

**RAINFALL, GEOLOGY AND LANDSCAPE POSITION GENERATE
LARGE-SCALE SPATIO-TEMPORAL FIRE PATTERN HETEROGENEITY
IN AN AFRICAN SAVANNA**

Izak P.J. Smit¹, Chris F. Smit², Navashni Govender¹, Mike van der Linde² & Sandra MacFadyen¹

¹ Scientific Services, South African National Parks, Private Bag X402, Skukuza, South Africa, 1350

² Department of Statistics, University of Pretoria, Private Bag X20, Pretoria, 0028

izak.smit@sanparks.org (corresponding author); Chris.Smit@up.ac.za;

navashni.govender@sanparks.org; Mike.VanderLinde@up.ac.za;

sandra.macfadyen@sanparks.org

ABSTRACT

Fire is considered a critical management tool in fire prone landscapes. Often studies and policies relating to fire focus on why and how the fire regime should be managed, often neglecting to subsequently evaluate management's ability to achieve these objectives over long temporal and large spatial scales. This study explores to what extent the long-term spatio-temporal fire patterns recorded in the Kruger National Park, South Africa has been influenced by management policies and to what extent it was dictated by underlying variability in the abiotic template. This was done using a spatially explicit fire-scar database from 1941 to 2006 across the 2 million hectare Park. Fire extent (hectares burnt per annum) (i) is correlated with rainfall cycles (ii)

exhibits no long-term trend and (iii) is largely non-responsive to prevailing fire management policies. Rainfall, geology and distance from the closest perennial river and the interactions between these variables influence large-scale fire pattern heterogeneity: areas with higher rainfall, on basaltic substrates and far from rivers are more fire prone and have less heterogeneous fire regimes than areas with lower rainfall, on granitic substrates and closer to rivers. This study is the first to illustrate that under a range of rainfall and geological conditions, perennial rivers influence long-term, landscape-scale fire patterns well beyond the riparian zone (typically up to 15 km from the river). It was concluded that despite fire management policies which historically aimed for largely homogeneous fire return regimes, spatially and temporally heterogeneous patterns have emerged. This is primarily because of differences in rainfall, geology and distance from perennial rivers. We postulate that large-scale spatio-temporal fire pattern heterogeneity is implicit to heterogeneous savannas, even under largely homogenizing fire policies. Management should be informed by these patterns, embracing the natural heterogeneity-producing template. We therefore suggest that management actions will be better directed when operating at appropriate scales, nested within the broader implicit landscape patterns, and when focusing on fire regime parameters over which they have more influence (e.g. fire season).

INTRODUCTION

Fire is an important driver of spatio-temporal patterns in savannas. Different fire regimes can result in diverse outcomes in vegetation biomass (Bond et al., 2005), composition (Whateley and Wills 1996) and structure (Higgins et al. 2007). Furthermore, fire frequency, fire intensity and season of fire interact to enable the

coexistence of trees and grass in savannas. Together with other factors like geology, rainfall and herbivory, fire widens the range of tree:grass ratios (Higgins et al., 2000; Anderies et al., 2002; Bond and Keeley, 2005; Sankarran et al., 2005; Nano and Clark, 2010). For example, it has been shown in the Kruger National Park (KNP) that landscapes that repeatedly experience frequent and high intensity fires have less woody cover than comparable landscapes continuously experiencing less frequent and low intensity fires (Smit et al., 2010). Fire regime characteristics also sometimes influence vegetation composition, with certain plant species persisting in frequently burned habitats (Silva et al., 1991; Canales et al., 1994) whereas long-unburned habitats create important refugia for fire sensitive species (Russell-Smith et al., 2002; Russell-Smith et al., 2012). The above suggests that both frequently and infrequently burned areas may be important for biodiversity conservation (Woinarski et al., 2004).

Fire management receives considerable attention in savanna regions. A search on Google Scholar with “savanna fire management” as keywords returned 45 600 hits (accessed 25 April 2012). Furthermore, most managed lands in fire prone regions, across different land uses like conservation and agriculture, have either a formal policy or at least an informal operational approach towards fire management for achieving their specific objectives. It was therefore of considerable interest to conservation managers when van Wilgen et al (2004) showed that *fire temporal patterns* in the “actively fire managed” Kruger National Park were primarily driven by rainfall of the preceding two years despite management policy. This triggered us to ask a follow-up question, namely whether KNP management can influence *fire spatial patterns* or whether this is also largely controlled by underlying abiotic variation of the Park. The main aim of our paper is therefore to explore the influence (or lack

thereof) of management on fire temporal and spatial patterns over the period 1941 to 2006 in the KNP. We do this by exploring how fire extent changed over time under different fire management regimes (temporal dimension) and how various fire frequency metrics correlate spatially with underlying abiotic variability (rainfall, geology and landscape position) (spatial dimension).

A second question we aim to address in this paper, closely linked to the question posed above, is whether a range of fire return metrics are expressed differently across the Park due to the variability in the environment. For example, does the maximum time observed between two successive fires between 1941 and 2006 have the same spatial pattern as mean fire return period, and do these patterns correlate with inherent underlying environmental variability? This is important since there is a growing realization that *variability* of fire regime (Martin and Sapsis, 1992) and *extreme fire events* (Hoffmann et al., 2009) may be equally if not more important in shaping savannas than the more regularly reported *average* conditions. For example, a prolonged fire free period which allows woody vegetation to grow out of the fire-trap, may be of greater importance for recruiting trees into taller height classes than a longer average fire return period (Hoffmann et al., 2009; Wakeling et al., 2011). Despite this, most studies still report on average/median fire return periods when in fact an extensive suite of fire return metrics may be needed to characterize fire return and the influence it may have on savannas. Therefore, in this study we calculate, in addition to the more regularly reported fire metrics (i.e. fire frequency, F_n ; mean fire return period, F_{mean} ; median fire return period, F_{med}) the (i) variability in fire return period (F_{stddev}), (ii) a representation of important rare events, i.e. the maximum recorded fire free period between 1941-2006 (F_{max}), and (iii) the uniqueness of the fire

history at each location compared to the fire history at all other locations (i.e. a heterogeneity index based on the Bray-Curtis dissimilarity index) (F_{BC}) (see Table 1 for a summary). We explore the spatial correlation between these different fire metrics in order to (i) assess the potential importance of considering these metrics additionally and/or separately when characterizing the fire return of an area, and (ii) to understand if they respond differently to the variable underlying abiotic environment.

MATERIAL AND METHODS

Study site

The KNP covers an area of almost 20 000km² in the north-eastern corner of South Africa, bordering Zimbabwe to the north and Mozambique to the east. The Park has a north-south rainfall gradient with average rainfall increasing from approximately 400mm per annum in the north to 700mm per annum in the southwest. The western part of the Park is predominantly underlain by granitic-derived soils and the east by soils of basaltic origin. This abiotic template gives rise to 35 main vegetation types (Gertenbach, 1983) dominated by *Acacia nigrescens*, *Sclerocarya birrea*, *Combretum imberbe*, *Combretum apiculatum*, *Terminalia sericea* and *Colophospermum mopane*.

Fire scar history: 1941 to 2006

Fire scars have been mapped in KNP since 1941, evolving from hand-drawn maps to satellite-derived fire-scar grids, increasing in spatial accuracy with associated technology advancement (N. Govender, submitted). The older maps were on-screen and tablet digitized (using a GTCO Super L II A0 Digitizing Tablet) enabling us to overlay all annual fire scar maps on top of each other within a Geographical Information System (GIS). As a result, a continuous fire scar history was derived for

each location in the Park (1941-2006). This spatially explicit fire scar history (i.e. burn or no-burn for each year) was then used to derive various fire return metrics for each location (Table 1). Note that we did not consider the month in which an area burnt, only the year, and therefore we based our analyses on metrics calculated for intervals of years (integer values) between subsequent fires.

The analysis of fire history data using intervals between successive fires may naturally result in incomplete information on fire return periods at the start and end of the time series. This problem of open-ended fire return periods at the start and end of a time series is commonly referred to as data censoring (Polakow and Dunne, 1999). Data censoring is not a problem when calculating F_n and F_{BC} , but presents a challenge for F_{max} , F_{mean} , F_{med} and F_{stdev} . Censoring at the start of the time series occurs when, for example, the first fire noted at a specific location is not year one of the data time series (1941 in this case) but any later year (e.g. 1945). Similarly, censoring at the end of the dataset occurs when the last fire noted is before 2006. Since we were working with an extended time series in a fire prone system, we expected the effects of data censoring to be small (with the exception of areas with very few fires). Therefore, in order to derive F_{max} , F_{mean} , F_{med} and F_{stdev} , we assumed a fire in the year before and in the year after the time series (1940 and 2007 respectively). This means that the estimated fire metrics are in most cases approximations and could be considered as *lower bounds*. For example, if F_{max} is reported as 12 years it should be interpreted as $F_{max} \geq 12$ years, in other words, the maximum fire return period between 1941 and 2006 is *at least* 12 years. We believe this is a reasonable way to treat data censoring in the context of the present study where the focus is more on the spatial patterns of the fire metrics than on the exact value of the metrics.

The dataset described above is arguably the best dataset available for exploring long-term spatio-temporal patterns over large landscape-scales in a fire-prone semi-arid African savanna. As such it afforded us a unique opportunity to study fire return across several climatic cycles, six different management policies and over a sufficiently diverse and large geographic area so that the effects of a rainfall gradient, contrasting geologies and variable landscape positions on fire return metrics could be assessed.

Abiotic drivers of variability - Rainfall, geology and landscape position as drivers of fire spatio-temporal patterns

We used a 1 km spaced point-grid as the basic unit of analyses. Each point location was assigned the relevant geology, distance from closest perennial river and long-term average annual rainfall. The long-term average annual rainfall for each location was based on a spline interpolation of rainfall measured at 22 rainfall stations scattered across the Park (each rainfall station had at least 18 records of annual rainfall). The values of the fire return metrics were calculated for polygons of unique fire history, derived by intersecting polygons generated by all fire records. Each point on the grid was assigned the fire return metrics of the associated polygon. If the point was located in a fire history polygon smaller than 1 hectare, it was excluded from the analysis as such small polygons were often due to mapping inaccuracies. This resulted in 18,315 regularly spaced points across the entire Park with known (i) geology, (ii) distance from closest perennial river, (iii) long-term average annual rainfall, and (iv) a calculated set of six fire return metrics.

