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Savanna burning and the assessment of long-term fire experiments with particular reference to Zimbabwe

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Abstract: Long-term fire experiments in savannnas are rare, given the difficulties and demands of operation. Controlled fire experiments date from colonial times in West Africa, although the largest and best-known is located in the Kruger National Park, South Africa. The achievements of these experiments are assessed from examples in Africa, South America and Australia. A less well-known experiment in Zimbabwe was sited at the Marondera Grassland Research Station and ran from 1953 to 1991. Some of the preliminary results on the impact of fire on vegetation are analysed and compared with further vegetation surveys in 2007. Studies on tree growth in this *miombo* savanna woodland indicate that the plots burned at 3 and 4-year intervals recovered to greater mean heights than the unburned control plots. There was no significant variation between treatments, suggesting that the few trees that did survive in the frequently burned plots were large specimens. *Brachystegia* and *Julbernadia* dominated the plots throughout and after the experiment. Basal area and stocking density were highest in the 4-yearly burned plots but there was a high variability throughout the experiment, suggesting that many trees may have attained heights and bark thicknesses sufficient to protect from fire damage. Fire also affected the composition of the herbaceous plant community, but not the number of species. By the end of the experiment some grass and sedge species had flourished whilst others revealed greater susceptibility to fire, and fire-tolerant species predominated in the most frequently burned areas. The experimental design appeared to cope well with the variability between plots and indicated the soundness of the initial design and its implementation.

Key words: Fire experiments; savanna woodlands; *miombo*; Zimbabwe; Marondera

I. Introduction

Tropical savannas are characterised by a co-dominance of trees, shrubs and herbaceous plants, whose proportions are determined by environmental, ecological and human parameters and fashioned by fire. Savanna vegetation is typically fire-adapted and resilient. Nevertheless, the frequency and intensity of fires affects vegetation structure and many of its constituent and associated characteristics, presenting major problems for ecosystem assessment and resource management (Bond *et al.,* 2005; Bond and Keeley, 2005; Bowman and Franklin, 2005; Van Langevelde *et al.,* 2003; Bond and Van Wilgen, 1996; Braithwaite, 1996; Lonsdale and Braithwaite, 1991; Frost and Robertson, 1987). Recent developments in spatial data acquisition from remote sensing, and handling within GIS, have provided more accurate information on the incidence, frequency and impact of fires (e.g. Schroeder *et al.,* 2008, 2005; Chongo *et al.,* 2007; Verbesselt *et al.,* 2007; Lentile *et al.,* 2006; Palacios-Orueta *et al., 2005,*2004). The ability to observe fires from airborne and satellite sensors has been a catalyst in stimulating interest in the role of biomass burning particularly in the tropics. Satellite images, linked to models, reveal that in Africa over 2.6 million km^2 of savanna burn every year, contributing 48% of global pyrogenic CO_2 emissions, albeit with a high estimated error (Schultz *et al.,* 2008).

Numerous experiments and models of savanna burning have been developed in order to understand the nature and impact of fires, although relatively few have been maintained for a long period or have been sufficiently replicated to give a consistent picture (Bond and Van Wilgen, 2005). In any case it has proved difficult to include all possible variables in field experiments that might influence burning, and therefore all controlled experiments are to some degree distant from the reality of highly varied wildfire events.

Field experiments manipulating burning episodes vary greatly in rigour but have been assayed for well over half a century. Four main groups of objectives can be extracted from the literature: i. many of the early experiments were carried out in savanna woodlands or tropical forests with the aim of assessing the effects of slash and burn agriculture or clearance on the environment; ii. field experiments were established to analyse the effects of burning or various forms of management for grazing; iii. several long term experiments were set up by colonial regimes to test the impact of fire frequencies or timing on vegetation, mostly to improve management; iv. wider environmental concerns have more recently been addressed particularly, such as biomass change, carbon emissions and sequestration, as well as particulate and other gas fluxes.

The present objective is to assess the information from global fire experiments with particular reference to the impact of long term fire trials on vegetation, and to illustrate the inherent concepts in a long-term fire experiment in Zimbabwe.

