Gould, 1995; Hoffmann et al., 2012), which are known to be highly combustible (Pausas et al., 2017), we expect a higher percentage of such vegetation in a given area to accelerate fire spread (H1). Furthermore, because fires occurring in savannahs with vegetation types characterized by a higher fuel load and bulk density tend to spread less rapidly than fires occurring in areas characterized by vegetation types with lower burnable mass (Hoffmann et al., 2012), we expect fire size to be smaller in forest, tree savannah and woodland savannahs than in grass and shrub savannah (H2). Because reduced rainfall is known to lead to drier vegetation, we expect fires to spread at a faster rate in all types of vegetation (H3) and fires to be larger after periods of relatively low rainfall (H4). Finally, natural and anthropogenic barriers such as rivers and roads can act as natural barriers and stop fire spread (Oliveira et al., 2016; Syphard et al., 2007). We therefore expect fires to be constrained by potential fire breaks, with fires near natural and anthropogenic barriers spreading slower than fires away from potential fire breaks (H5; Table 1).

Materials and Methods

Study area

Pendjari National Park is a 2800 km^2 protected area situated in the north-west part of Benin (Fig. 1). In 1986 its national park status was upgraded to a UNESCO (United Nations Educational, Scientific and Cultural Organization) Man and the Biosphere Reserve (Sogbohossou et al., 2011). In the north and west, the Park is delimited by the Pendjari River. The climate is Sudanese-Guinean, with total annual rainfall varying between 900 and 1000 mm (Gnonlonfoun et al., 2019). Annual rainfall is high in the southwest of the Park and declines towards the northeast

Table 1. Summary of the hypotheses and outcomes of our analyses.

Hypothesis	Observed outcome
(H1) Fires spread more quickly where there is more highly ignitable vegetation	*Partially supported*. Areas with high level of shrubby vegetation experienced faster rates of fire spread, but coverage of grass savannah had no effect on fire spread.
(H2) Fires remain small in areas with a high proportion of trees (forests, tree savannah and woodland savannas) than in grass and shrub savannahs	*Partially supported*. Fire size was larger in shrub savannah, but neither the proportion of grass, or forest, tree and woodland savannas affected fire size
(H3) Fires spread at a faster rate later during the dry season, when rainfall has been low for many months. Fires also spread faster after weeks of little rainfall	*Partially supported*. Mean precipitation during the week preceding a given fire was shown to have a negative effect on rates of fire spread, with drier vegetation being associated with faster fires. Fires spread more slowly at the end of the wet season (October) and the end of the dry season (March/April).
(H4) Fires are smaller early during the dry season (due to a heterogenous patterns of vegetation curing), and after weeks in which rainfall has been high	*Partially supported*. Fire size did not vary over time within a given fire season, but higher amounts of rainfall in the week preceding the start of a fire led to bigger fires.
(H5) Fires near natural and anthropogenic barriers spread slower than fires far away from potential fire breaks	*Supported*. Rate of spread was reduced near firebreaks.