Generalised linear modelling (GLM) with the fire metrics (F_n , F_{\max} , F_{mean} , F_{med} , F_{stdev} , F_{BC}) as dependent variables and geology, distance to closest perennial river and long-term average annual rainfall (with 1st and 2nd order interactions) as independent variables, was conducted on the 18,315 grid points spread across the entire Park. The unique additional contribution that each independent variable made towards explaining the fire metrics of interest was determined as the reduction in the adjusted R^2 value when comparing the full model (i.e. model including all independent variables) to the reduced model (i.e. model excluding the variable of interest).

Biotic drivers of and responders to fire variability: Herbaceous biomass and woody cover

Herbaceous biomass and woody vegetation cover are both drivers of, and responders to, fire patterns (e.g. van Wilgen et al., 2003; Murphy et al., 2010). Investigating how herbaceous biomass and woody cover patterns correlate with the fire patterns will provide insights into how herbaceous biomass and woody cover may be driving and/or responding to fire. Therefore, long-term (1989-2004) average herbaceous biomass (kg/ha) (described in Smit, 2011) and woody vegetation cover (%) (described in Bucini et al., 2010) were recorded for the point locations within the 1 km spaced grid mentioned above.

All statistical analyses were carried out using SAS® Version 9.2 and Statistica® Version 10, whilst GIS analyses were conducted using ESRI® ArcMap™ 9.2.

RESULTS

Long-term (1941-2006) fire metrics across Kruger National Park

The areas with highest F_n and lowest F_{max} , F_{mean} , F_{med} , and F_{BC} occur in the wetter regions of the Park (south-western granites), further from perennial rivers, and (mainly, but not exclusively) on the nutrient rich basaltic plains in the east (Fig. 1). Although the fire metrics are spatially correlated, each fire metric does contain information not explained by the other fire metrics (Table 2). F_{stddev} correlates the weakest with the other metrics, but all correlations are statistically significant.

Rainfall and fire temporal patterns

Using an extended time series (1941 to 2006) and a different analytical approach, this study concurs with the results presented by van Wilgen et al. (2004) (study period 1957 to 2001) who found that rainfall rather than fire management policy influenced the yearly extent of fires in the KNP. Total area burnt and average annual rainfall over KNP follow similar trajectories over time despite changing management regimes (Fig. 2). The data was smoothed using LOESS scatter plot smoothing (smoothing parameter = 0.1). The Pearson correlation coefficients between observed and smoothed area burnt and between observed and smoothed annual rainfall was 0.56 and 0.59 respectively. Furthermore, the correlation between area burnt and rainfall is 0.60 and 0.71 respectively for the observed and smoothed data respectively. Importantly, we detected no significant long-term trend of either increasing or decreasing fire extent (area burnt) or annual rainfall over the 66 years (p -value for burnt area trend line slope = 0.57; p -value for rainfall trend line slope = 0.24).

Rainfall, geology and landscape position as drivers of fire spatial patterns

Figs 3(a) to 3(f) illustrate how the various fire metrics change at increasing distance from the closest perennial river. It is clear that the fire metrics generally show a trend with increasing distance from the closest perennial river. F_n increased with increasing distance from the closest perennial river, whilst the other fire metrics decreased.

It is clear that fire spatial patterns are strongly related to geology, rainfall and landscape position (distance to the closest perennial river). Basaltic landscapes burn more frequently (Fig. 3(a)), at shorter between fire intervals (Figs 3(b) and 3(c)), with shorter maximum fire return periods (Fig. 3(d)), and have less variable and less dissimilar fire histories (Figs 3(e) and 3(f)) than granitic landscapes. Furthermore, areas with higher average annual rainfall generally burn more frequently (Fig.3(a)), at shorter between fire intervals (Figs 3(b) and 3(c)), with shorter maximum fire return periods (Fig. 3(d)), and have less variable and less dissimilar fire histories (Figs 3(e) and 3(f)) than areas with lower average annual rainfall.

The relationship between distance from rivers and fire patterns extends much further than the narrow riparian zone directly bordering the river. Depending on the fire metric of interest and the rainfall and geology under consideration, the gradient of fire metrics radiating from the rivers are often still clearly detectable up to 10-15 km from the closest perennial river, and sometimes even further than 20km (Figs 3(a) to 3(f)).

Contribution of abiotic variability of environmental variables towards explaining fire metrics

Generalised linear models with average annual rainfall, distance to closest perennial river and geology (and all interaction terms) were all highly significant ($p < 0.01$) and explained between 16% and 45% of the observed variation for the various fire return metrics (Table 3) (the remainder being random variation, possible mapping inaccuracies and variables not assessed in this study, for example herbivory, soil conditions, availability of ignition sources and other localised conditions). When considering the unique additional contribution of each environmental variable to the full model, it was apparent that the adjusted- R^2 dropped considerably if either average long-term annual rainfall or distance to closest perennial river was excluded from the full model (Table 3). Geology demonstrated less additional explanatory power. Dropping the four interaction terms one-by-one from the full model reduced the adjusted- R^2 by less than 0.05.

Herbaceous biomass and woody cover as biotic drivers of and responders to fire patterns

Spatial patterns of herbaceous biomass and woody cover are strongly related to geology, rainfall and landscape position. Basaltic landscapes are more productive in terms of average standing herbaceous biomass than granitic landscapes (Fig. 4). Furthermore, herbaceous biomass generally increases further away from rivers. Not surprisingly, areas with higher average annual rainfall have higher herbaceous biomass than areas with lower average annual rainfall (Fig. 4). These patterns in average herbaceous standing biomass (i.e. fuel load) may be partly responsible for

differences observed in fire patterns between the different rainfall regions, different geologies and different distances from the closest perennial river. However, caution should be exercised when interpreting the average herbaceous biomass as it is *yearly* herbaceous biomass rather than *average* herbaceous biomass that drive fire patterns. In the absence of spatially explicit data on yearly herbaceous biomass, we assumed that areas with higher long-term average herbaceous biomass would more likely have higher yearly biomass than areas with lower long-term average herbaceous biomass.

Granitic and higher rainfall landscapes are more wooded than basaltic and lower rainfall landscapes (Fig. 5). Furthermore, woody cover generally decreases further away from perennial rivers on the basaltic and lower rainfall landscapes, while this relationship holds only weakly on the granitic and higher rainfall landscapes (Fig. 5).

DISCUSSION

Across the world, especially in fire-prone savannas, many conservation managers have moved from a fixed-period, burn-block fire management regime, towards a more variable-period, patch-mosaic fire approach (Parr and Brockett 1999; Brockett et al., 2001; Bond and Archibald, 2003; Bilbao et al., 2010; Mulqueeny et al., 2010). Without knowledge of the most “appropriate” or “natural” fire regime, many conservation managers practise pyrodiversity as a kind of “insurance policy” (Keith et al., 2002; van Wilgen et al., 2003). This approach assumes that when uncertainty exists regarding the optimal/desirable fire regime, variable rather than homogeneous fire regimes are preferable as they ensure a range of outcomes. Furthermore, some researchers hypothesise that “pyrodiversity-begets-biodiversity” (Martin and Sapsis,

1992), although this is largely unproven (Parr and Andersen, 2006). Within the “pyrodiversity-begets-biodiversity” and a “pyrodiversity-insurance-policy” context, it would be of concern for KNP managers if spatio-temporal fire patterns were homogeneous across the KNP landscape. Consequently, the current KNP fire policy aims to stimulate a range of fire frequencies and intensities over space and time (van Wilgen et al., 2008). Here we illustrate that the KNP has indeed had a spatially and temporally heterogeneous fire history over the past six-and-a-half decades. We believe this is a significant and encouraging result considering the fact that many previous management policies were not specifically aimed at creating heterogeneity. In fact, the earlier policies were largely aimed at both temporal and spatial homogenisation, e.g. large rotational management blocks were burnt at regular intervals and in a specific season. Only the more recent policies have actively tried to encourage spatio-temporal variability in the fire regime. The results presented here suggest that the natural template – as defined by rainfall, geology and landscape position - creates desired heterogeneous fire regimes (at a landscape scale) despite management actions often managing/aiming for the contrary in the past. Our finding regarding the *inherent spatio-temporal heterogeneity* of fires in KNP complements the earlier finding of van Wilgen et al. (2004) regarding the *inherent temporal heterogeneity* in KNP fire regime. Van Wilgen et al. (2004) illustrated that management policy has little effect on the extent of fires, and noted that it is largely driven by rainfall (supported by Fig. 2 in this study). We therefore postulate that the natural abiotic template drives the large-scale spatio-temporal heterogeneity of KNP fire patterns, probably more so than any management policy/intervention will be able to do (especially considering the current abundance of natural and anthropogenic ignition sources). It is also interesting to note the big contribution perennial rivers

make towards fire heterogeneity, with distance from closest river being the variable explaining most of the observed variability in the “uniqueness/dissimilarity” (F_{BC}) of fire patterns (Table 3). Although this study focused on the larger perennial rivers, we postulate that smaller drainage lines also plays a role in generating fire pattern heterogeneity by creating discontinuities in the fuel loads and variability in fuel moisture content – this may be especially important for creating heterogeneity in the highly incised granitic landscapes.

It is well documented that large rivers (and other natural features like water bodies, rocky areas, etc.) can act as natural fire barriers (Baker, 1992). Furthermore, it has been noted that many riparian zones burn less frequently than the surrounding matrix due to patchiness of fuel loads and higher fuel moisture content (Dwire and Kauffman, 2003; Pettit and Naiman, 2007). However, here we show that rivers are related to fire patterns at much larger scales, far beyond the riparian/upland boundary. This result is particularly interesting as it is consistent across both geologies and rainfall subdivisions of KNP (also across different perennial river catchments – results not shown). We postulate four possible mechanisms, which may act individually or in combination, to create these patterns. Firstly, burning likelihood decreases the closer one gets to a perennial river (or any firebreak). In other words, a fire is more likely to originate and spread if it is located far from a firebreak, as opposed to locations close to firebreaks where protection from fire is afforded from one side. In addition, these areas of low fire frequency close to rivers are often wedged between a natural firebreak (perennial river) on the one side and an artificial firebreak (road) on the other side, essentially creating an area protected from fires originating from outside that area. Secondly, stream order typically increases closer to perennial rivers as drainage lines connect to form increasingly larger streams. Usually the higher the

stream order, the larger and “wetter” the river (i.e. draining larger areas) and the more effective it will be at reducing fire spread (Dwire and Kaufman, 2003). In other words, the closer one gets to perennial rivers, the more the landscape is incised by larger natural fire barriers, reducing fire spread. Thirdly, the reduction of fire frequency closer to rivers may be influenced by herbivory. Elephants, buffalo, impala and hippo, which form the bulk of the grazer biomass in KNP, have a decreasing density gradient further from rivers in the dry season (Smit et al., 2007; Smit and Ferreira, 2010). These herbivores may be partly responsible for the gradient of decreasing grass biomass observed closer to rivers (Fig. 4), resulting in reduced fuel loads and consequently reduced fires. And finally, reduced fire frequency closer to rivers may be due to topography. Fire spreads more readily up slopes rather than down slopes due to wetness gradients and upslope areas preheating prior to combustion, facilitating fire spread (Maingi and Henry, 2007). Considering that rivers are locally low lying areas and that elevation is strongly related to distance from perennial rivers, fire may often burn up slopes and therefore away from the rivers, resulting in drier upslope areas burning more frequently than moister downslope areas (Maingi and Henry, 2007). We postulate that all four reasons described above can, to a greater or lesser extent, interact to reduce fire occurrence closer to perennial rivers in KNP. These mechanisms may act in a positive feedback loop - fewer fires close to rivers result in increased woody vegetation (Fig. 5), which suppress herbaceous biomass (Fig. 4), which subsequently results in even fewer fires, reinforcing the feedback loop.

Local, regional and global pyrogeography models predict that under future climatic conditions some areas will experience increased fire activity, whereas other areas may

experience reduced fire activity (e.g. Balshi et al., 2008; Krawchuk et al., 2008; Krawchuk et al., 2009). This may be due to changing rainfall influencing fuel loads, or changing temperature, humidity and wind speed influencing fire danger index and consequently fire extent and intensity. Such changes in fire patterns due to climate change will in turn significantly influence local and global vegetation patterns (Bond et al., 2005; Staver et al., 2011). However, in the absence of long-term spatially explicit datasets, most pyrogeography studies rely heavily on modelling or extrapolation to understand how fire patterns may have changed or may change due to changing future climatic conditions. The dataset collated for this study afforded an opportunity to establish empirically whether fire extents have changed over the past almost seven decades in KNP. No directional trend has been observed in fire extent since 1941. This implies that any climate change that may have occurred in this region over the past seven decades has not exerted a directional change in annual burning extent across this two million hectare savanna landscape. However, we have not investigated whether changes in climatic conditions or changes in management policies have influenced other fire regime variables (e.g. fire season or fire intensity) (see e.g. van Wilgen et al., 2008).

Despite the growing realization that *variability* of fire regime (Martin and Sapsis, 1992) and *extreme fire events* (Hoffmann et al., 2009) may be equally if not more important in shaping savannas than *average* conditions, many studies still characterise fire regime using only the mean/median fire return period. However, a prolonged fire free period (represented by F_{\max}) which allows woody vegetation “windows of opportunity” to grow out of the fire-trap, may be of greater importance for recruiting trees into taller height classes than a longer average fire return period (Higgins et al.,

2000; Hoffmann et al., 2009). Therefore, since vegetation structure heterogeneity is often the result of variable disturbance regimes or extreme events in space and time, it is important to not only consider *average* conditions when describing heterogeneity of fire patterns (e.g. using F_{mean} , F_{median}), but also fire return metrics associated with *variability* (F_{stddev}), *dissimilarity* (F_{BC}) and *rare events* (e.g. F_{max}). It is evident from the results presented in Table 2 that a single measure of location (e.g. mean or median fire return period) does not characterise the fire return process fully - measures of spread, dissimilarity and extreme events must also be considered as it shapes savannas through different mechanisms (e.g. event and variability driven patterns).

MANAGEMENT IMPLICATIONS

Fire management objectives versus fire management outcomes

The fact that heterogeneous fire patterns emerged in KNP despite many years of fire policies largely aimed at homogenising spatio-temporal patterns, suggest that either (i) the policies were largely unattainable, or (ii) the managers were not executing the policy effectively. The fact that the emerging fire spatio-temporal patterns show such clear relationships with rainfall, geology and landscape position rather than with management policy or administratively defined management units, supports the former possibility. This indicates how important it is to not only have a fire policy in place, based on experimental or theoretical knowledge - or to indiscriminately transfer policies from one area to another - but to also evaluate the outcomes of policies. The aim should be to assess the practical relevance in a specific context and to determine management's operational ability to achieve the goals outlined in the policy.

Although the results presented here suggest that it is largely the inherent landscape variability rather than management policy and actions that influence the observed large-scale spatio-temporal fire scar patterns, it must be emphasized that managers may be able to (i) influence a range of other fire regime parameters than those considered in this study, and (ii) manipulate fire regimes at other scales than the scales considered here. For example, managers may be able to change the seasonality of fires by pre-empting late dry-season fires through igniting early dry-season fires (Price et al., 2012). Early season fires would result in more patchy (e.g. Werner, 2010) and lower intensity fires (e.g. Govender et al., 2006). A good example is provided by Price et al. (2012) for Western Arnhem Land in northern Australia where a changed fire management regime, which imposed prescribed early dry-season burning, was more effective in changing the fire season (reducing late dry-season fire extent from 29% to 12.5%) than the total fire extent (reduced from 38% to 30%). Furthermore, it has been demonstrated that managers can effectively manipulate fire return over smaller scales in the KNP (i.e. experimental burn plots – see Biggs et al. 2003) - however, this manipulation comes at large logistical costs (e.g. creating fire breaks; fire suppression efforts). Therefore, although managers do have some influence on fire, the influence may not be at the scales and on the fire variables they aim to manage (e.g. large-scale fire frequency). We postulate that fire policies and management actions will be more successful if it focuses on fire metrics and at scales where management actions are more likely to be effective.

Furthermore, it must also be remembered that management's ability to influence fire regimes is dependent on local context. For example, managers' ability to influence fire regimes will, *inter alia*, be dependent on the localised rainfall patterns, the

prevalence of obstructions to fire spread (e.g. roads, firebreaks, rivers), the level of human impact in the area, the number of managers responsible per land area and the resources available to them and, very importantly, the number of (natural and accidental) ignition sources (see e.g. Archibald et al., 2010).

Heterogeneity on multiple scales

Although we believe our results of heterogenous fire patterns will be positively received by Park Management in the context of the current patch-mosaic burning policy, fire pattern heterogeneity must be considered at multiple scales (Rogers, 2003). Here we have shown that at the park-wide scale the natural template gave rise to variable fire return histories, in spite of past management actions. However, when examined at smaller scales, homogenisation may occur, e.g. basaltic plains far from rivers may be homogenised by frequent fires, as may granitic areas close to rivers due to a lack of fires. Managers may therefore decide to actively increase heterogeneity at these smaller scales, e.g. by reducing fire frequency and intensity on sections of the basaltic plains, and creating high intensity fires (so-called “fire storms”) in specific granitic areas. Although concerted management actions may increase heterogeneity at the smaller scale, it can be argued that heterogeneity created by actively “opposing” the natural template/patterning, may be unnatural and hence undesirable. This relates to questions of how much heterogeneity is needed and at what scales – and more fundamentally, whether heterogeneity of fire patterns should be the objective at all (Parr and Andersen, 2006).

Monitoring framework

The results presented here provide a useful spatial framework for monitoring fire effects, illustrating that fire is a more prominent driver in areas far from rivers and on basaltic landscapes (“fire dominated systems”) than on granitic landscapes and areas close to rivers where other processes, e.g. herbivory, may be a more important disturbance factor (e.g. elephants, as illustrated by Smit and Ferreira, 2010). Since fire and elephants alone are less likely to kill large trees than their combined and interacting effects (Trollope et al., 1998; Holdo, 2007, Vanak et al., 2011), large trees may be, at a landscape scale, more susceptible to mortality at intermediate distances from perennial rivers. In other words, managers need to ensure that sites being monitored are located at different distances from rivers, both on the basalts and the granites, as ecological drivers like fire and herbivory operate at different intensities across these gradients. Usually monitoring programmes in KNP focus on the representativeness of vegetation types, which is nested within geology and rainfall regions, but do not take distance from rivers explicitly into account.

Defining the template driving large-scale patterns in KNP

Managers and scientists often subdivide the Park into four main abiotic landscapes. This delineation is based on the north-south rainfall gradient and the main geological substrates, giving rise to four landscapes namely *high rainfall granites*, *high rainfall basalts*, *low rainfall granites* and *low rainfall basalts* (see e.g. Grant et al., 1995; Codron et al., 2005). Here we show that there is a crucial third variable namely, a gradient in distance from the closest perennial river, that needs to be considered. Many different drivers (fire {Figs 3(a) to 3(f)}; herbivores {Smit et al., 2007; Smit and Ferreira, 2010}) and responders (herbaceous biomass {Fig. 4}; woody cover

{Fig. 5}) are influenced by the “distance from the closest perennial river” gradient *within* the rainfall and geology template. Since both drivers and responders exhibit strong directional trends on the “distance from closest perennial river” gradient, it needs to be explicitly taken into account in future when management and monitoring landscapes are defined.

Delineation of Park into fire management zones

Past KNP fire policies promoted a single management approach across most of the park (see van Wilgen et al., 2003 for an overview). However, it is now recognised that a single fire management approach for the entire park is not appropriate. Fire can be used more strategically since fire presents diverse threats - and opportunities - in different parts of the park. For example, lower intensity and less frequent fires can be used to counter loss of tall trees on basalts, whereas high intensity late dry-season fires can be used to curb bush encroachment on granites (Eckhard et al., 2000). Consequently, a new fire policy is currently in review (proposed for implementation 2012) which will replace the single management approach with a variable approach, tailored for ecologically defined fire-management zones. The new proposed fire policy uses F_{mean} , presented in this paper, in combination with geology and long-term rainfall, to delineate fire management zones. Fire management actions and monitoring will vary between the zones based on the objectives and concerns specific to each area.

CONCLUSIONS

Long-term, landscape-scale spatio-temporal fire patterns in the KNP are heterogeneous and are strongly influenced by the underlying abiotic template as

defined by rainfall, geology and landscape position. Smaller-scale and localized processes will act within these broad, abiotically-controlled patterns. We therefore argue that large-scale spatio-temporal heterogeneity of fire patterns is implicit at the park-wide level, notwithstanding management policy, with management actions best directed at embracing these patterns and influencing outcomes at appropriate scales and focusing on appropriate fire metrics (e.g. seasonality) within the larger template of heterogeneity.