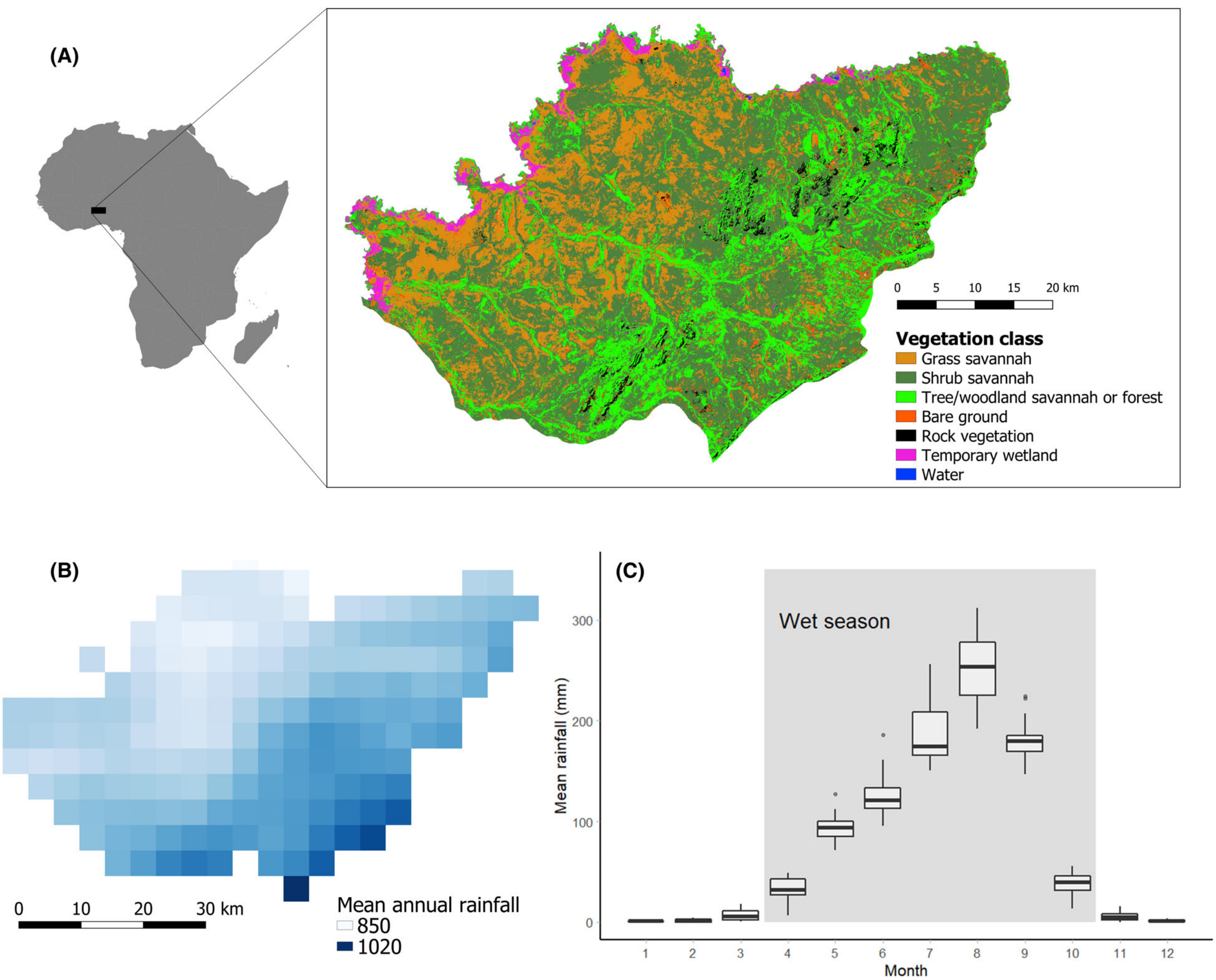

Figure 1. (A) Location of our study area (Pendjari National Park), with information related to vegetation type distribution. Pendjari National Park (ca. 2800 km^2) is part of the W-Arly-Pendjari transboundary protected area complex. (B) Mean annual sum of rainfall across Pendjari National Park (based on TAMSAT records from 2000 to 2018). (C) Monthly rainfall between 1998 and 2018 in Pendjari National Park based on rainfall records from the TAMSAT dataset.

(Fig. 1). Rain primarily falls during a single wet season, starting in April and continuing until October, with peak rainfall occurring in August (Fig. 1). During the dry season (November to March), mean monthly rainfall across Pendjari National Park is less than 7 mm, whereas mean monthly rainfall in August during that same time is 250 mm (average over the 1998–2018 period). The park is a mix of grass, shrub and tree savannah, with vegetation getting higher and denser along the Pendjari River (Lopes et al., 2020), but showing no strong relationship with the spatial rainfall gradient (Fig. 1). The Park is important for regional biodiversity, providing habitats that are essential to several endangered species (Sogbohossou et al., 2011). There is currently some fire management in Pendjari National Park, primarily late in the dry season when managers try to limit large fires (Schulte to Bühne, pers. obs.). All fires are presumed to have anthropogenic origin, either deliberate or accidental.

Data acquisition

Because the only detailed vegetation map for our study area was derived from ground-based information and satellite data collected in 2018 and 2019 (Lopes et al., 2020), we focused our analyses on two fire seasons, from October until April 2017–2018 and 2018–2019, respectively. To track fires, we used the burned area product MCD64A1 from the Moderate Resolution Imaging Spectroradiometer (MODIS) sensor (Giglio et al., 2018), which provides information on the last date at which a pixel has burned. This regularly updated product, which has a spatial resolution of 500 m and a daily temporal resolution, is widely used to do identify and map fires (Frantz et al., 2016) that are at least one MODIS pixel large (approximately 21 ha; Andela et al., 2019; Giglio et al., 2018). To extract information about fire spread and size from this burned area product, we first needed to identify individual fires. To do so, we used the following two principles: (1) a fire can have one or several starting points (ignition points), which are defined as areas that burned earlier than any contiguous burned areas; (2) a fire can spread from a starting point in one or more directions (see also Supplementary Materials, Part A). Based on the approach developed by Andela and colleagues (2019), we then applied the following rule to track fire progression: (1) burned areas adjacent to the ignition point are assumed to be part of the same fire if their burn date was close enough in time (defined as not later than twice the median temporal uncertainty of the new burned area and the ignition point combined); (2) other burned areas are added as long as they burned close enough in time; (3) each individual fire is spatially continuous (i.e. a fire ends when there are no other adjacent burned pixels that burned close enough in time). For each fire, the following parameters were measured: (1) maximum rate of spread (MROS, km day^{-1}) calculated as the maximum distance that a fire spread between two consecutive days (see Supplementary Materials, Part A) and (2) total area burned (in km^2). Following Hantson and colleagues (2015) and Tsela and colleagues (2014), all fires covering fewer than 6 pixels (approximately 1.25 km^2) were not considered for further analyses. We moreover only considered a fire for analysis if more than half of its final area fell inside Pendjari National Park.

We used the Tropical Applications of Meteorology using Satellite data and ground-based observations (TAMSAT) daily rainfall dataset (in mm day^{-1}; Maidment et al., 2017) to track changes in precipitation in our area. This dataset has a spatial resolution of 0.0375° (approx. 4 km) and covers the entire African continent from 1983 to present. From this dataset, we extracted for each observation of fire spread the sum of all precipitation in the week immediately prior to the observation. We also calculated the sum of all precipitation in the week prior to the start of each fire across the total area covered by each fire. Rainfall is highly seasonal in Pendjari National Park, with limited precipitation over the November to March period (Fig. 1). Since long-term variability in rainfall has previously been shown to affect fire dynamics, we considered month as a categorical predictive variable for both fire spread and fire size.

The land cover map developed by Lopes and colleagues (2020) was used to derive information on vegetation distribution in Pendjari. This vegetation map (used for both seasons) was created using a supervised land cover classification based on Sentinel 2 data (spatial resolution: 10 m) and was validated with field observations and very high-resolution satellite imagery. The map distinguishes between nine natural vegetation types ranging from grass savannahs to dense forest, as well as agricultural areas and bare grounds, with high overall accuracy (average F-score ca. 72%, ibidem). To test our hypotheses, we calculated the percentage of the area covered with grass savannah, shrub savannah, woodland savannah, tree savannah and dense forest for each fire spread observation and each individual fire. We also calculated the distance to any potential fire break, defined as roads, Pendjari river and any area that had previously burned within the same fire season for each fire spread observation and each individual fire. The course of the Pendjari river was digitized based on imagery available on Google Earth. Information on the distribution of roads were made available by Audrey Ipavec (pers. comm.). For analysis of MROS, the distance to the fire break was calculated as the minimum distance from the area for which the maximum rate of spread had been calculated to any fire break. For the fire size analysis, the smallest distance to any fire break was calculated for each potential starting point, and the median of these values was used, to reflect the average distance to fire breaks across the entire fire.

Statistical analyses

MROS was modelled using a generalized linear mixed model with a Gamma distribution; the identity of the fire season (either 2017/18 or 2018/19) was modelled as a random effect. Explanatory variables (defined for each area for which a MROS was calculated) included the percentages of all five vegetation types (namely, grass savannah, shrub savannah, woodland savannah, tree savannah and dense forest), the sum of rainfall during the preceding week, the month during which the observation was made, and distance (in metres) to the nearest fire break (which in this case refers to the distance to the nearest river, road or already burned area). Because woodland savannah and forests are relatively rare (mean cover of 2.9% and 1.6% respectively in Pendjari National Park), they were combined with the percentage tree savannah cover to form a general tree class. All pairs of continuous predictive variables had a correlation coefficient below 0.6 (see Table S1 in Supplementary Materials Part B).