ACKNOWLEDGEMENTS

We acknowledge all the scientists and rangers who have collected fire scar data from the 1940s until today in the KNP. Brian van Wilgen (CSIR) is thanked for discussion and inputs on an earlier draft that has benefited this manuscript. Jeremy Russell-Smith and another anonymous reviewer are thanked for their inputs and suggestions that have considerably improved the manuscript. We acknowledge SANParks for making this dataset available for this study and Ms J Sommerville (University of Pretoria) for help with the SAS programming.

REFERENCES

- Anderies, J.M., et al. 2002. Grazing management, resilience, and the dynamics of a fire-driven rangeland system. - *Ecosystems* 5: 23-44.
- Archibald, S. et al. 2010. Climate and the inter-annual variability of fire in southern Africa: a meta-analysis using long-term field data and satellite-derived burnt area data. - *Glob. Ecol. Biogeogr.*, 19: 794-809.

- Baker, W.L. 1992. The landscape ecology of large disturbances in the design and management of nature reserves. - *Landsc. Ecol.* 7: 181-194.
- Balshi M. S. et al. 2008. Assessing the response of area burned to changing climate in western boreal North America using a Multivariate Adaptive Regression Splines (MARS) approach. - *Global Change Biology* 14: 1–23.
- Biggs, R. et al. 2003. Experimental burn plot trial in the Kruger National Park: history, experimental design and suggestions for data analysis. - *Koedoe* 46: 1–15.
- Bilbao B.A et al. 2010. Indigenous use of fire and forest loss in Canaima National Park, Venezuela. Assessment of and tools for alternative strategies of fire management in Pemon Indigenous Lands. – *Human Ecology* 38: 663-673.
- Bond, W.J. et al. 2005. The global distribution of ecosystems in a world without fire. - *New Phytol.* 165: 525-538.
- Bond, W.J. and Keeley, J.E. 2005. Fire as global “herbivore”: the ecology and evolution of flammable ecosystems. - *Trends Ecol. Evol.* 20: 387-394.
- Bond, W.J. and Archibald, S. 2003. Confronting complexity: fire policy choices in South African savanna parks. - *Int. J. Wildland Fire* 12: 381-389.
- Brockett, B.H. et al. 2001. A patch mosaic burning system for conservation areas in southern Africa. - *Int. J. Wildland Fire* 10: 169–183.
- Bucini, G. et al. 2010. Woody fractional cover in Kruger National Park, South Africa: remote-sensing-based maps and ecological insights. – In: Hill, M.J. and Hanan, N.P. (eds), *Ecosystem function in savannas: measurement and modeling at landscape to global scales.* CRC/Taylor and Francis, pp. 219-237.
- Canales, J. et al. 1994. A Demographic Study of an Annual Grass (*Andropogon brevifolius* Schwartz) in Burnt and Unburnt Savanna. - *Acta Oecol.* 15: 261-273.

- Codron, J. et al. 2005. Taxonomic, anatomical, and spatio-temporal variations in the stable carbon and nitrogen isotopic composition of plants from an African savanna. - *J. Archaeol. Sci.* 32: 1757-1772.
- Dwire, K.A. and Kauffman, J.B. 2003. Fire and riparian ecosystems in landscapes of the western USA. - *For. Ecol. Manag.* 178: 61-74.
- Eckhardt, H.C. et al. 2000. Trends in woody vegetation cover in the Kruger National Park, South Africa, between 1940 and 1998. - *Afr. J. Ecol.* 38: 108-115.
- Gertenbach, W.P.D. 1983. Landscapes of the Kruger National Park. - *Koedoe* 26: 9-121.
- Govender N. et al. 2006. The effect of fire season, fire frequency, rainfall and management on fire intensities in savanna vegetation in South Africa. *J. Appl. Ecol.* 43: 748–758.
- Grant, C.C. et al. 1995. The nutritive value of veld as indicated by faecal phosphorus and nitrogen and its relation to the condition and movement of prominent ruminants during the 1992-1993 drought in the Kruger National Park. - *Koedoe* 38: 17-31.
- Higgins, S.I. et al. 2000. Fire, resprouting and variability: a recipe for grass–tree coexistence in savanna. - *J. Ecol.* 88: 213–229.
- Higgins, S.I. et al. 2007. Effects of four decades of fire manipulation on woody vegetation structure in savanna. - *Ecology* 88: 1119–1125.
- Hoffmann, W.A. et al. 2009. Tree topkill, not mortality, governs the dynamics of savanna-forest boundaries under frequent fire in central Brazil. - *Ecology* 90: 1326-1337.
- Holdo, R.M. 2007. Elephants, fire, and frost can determine community structure and composition in Kalahari woodlands. - *Ecol. Appl.* 17: 558-568.

- Keith, D.A. et al. 2002. Fire management and biodiversity conservation: key approaches and principles. – In: Bradstock, R.A. et al. (eds), *Flammable Australia: The Fire Regimes and Biodiversity of a Continent*. Cambridge Univ. Press, pp. 401-428.
- Krawchuk M.A. et al. 2009. Global Pyrogeography: the Current and Future Distribution of Wildfire. - *PLoS ONE* 4: e5102. doi:10.1371/journal.pone.0005102
- Krawchuk M.A. et al. 2008. Predicted changes in fire weather suggest increases in lightning fire initiation and future area burned in the mixedwood boreal forest. - *Clim. Change* 92: 83-97.
- Maingi, J.K. and Henry, M.C. 2007. Factors influencing wildfire occurrence and distribution in eastern Kentucky, USA. - *Int. J. Wildland Fire* 16: 23-33.
- Martin, R.E. and Sapsis, D.B. 1992. Fires as agents of biodiversity: pyrodiversity promotes biodiversity. In: Kerner, H.M. (ed), *Proceedings of the symposium on biodiversity in northwestern California, 1991*. Univ. of California, pp. 150-157.
- Mulqueeney, C.M. et al. 2010. Landscape-level differences in fire regime between block and patch-mosaic burning strategies in Mkuzi Game Reserve, South Africa. – *Afr. J. Range For. Sci.* 27: 143-150.
- Murphy, B.P. et al. 2010. Using generalized autoregressive error models to understand fire-vegetation-soil feedbacks in a mulga-spinifex landscape mosaic. - *J. Biogeogr.* 37: 2169-2182.
- Nano, C.E.M and Clarke, P.J. 2010. Woody-grass ratios in a grassy arid system are limited by multi-causal interactions of abiotic constraints, competition and fire. – *Oecologia* 162: 719-732.

- Parr, C.L. and Andersen, A.N. 2006. Patch Mosaic Burning for Biodiversity Conservation: a Critique of the Pyrodiversity Paradigm. - *Conserv. Biol.* 20: 1610-1619.
- Parr, C.L. and Brockett, B.H. 1999. Patch-mosaic burning: a new paradigm for savanna fire management in protected areas? - *Koedoe* 42: 117–130.
- Pettit, N.E. and Naiman, R.J. 2007. Fire in the Riparian Zone: Characteristics and Ecological Consequences. - *Ecosystems* 10: 673-687.
- Polakow, D.A. and Dunne, T.T. 1999. Modelling fire-return interval T: stochasticity and censoring in the two-parameter Weibull model. - *Ecol. Model.* 121: 79-102.
- Price et al. 2012. The influence of prescribed fire on the extent of wildfire in savanna landscapes of western Arnhem Land, Australia. – *Int. J. Wildland Fire* doi 10.1071/WF10079.
- Rogers, K.H. 2003. Adopting a Heterogeneity Paradigm: Implications for Management of Protected Savannas. – In: du Toit, K.H. et al. (eds), *The Kruger Experience: Ecology and management of savanna heterogeneity*. Island Press, pp. 41-58.
- Russell-Smith, J. et al. 2002. Fire regimes and the conservation of sandstone heath in monsoonal northern Australia: frequency, interval and patchiness. - *Biol. Conserv.* 104: 91– 106.
- Russell-Smith, J. et al. 2012. Simplifying the savanna: the trajectory of fire-sensitive vegetation mosaics in northern Australia. *Journal of Biogeography*. doi: 10.1111/j.1365-2699.2012.02679.x
- Sankaran, M. et al. 2005. Determinants of woody cover in African savannas. - *Nature* 438: 846–849.

- Silva, J.F. et al. 1991. Population responses to fire in a tropical savanna grass: a matrix model approach. - *J. Ecol.* 79: 345-356.
- Smit, I.P.J. 2011. Resources driving landscape-scale distribution patterns of grazers in an African savanna. - *Ecography* 34: 67-74.
- Smit, I.P.J. et al., 2010. Effects of fire on woody vegetation structure in African savanna. - *Ecol. Appl.* 20: 1865-1875.
- Smit, I.P.J. and Ferreira, S.M. 2010. Management intervention affects river-bound spatial dynamics of elephants. - *Biol. Conserv.* 143: 2172-2181.
- Smit, I.P.J. et al., 2007. Do artificial waterholes influence the way herbivores use the landscape? Herbivore distribution patterns around rivers and artificial surface water sources in a large African savanna park. - *Biol. Conserv.* 136: 85-99.
- Staver, A.C. et al., 2011. The global extent and determinants of savanna and forest as alternative biome states. – *Science* 334: 230-232.
- Trollope, W.S.W. et al. 1998. Long-term changes in the woody vegetation of the Kruger National Park, with special reference to the effects of elephants and fire. *Koedoe* 41: 103-112.
- Vanak, A.T. et al. in press. Biocomplexity in large tree mortality: interactions between elephant, fire and landscape in an African savanna. - *Ecography*
- van Wilgen, B.W. et al. 2003. Fire as a driver of ecosystem variability. – In: du Toit, K.H. et al. (eds), *The Kruger Experience: Ecology and management of savanna heterogeneity*. Island Press, pp. 149-170.
- van Wilgen, B.W. et al. 2004. Response of savanna fire regimes to changing fire management policies in a large African national park. - *Conserv. Biol.* 18: 1533–1540.

- van Wilgen, B.W. et al. 2008. An assessment of the implementation and outcomes of recent changes to fire management in the Kruger National Park. - *Koedoe* 50: 22-31.
- Wakeling, J.L. et al. 2011 Simply the best: the transition of savanna saplings to trees. – *Oikos* 120: 1448-1451.
- Werner, P.A. 2010. Fine-scale patchiness of burns in a mesic eucalypt savanna differs with fire season and sorghum abundance. – *Northern Territory Naturalist* 22: 31-44.
- Whateley, A.M. and Wills, A.J. 1996. Colonization of a sub-tropical woodland by forest trees in South Africa. - *Lammergeyer* 44: 19-30.
- Woinarski, J.C.Z. et al. 2004. Response of vegetation and vertebrate fauna to 23 years of fire exclusion in a tropical *Eucalyptus* open forest, Northern Territory, Australia. - *Austral. Ecol.* 29: 156– 176.

Table 1: Fire metrics derived for Kruger National Park based on fire scars from 1941 to 2006.

Fire parameter	Abbreviation	Explanation
Number of fires	F_n	Total number of fires observed over full period
Maximum fire return period	F_{max}	Longest time period (in years) between successive fires
Mean fire return period	F_{mean}	Average number of years between successive fires
Median fire return period	F_{med}	Median number of years between successive fires
Standard deviation of fire return period	F_{stdev}	Standard deviation of numbers of years between successive fires
Bray-Curtis Dissimilarity*	F_{BC}	Bray-Curtis Dissimilarity Index between fire history at one specific location and all other locations

* This value varies between 0 and 1, with values closer to 1 indicating that a location has a very unique fire history compared to the rest of the Park and values closer to 0 indicating it has a more frequently occurring fire history.

Table 2: Pearson correlation coefficients between the different fire metrics (Note: All correlations are statistically significant with p -value < 0.0001 , $n = 18\ 315$).

	F_n	F_{mean}	F_{max}	F_{median}	F_{stdev}	F_{BC}
F_n	1.00	-0.60	-0.71	-0.54	-0.65	-0.77
F_{mean}		1.00	0.83	0.98	0.31	0.73
F_{max}			1.00	0.76	0.68	0.74
F_{median}				1.00	0.21	0.69
F_{stdev}					1.00	0.57
F_{BC}						1.00

Table 3: Adjusted- R^2 values for Generalised Linear Models with fire return metrics as dependent variables and geology, long-term average annual rainfall and distance from closest perennial river (and all interactions) as independent variables (row 1), as well as models where different variables were dropped from the full model (rows 2-4). The bigger the decrease in the adjusted R^2 when dropping a variable and its interactions, the more important that variable is in explaining the variation.

	F_n	F_{mean}	F_{med}	F_{max}	F_{stdev}	F_{BC}
Full model	0.45	0.21	0.16	0.27	0.25	0.30
Geology (and interactions) dropped from full model	0.35	0.18	0.13	0.23	0.22	0.23
Rainfall (and interactions) dropped from full model	0.16	0.09	0.07	0.12	0.12	0.24
Distance to river (and interactions) dropped from full model	0.31	0.09	0.07	0.13	0.10	0.10

Fig. 1: Fire history metrics for the Kruger National Park between 1941 and 2006.

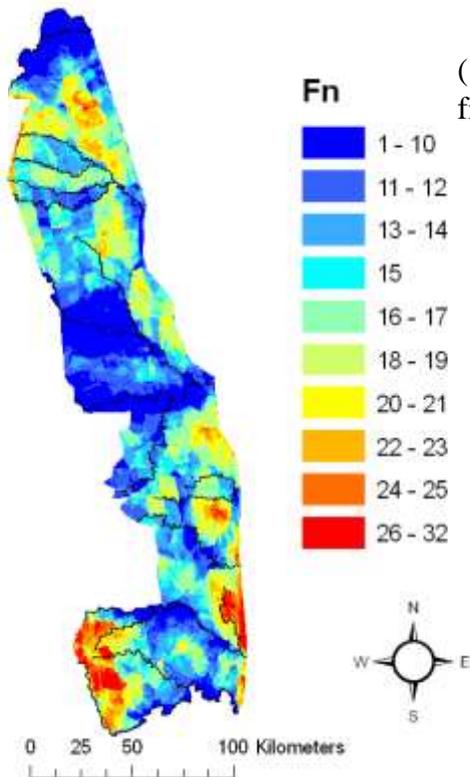
Fig. 2: Area burnt and rainfall between 1941 and 2006 across the entire Kruger National Park (LOESS smoothed, smoothing parameter = 0.1) See van Wilgen *et al.* (2003 and 2004) for more detail on the various fire management regimes.

Figs. 3: Average of each fire history metric at 1km incremental distance bands from perennial rivers in the Kruger National Park. Lower rainfall was defined as areas with recorded rainfall less than the park-wide average, whereas high rainfall areas were defined as areas with rainfall more than the park-wide average (F_{mean} and F_{med} were double log-transformed to improve readability of the graphs).

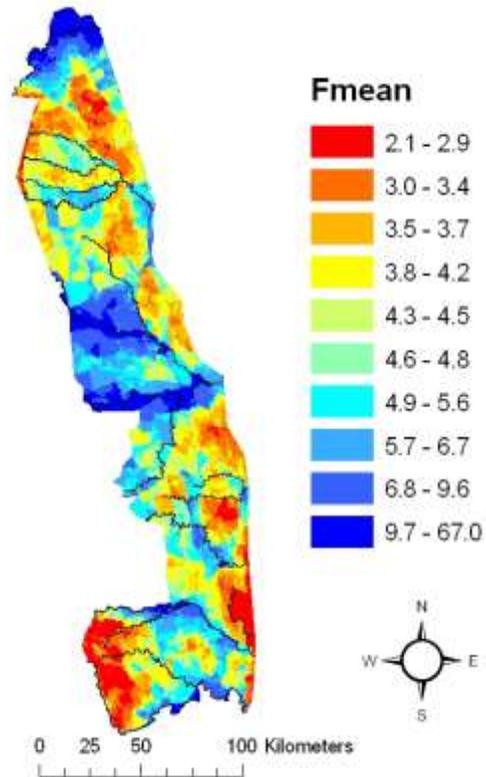
Fig. 4: Long-term average herbaceous biomass at increasing distance from perennial rivers.

Fig. 5: Average percentage woody cover at increasing distance from perennial rivers.

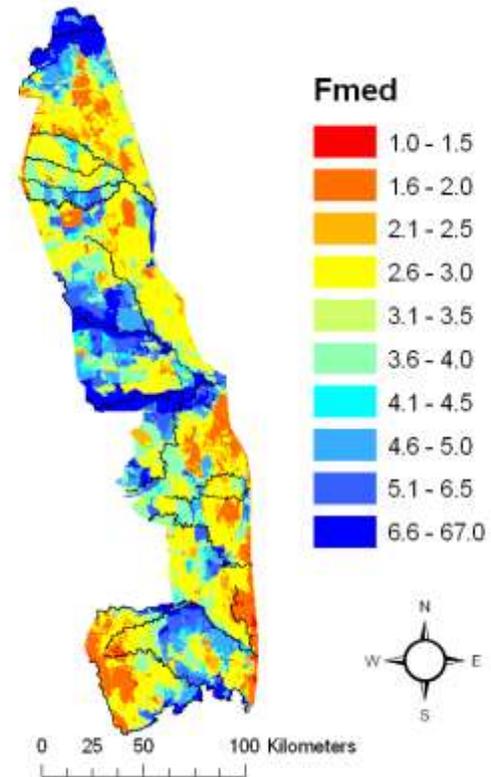
(1a) Fire frequency
(1941 to 2006)