Fire size was modelled using a linear mixed model with fire season as a random effect and the response variable being log-transformed. Explanatory variables included percentage of vegetation cover (as defined above), the sum of rainfall during the preceding week and the month during which the fire started. Forest, tree savannah and woodland savannah were again combined into a general tree class; all pairs of continuous predictive variables had a correlation coefficient below 0.7 (see Table S1 in Supplementary Materials Part B).

All geospatial and statistical analyses were carried out in R (version 3.2.5). Nonlinear relationships are common in ecology (Burkett et al., 2005) and fire ecology (Archibald et al., 2009). We therefore tested for non-linear relationships between our predictive variables and continuous explanatory variables such as rainfall using polynomial links. All continuous predictive variables were standardized (i.e. the mean was subtracted, and the values then divided by their standard deviation) before modelling; this means that the relative size of coefficient estimates can be used as proxy for effect size (Schielzeth, 2010). Variance inflation factors were calculated for both models to ensure that the coefficient estimates were not influenced by multicollinearity. We checked for spatial independence in the distribution of residuals of our best models by calculating Moran's I.

Results

129 fires were considered for analysis, comprising 1241 individual rate of spread measurements with a median rate of spread of 464 m per day and an average rate of spread of 1168 m per day. The maximum rate of spread ranged from 65.9 to 17 873 m day^{-1}. Fire sizes ranged from 1.28 to 359 km^2 with a median total burned area of 8.3 km^2. No correlation between fire size and MROS could be detected (Pearson's $r = 0.09$).

Our MROS model explained 16% of the variability in our observations, whereas the model for fire size explained 30% of the variation in our dataset. No significant spatial autocorrelation was detected in the residuals of the best model for MROS (observed Moran's $I = -0.001$, expected Moran's $I = -0.0008$, $P = 0.35$) and fire size (observed Moran's $I = -0.02$, expected Moran's $I = -0.008$, $P = 0.39$). All variance inflation factors were <1.5 for both models (see Table S2 in Supplementary Materials Part B), indicating no multicollinearity (Dormann et al., 2013).

Most of our expectations were met (Tables 1 and 2, Fig. 2). Specifically, fires in areas with a high percentage of highly combustible vegetation (i.e., vegetation types with low fuel load and bulk density, such as shrub savannah) spread faster (i.e., had a larger MROS), although this relationship was non-linear, with the importance of the percentage of shrub savannahs in determining fire spread rapidly levelling off as the percentage of shrub savannah in the area increased (Table 2, Fig. 2A). Fires occurring in areas with a higher proportion of less combustible vegetation (forests, tree and woodland savannahs) spread more slowly, as expected. Mean precipitation during the week preceding a given fire was shown to have a negative effect on spread rates, with drier conditions being associated with fires spreading faster. Fires also spread faster over the November to February period (early to mid-dry season) compared to October (late wet season) and to the end of the dry season (March/April). Finally, fires starting far from a potential fire breaker were shown to spread faster than fires near anthropogenic or natural barriers.

Table 2. Effect size and significance of fixed effects for the model describing the rate of fire spread (MROS) for the 2017/2018 and 2018/2019 fire seasons in Pendjari National Park.

Explanatory variable	Coefficient (residual standard error)	*T* value	*P* value
Intercept	7.23 (±0.1)	72.5	<0.01
Rainfall	−0.06 (±0.03)	−2.025	0.04
Percentage shrub savannah	4.01 (±1.01)	3.96	<0.01
Percentage shrub savannah^2	−12.05 (±1.02)	−11.78	<0.01
Percentage forest, tree and woodland savannah	−0.63 (±0.2)	−3.23	<0.01
Distance to nearest fire breaker	0.14 (±0.03)	4.60	<0.01
November	0.29 (±0.14)	2.11	0.03
December	0.25 (±0.15)	1.69	0.09
January	0.44 (±0.17)	2.64	0.01
February	0.38 (±0.22)	1.75	0.08
March	0.02 (±0.22)	0.11	0.01
April	−2.03 (±0.96)	−2.60	0.03

Estimates for months are provided as a difference between October and the month considered. A *P*-value less than 0.05 indicates that the explanatory variable significantly affects MROS.