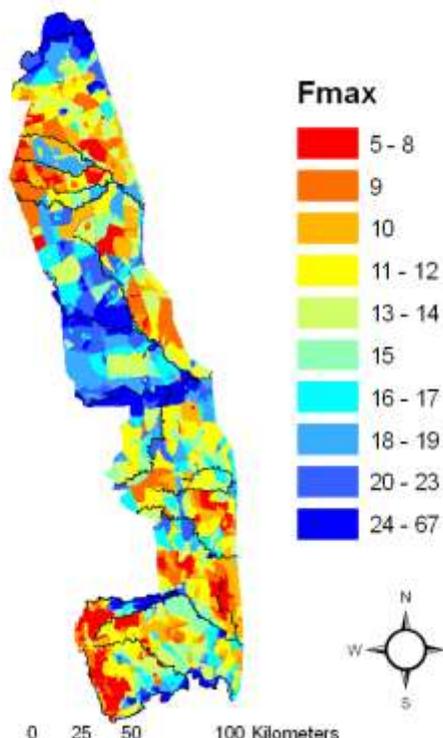
(1b) Mean
fire interval



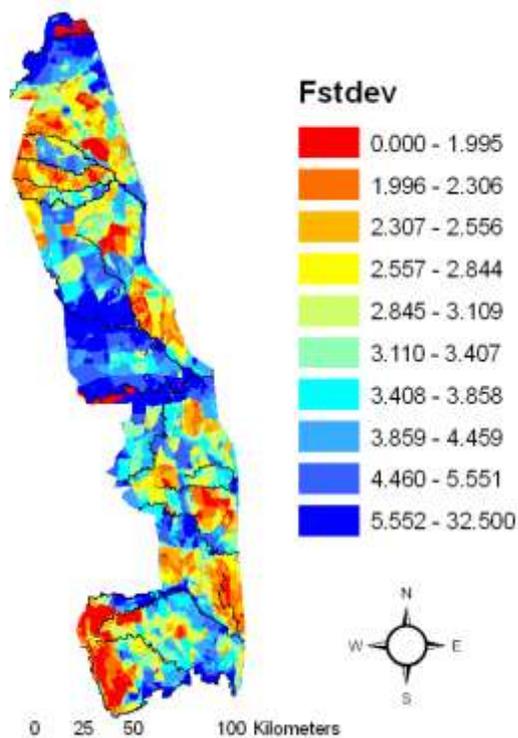
(1c) Median
fire interval



(1d) Maximum
fire interval



(1e) Standard
deviation of
fire interval



(1f) Bray-
Curtis index
for fire history

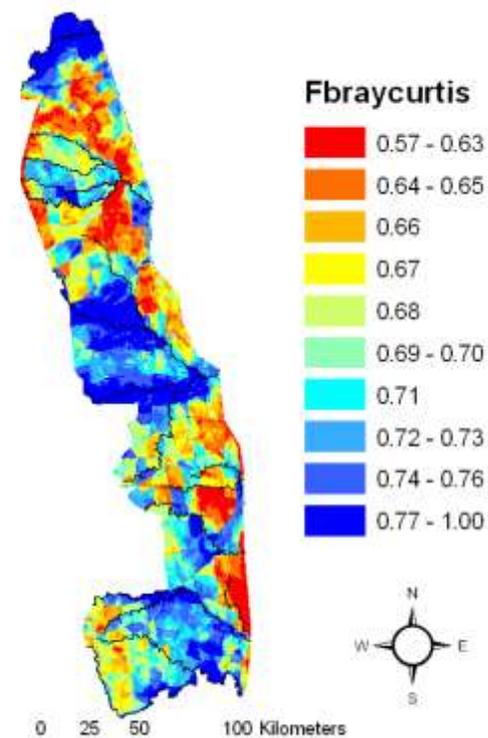


Fig. 2

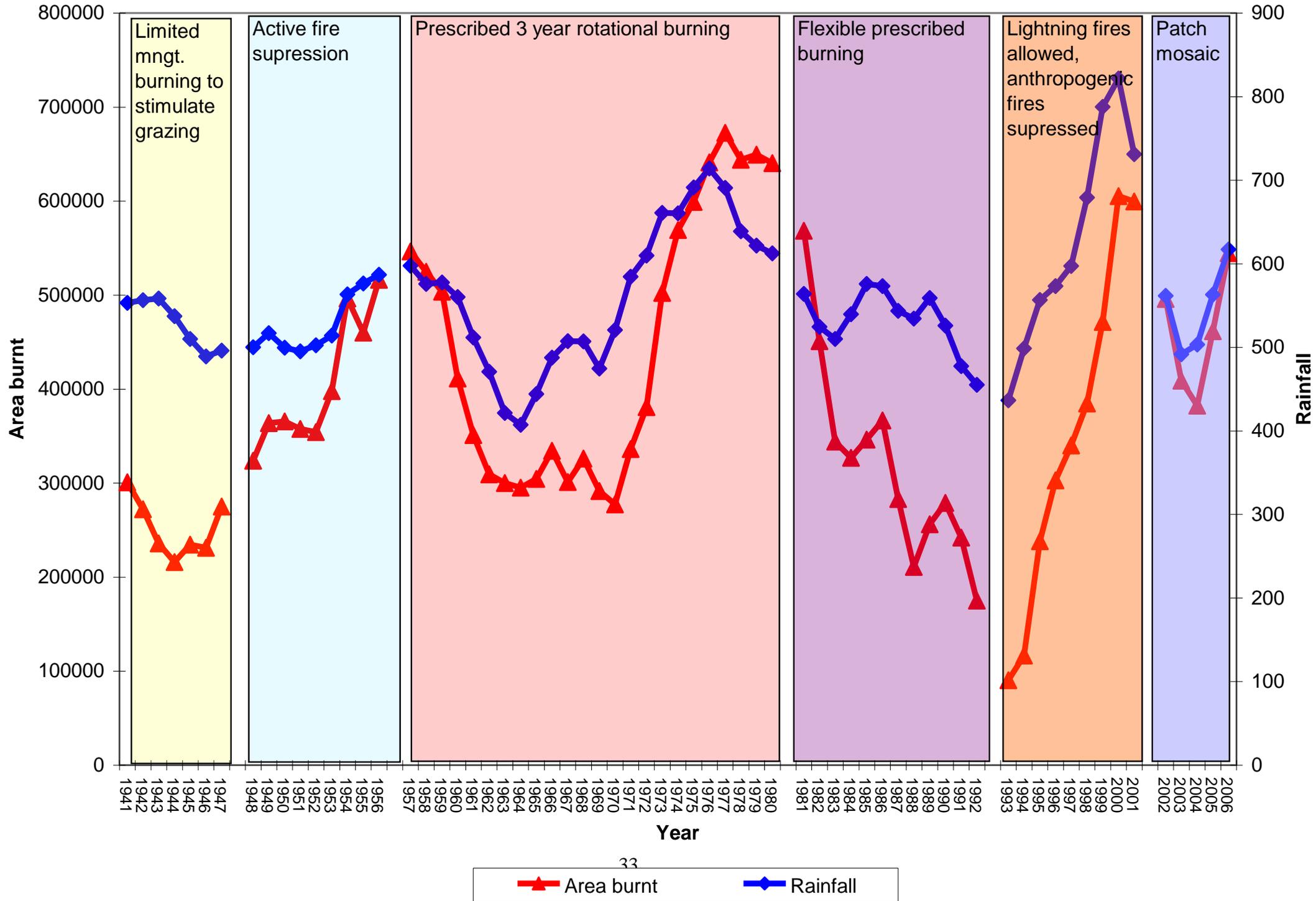


Fig. 3(a)

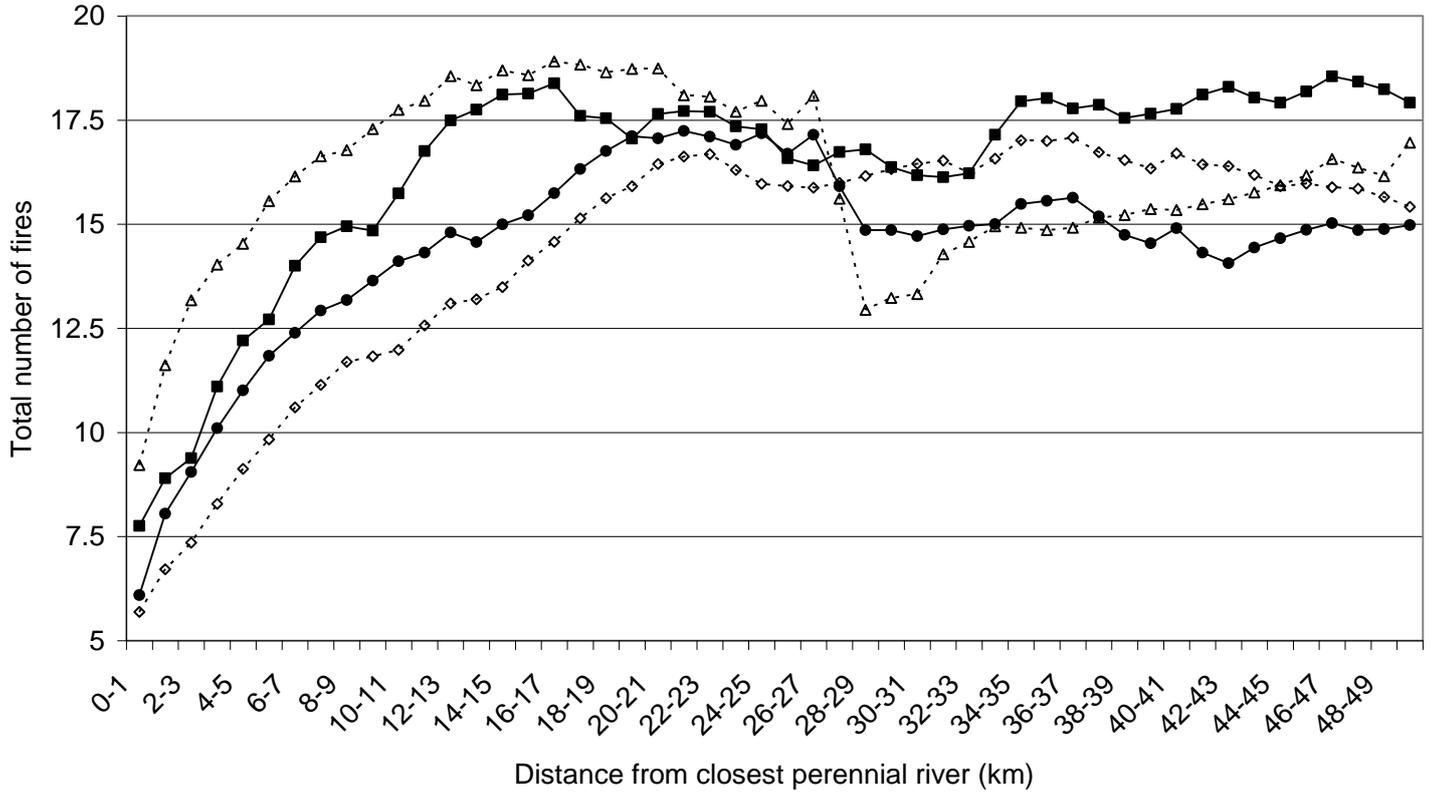


Fig. 3(b)