Fire size was best modelled using information on rainfall during the preceding week and the percentage of shrub savannah within the area burnt by the fire (Table 3, Fig. 2B). Fires were larger in areas covered by a high percentage of shrub savannah vegetation. Higher amounts of rainfall in the week preceding the start of a fire led to bigger fires; this relationship was non-linear, with rainfall affecting fire size particularly at low ranges of rainfall (ca. 0–150 mm) during the preceding week of the fire, after which fire size declined slightly. Neither the proportion of grass, tree and woodland savannahs, or the month during which a fire started, significantly affected fire size.

Discussion

Several studies have shown that frequent observations from moderate-resolution, polar-orbiting satellites can be used to identify fire-affected pixels. These pixels can be

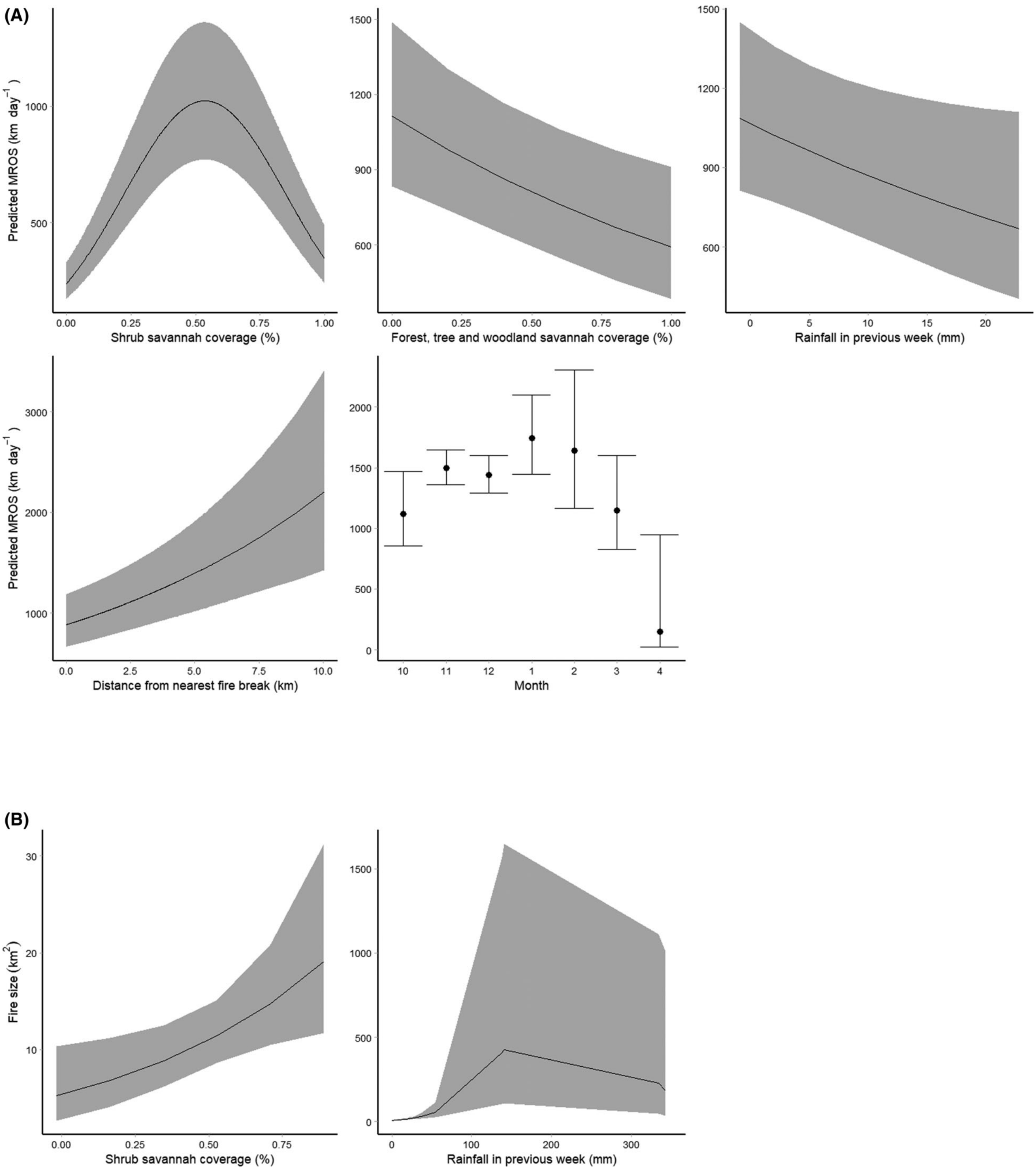

Figure 2. (A) Effects of significant predictors of fire spread (MRS). (B) Effects of significant predictors of fire size.

separated into clusters based on spatial and temporal proximity, and this information has been used to study the number and size distributions of individual fires (see e.g., Archibald & Roy, 2009; Hantson et al., 2015; Oom et al., 2016), fire shapes (Laurent et al., 2018; Nogueira et al., 2017) and the location of ignition points (Benali et al., 2016; Fusco et al., 2016). However, to our knowledge, this study represents the first attempt to identify both the biotic and abiotic factors shaping fire spread in an African savannah using satellite remote sensing

Table 3. Effect size and significance of fixed effects for the model describing fire size for the 2017/2018 and 2018/2019 fire seasons in Pendjari National Park.

Explanatory variable	Coefficient	*T* value	*P* value
Intercept	0.97 (±0.06)	17.34	<0.001
Rainfall	2.88 (±0.51)	5.66	<0.001
Rainfall^2	−1.80 (±0.51)	−3.55	<0.001
Percentage shrub savannah	0.11 (±0.04)	2.49	0.01

A *P* value less than 0.05 indicates that the explanatory variable significantly affects fire size.

information. The ranges we report for the maximum rate of spread are in line with estimates reported by Andela and colleagues (2019) in their global fire atlas, where they suggested an average rate of spread of 1 km per day over Benin and surroundings. Broadly, our results match expectations of fire size and spread being shaped by vegetation cover, rainfall and the distribution of fire breaks. However, they also illustrate the difficulties associated with fire spread and fire size predictions, with our best models accounting for less than a sixth to less than a third of the variability in our datasets.

An interesting pattern emerging from our analyses relates to the role of recent rainfall in shaping both fire size and spread in savannah ecosystems. Recent rainfall, that is, rainfall in the preceding week of a given fire, had an opposite effect on these two parameters, reducing the rate of fire spread but leading to larger fire size. The negative impact of recent rainfall on fire spread was expected, as reduced rainfall leads to drier vegetation, which is more ignitable (N'dri et al., 2018). The positive impact of recent rainfall on fire size was however unexpected and could be linked to the rapid growth of savannah vegetation in response to precipitation (Berry & Kulmatiski, 2017). Biomass accumulation of savannah grasses has indeed been shown to be closely linked to rainfall in the preceding week (Bonnet et al., 2010), meaning fuel could accumulate quickly after rainfall, which could positively affect fire size. This is supported by the observation that the positive effect of recent rainfall on fire size is most pronounced for small amounts of rainfall, which are observed throughout the dry season, where any additional rainfall could make the difference between additional growth and no additional growth (Bonnet et al., 2010).

The time of fire onset had diverging effects on fire size and fire spread. Fire size did not systematically vary across the fire season, contrasting with findings in other systems, where late season fires are often larger than early season fires (Archibald, 2016). This result may be an outcome of the rapid reduction in fuel amount and connectivity with relatively frequent fires early in the fire season (October/November). By contrast, the rate of fire spread was reduced in October and March/April. The former may be limited by vegetation moisture being overall still high, or declining unevenly across the landscape as rainfall declines, creating a mosaic of ignitable and not-ignitable vegetation. The latter may be an artefact of extremely reduced fuel availability, as much of the landscape will already have been burned. This discrepancy between seasonal changes of fire spread rates and fire size suggests that, although fire spread may be locally rapid throughout the middle of the dry season, other environmental factors (such as fire breaks and vegetation types) play an important role in limiting fire size.