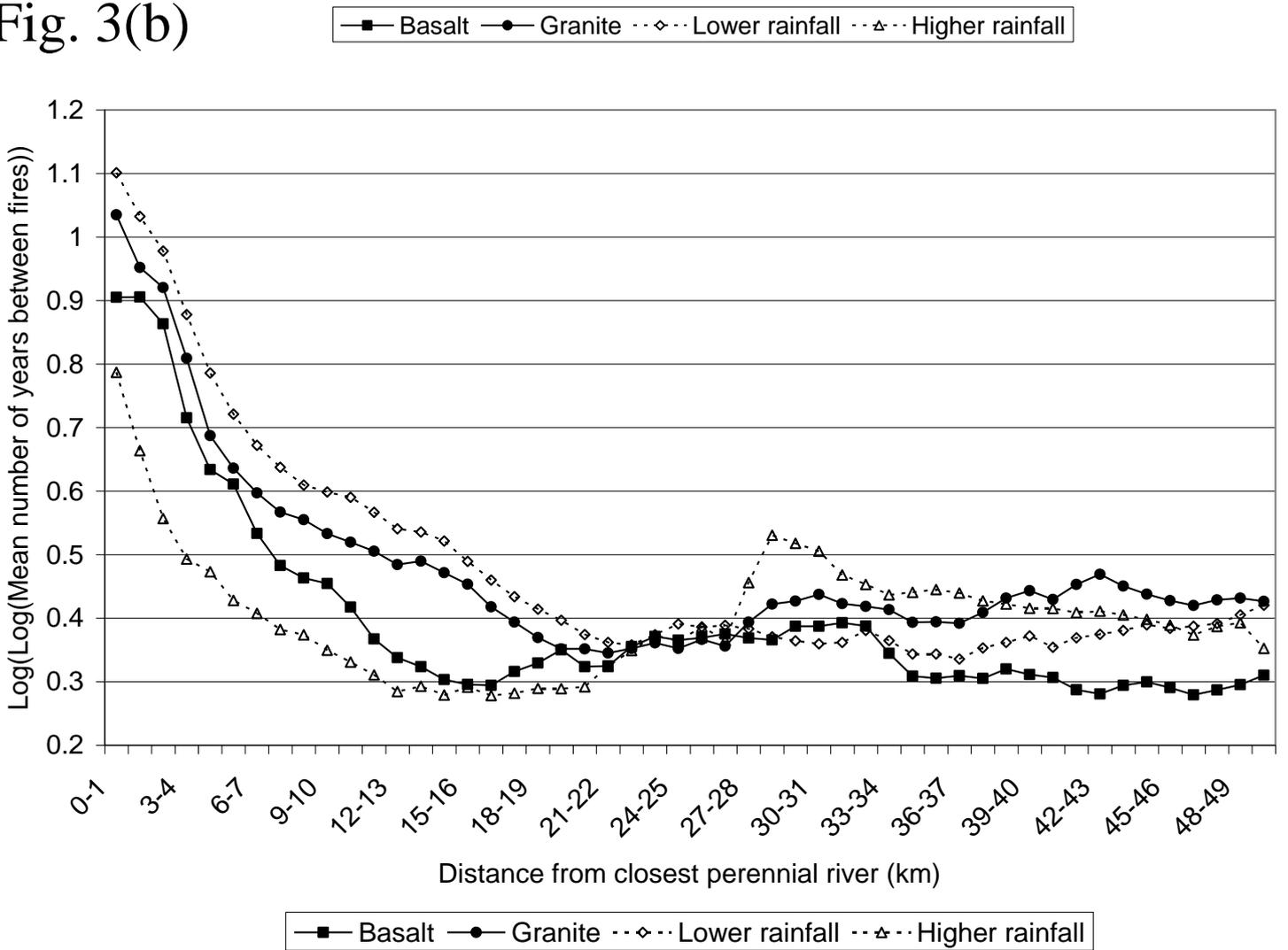


Fig. 3(c)

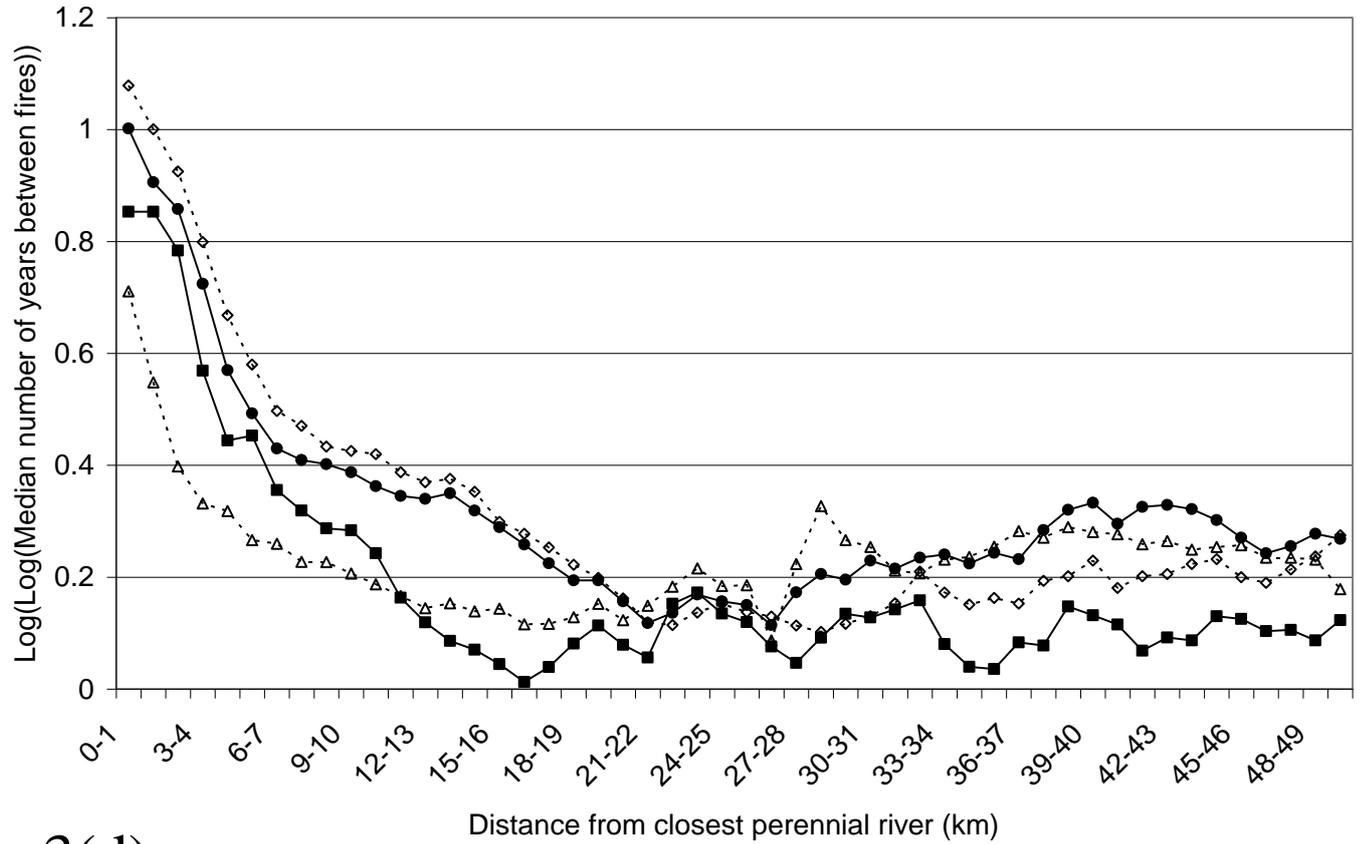


Fig. 3(d)

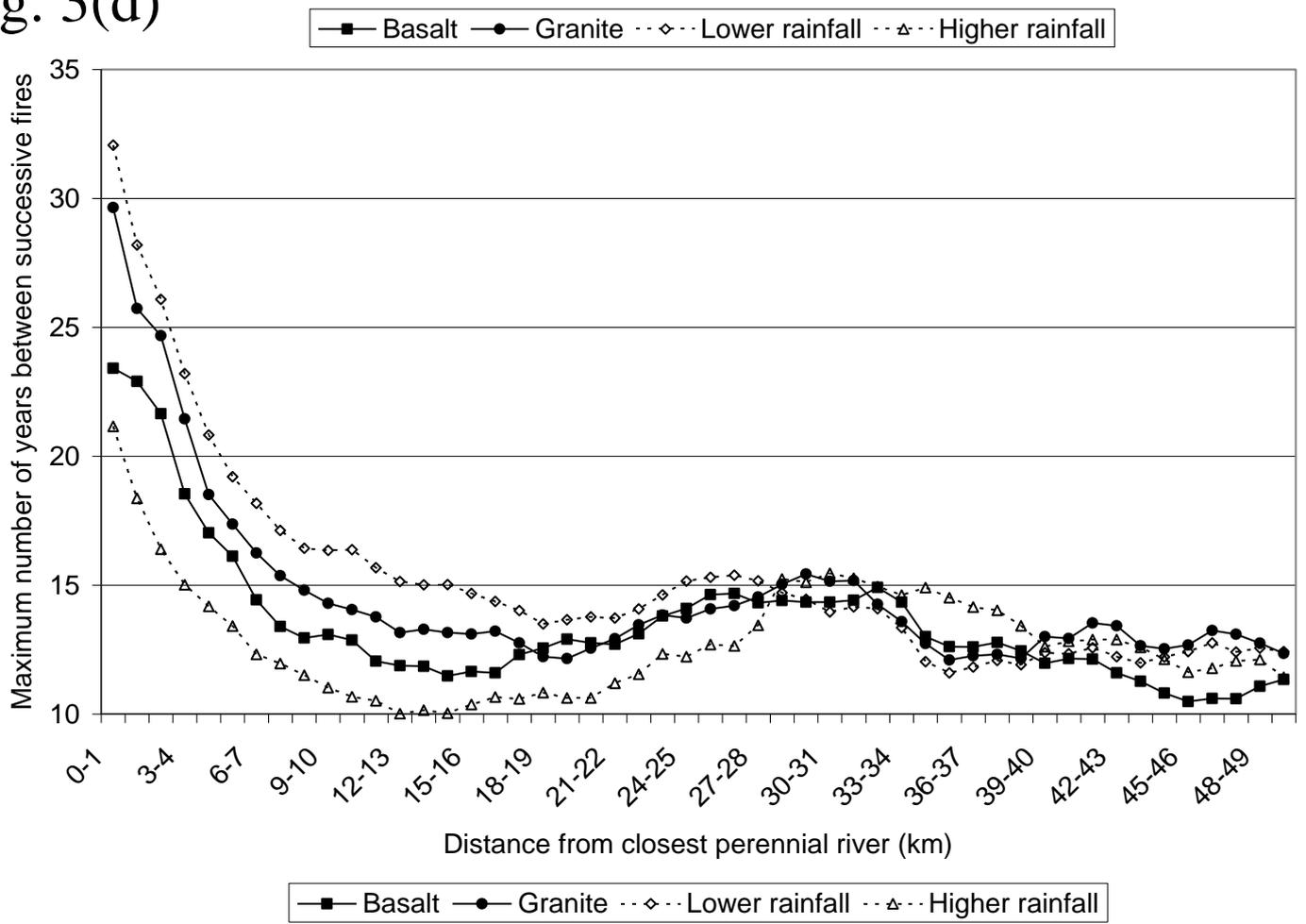


Fig. 3(e)

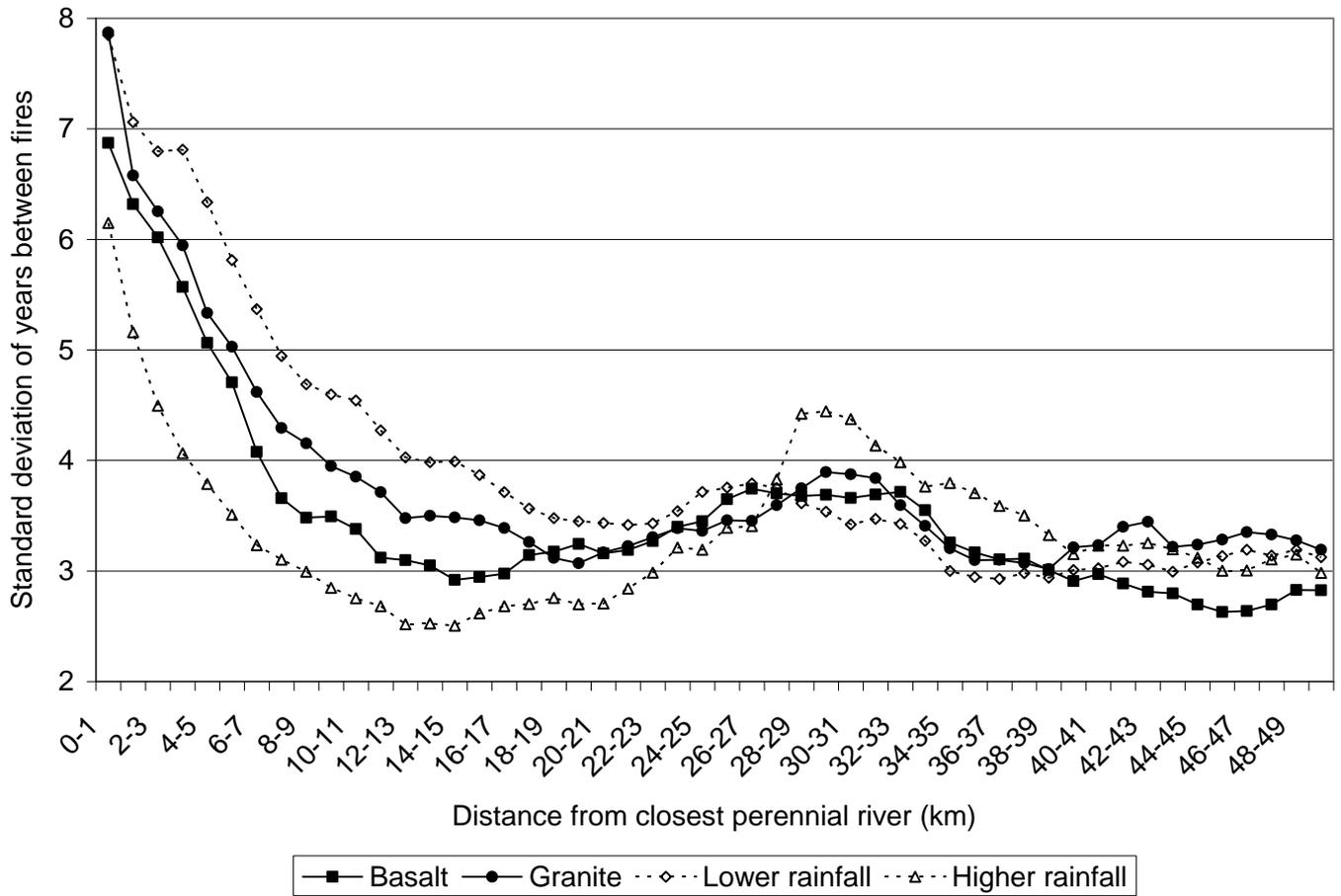


Fig. 3(f)

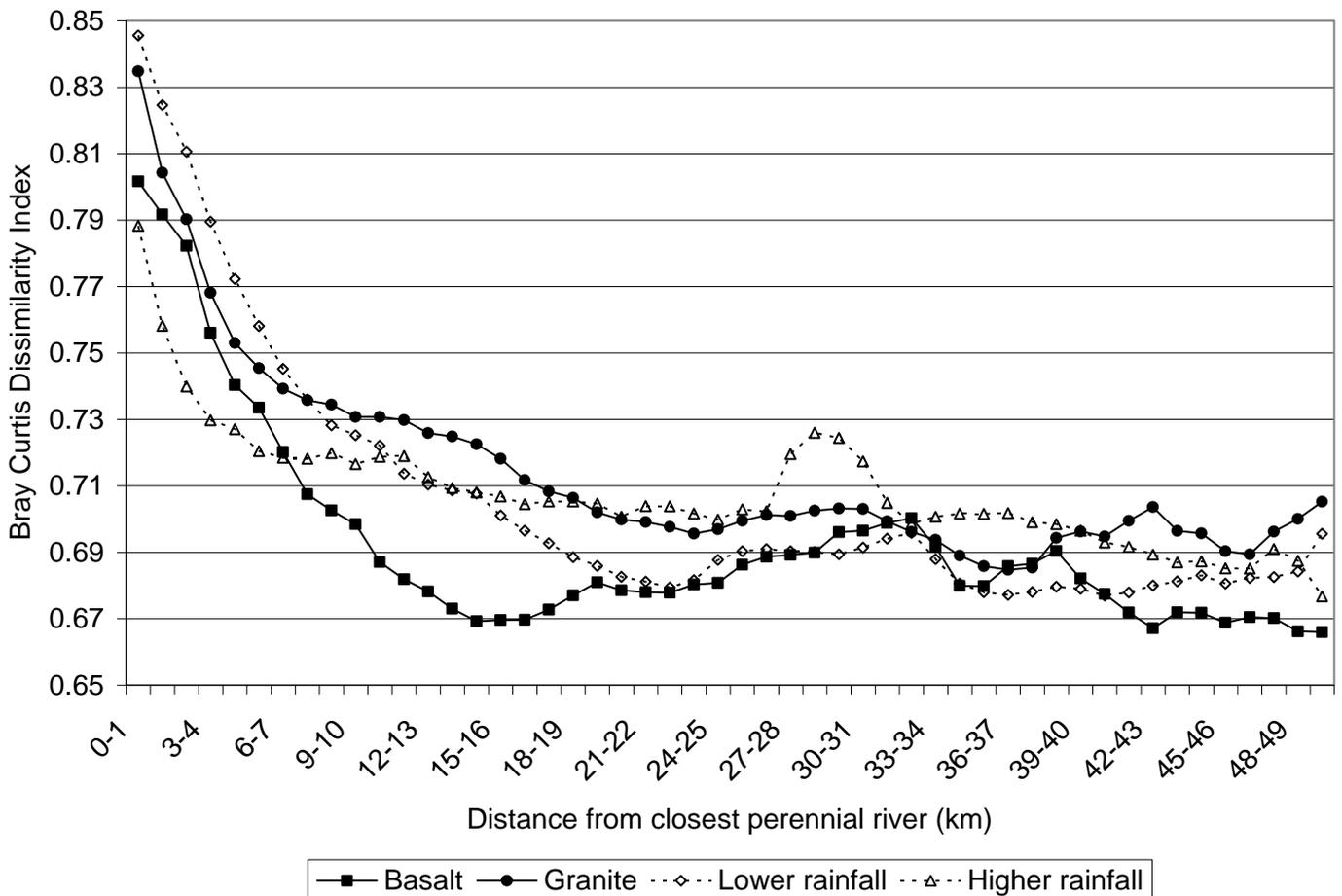


Fig. 4

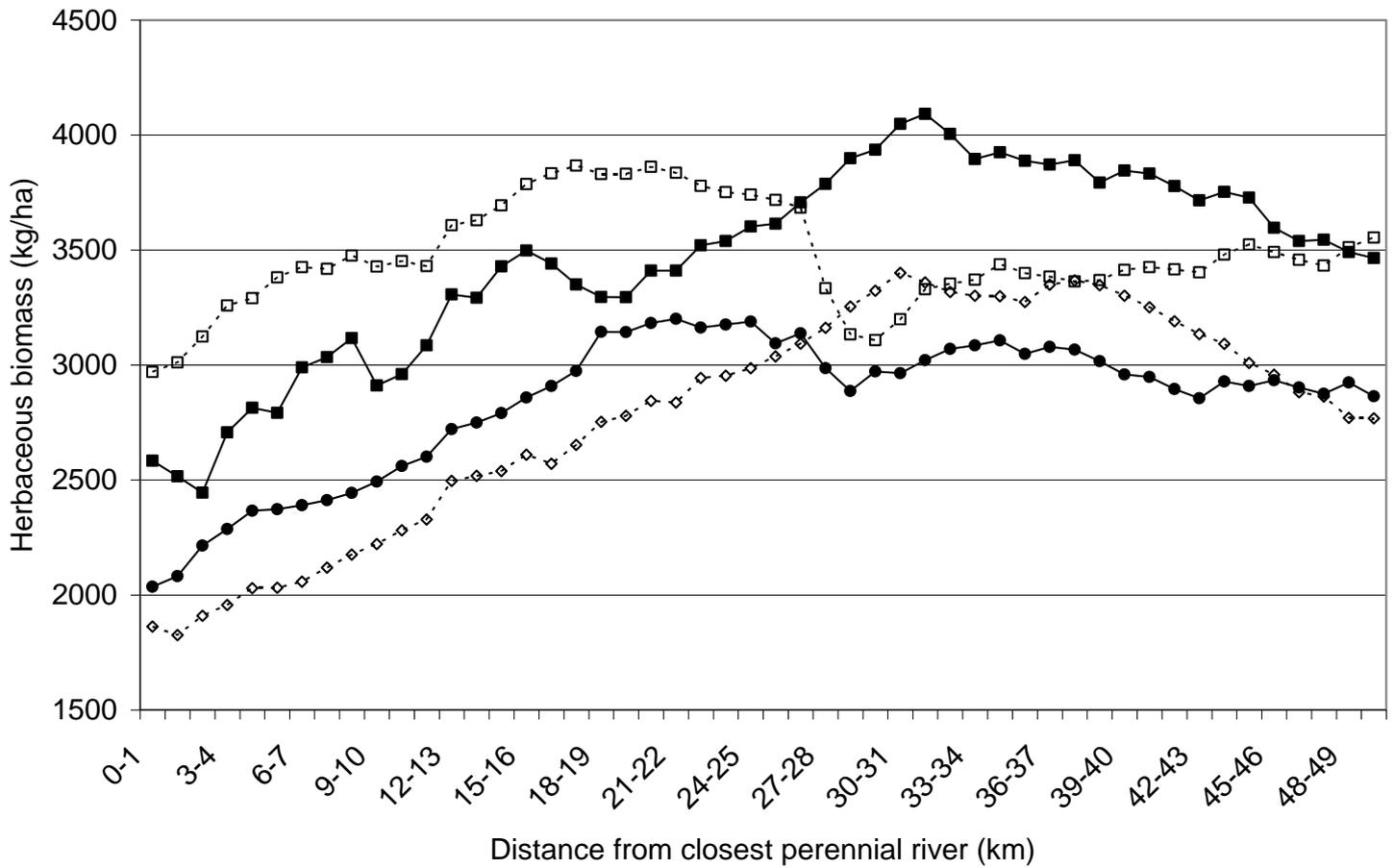


Fig. 5