Predictions of climate change are uncertain across many savannahs (IPCC, 2014), but our results show that fire spread rates and fire size may respond particularly strongly to changes in the distribution of short-term rainfall. Fewer weeks without any rainfall (for instance due to an increase in the length of the dry season) would likely lead to faster spreading fires, but a decrease in overall rainfall may eventually reduce the occurrence of large fires in the long-term as shrub savannahs contract (García Criado et al., 2020). By contrast, increases in the number of weeks with rainfall would likely reduce fire spread rates in the short-term, but would only benefit fire size if the cover of highly flammable vegetation (shrub savannah) increases (García Criado et al., 2020). A better understanding of future changes in rainfall regimes in fire-dominated ecosystems such as African savannahs will be key to informing future fire and wildlife management strategies.

The predominance of shrubs as a driver of both fire size and rate of fire spread in our study area was unexpected. Dry grasses as well as twigs and leaves dropped from shrubs become fine fuels, which burn easily; as such, a high coverage of shrubs and grass was expected to positively impact the rate of fire spread. Given the predominance of grasses as fuel for fires in tropical ecosystems (Simpson et al., 2016), the absence of a significant impact of grass savannah cover on fire size was surprising. Flammability as a vegetation property consists of several interdependent components (Anderson, 1970; Pausas et al., 2017), namely ignitability (the ease of ignition), combustibility (the intensity of combustion) and sustainability (the maintenance of burning over time). Our best model for the rate of fire spread confirms our expectations that shrubs have higher ignitability and/or combustibility than woodland and forests, and our fire size model may point towards shrubs better maintaining burning over time than other vegetation types. Sustainability can indeed be anticipated to influence fire duration and size, with vegetation burning for longer more likely to facilitate fires spreading in multiple directions, ultimately shaping fire size. As recognized by Simpson and colleagues (2016), little is currently known about interspecific differences in flammability of plant species. Given that plant flammability is a key determinant of fire behaviour, our study calls for more research on the relative sustainability of fires in grass and shrub vegetation, and its importance in shaping fire size in savannah ecosystems.

The approach presented in this contribution is associated with a number of methodological challenges and shortcomings. First, temporal uncertainty associated with the identification of the date at which a given pixel burned was highly variable, necessitating the identification of rules to reliably measure the maximum rate of fire spread. These rules are based on our current understanding of the behaviour of fires. However, they have not been validated using independent information (as independent validation data were unavailable for the study area and period considered). Such a validation process would particularly help when it comes to setting best practice standards for measuring maximum rates of fire spread using satellite information. Second, due to the limited availability of information on land cover distribution, we were only able to consider two fire seasons, reducing our sample size to 129 fires and constraining our ability to contrast years with very different climatic contexts. Third, a number of key factors known to influence fire dynamics, such as wind speed and direction, could not be taken into account, ultimately reducing the predictive power of our models. Despite these limitations, our study clearly provides a framework that can be replicated to different locations and contexts, where such data may become available. We therefore hope our study will trigger more research that use our approach to further explore how biotic and abiotic factors shape fire spread globally.

Fire is an important disturbance agent in savannahs (Caillault et al., 2015; Laris & Wardell, 2006) that promotes the coexistence of trees and grass (Sankaran et al., 2005) and modifies the structure and composition of vegetation communities (Devineau et al., 2010; Lehmann et al., 2014). Its effects on vegetation depend on the characteristics of local fire dynamics (Keeley et al., 2011), which can vary considerably in frequency, intensity, size and season (Archibald et al., 2013) and which ultimately shape the spatial configuration of the burned–unburned mosaic that characterizes many tropical ecosystems (Laris & Wardell, 2006). Altogether, our results highlight new avenues for satellite data to improve our understanding of fire dynamics in large, remote savannah ecosystems, and to improve our ability to predict fire spread, a key variable for wildlife management in the face of rapidly changing environmental conditions.

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Supporting Information

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

Figure S1. Single pixels or small groups of pixels surrounded by areas that burned on a later date were assigned to this later date if this was consistent with the data on temporal uncertainty (A).

Figure S2. Example for re-assigning the date of an area with the width of a single pixel that seems to separate two larger areas that burned on consecutive days.

Figure S3. Example for identifying the start and end point between which the path a fire travelled on one day is measured, which is the basis for calculating fire speed (A).

Figure S4. Example for calculating fire speed in cases where there are multiple potential starting points.

Table S1. Pearson correlation coefficient for candidate predictive variables explaining (A) the rate of fire spread (MROS) and (B) fire size.

Table S2. Variance Inflation Factors for predictive variables in the best models explaining rate of fire spread (A) and fire size (B).