II. Savanna fire experiments: a global perspective

Observations of the effects of fire on vegetation have been published since early colonial experiments in Africa (see Aubreville, 1947). The experiments that were established then form some of the longest running ecological experiments in the world. However, with some exceptions in richer countries, they have been little reported in the scientific literature and their results have rarely been synthesised. This is despite an upsurge of interest in the biogeochemistry of biomass burning and in the role of fire in the earth system (such as the NASA/ University of Maryland Fire Information for Resource Management System FIRMS; the Australian SENTINEL system for mapping fires; SAFARI, the Southern Africa Fire Atmosphere Research Initiative and SavFIRE (Savanna Fire Ignition Research Experiment). However, there are numerous difficulties in designing and operating such experiments, particularly when continued over decades where the aims may change over time, where there are funding difficulties, and where there are problems in carrying through the experiments consistently with sufficient replication (Vogl, 1974). There have been several other serious criticisms of some burning trials, relating to the lack of an initial sound protocol and the inevitability of a rigid design. The experimental plots have often been relatively small in relation to the area represented (to permit adequate control), contrived (since wildfires are frequently spontaneous and irregular), and artificial (in the sense that they try to simplify conditions whereas the reality is a heterogeneous ecosystem). The experimental conditions may also be considered untypical of the areas outside the fire trials. Sometimes the documentation of the pre-fire conditions has been shown to be inadequate and there is frequently a lack of supporting environmental and ecological information (such as soils and drainage, rainfall and seasonality, phenology, herbivory or the ecological context beyond the experimental boundaries; see Sankaran *et al.,* 2008). Despite these criticisms, a great deal of valuable information has been derived from past experiments, especially from an ecological and biogeographical perspective, providing a test bed for ideas as well as fundamental data on the effects of burning (Van Wilgen *et al.,* 2007; Higgins *et al.,* 2007).

Not only have fire trials have been conducted for diverse reasons, but the original objectives have often been added to or modified over time. For example, some experiments designed to test the effects of burning on bush encroachment for pasture research have later been utilised to study changing vegetation structure and plant composition. The development of concepts on savanna heterogeneity (such as non-equilibrium theory, patch mosaics and theories on the dynamics of tree-grass co-existence), has frequently drawn on the data from existing field experiments. Early controlled experiments were carried out in West Africa and were established by colonial regimes in many countries in East and Southern Africa, with the largest and best documented study located in the Kruger National Park. Other well documented studies have been published in Australia, notably in the Kakadu National Park; in the Brazilian *cerrados* (Projeto Fogo, administered by the environment agency, IBAMA, and earlier studies); and in the Venezuelan/Colombian *llanos*.

When controlled field experiments were first initiated, fires were generally perceived as detrimental leading to land degradation (see the summary of ideas in Laris and Wardell, 2006). They have long been seen as instrumental in influencing forest-savanna boundaries (Müller *et al.,* 2007; Hoffman *et al.,* 2003; Fensham *et al.,* 2003; Furley *et al.,* 1992). Aubreville's pioneer experiments in West Africa in the 1930s influenced a succession of later African studies (such as Rose -Innes, 1971; Charter & Keay, 1960 and Hopkins, 1965 at Olokomeji, Nigeria; Ramsay & Rose-Innes, 1963; Brookman-Amusah *et al.,* 1980; Swaine, 1992 and Swaine *et al.,* 1992 in Ghana; Trapnell, 1959 and Lawton, 1978 in what was then Northern Rhodesia, and later the work of Chidumayo and his associates (1997, 1996, 1988); Nangendo *et al.,*2005 in the Murchison Falls National Park; Eldroma, 1984, in the Queen Elizabeth National Park, Uganda; Van de Vijver *et al.,*1999 and Ferguson, 1998 in Tanzania). The results, also confirmed in other parts of the world, tended to show that over time, fires (especially late dry season), created a landscape with fewer trees and larger expanses of grassland, whilst fire suppression promoted a reverse trend (later substantiated by work such as Menault and Cesar, 1982 and Loupe, 1995 in Ivory Coast). Research since the 1960s has indicated a growing awareness and understanding of the complexity and dynamics of the savanna biome, and has included a multitude of environmental and human variables in process models. For example, savanna woodland diversity and structure has been shown to reflect combinations of factors such as prescribed fire along with grazing and termite activity (eg Traore *et al.,* 2008 in Burkina Faso), and seed germination (Gashaw and Michelsen, 2002 in Ethiopia), or numerous publications on fire and elephant interactions (eg Lock, 1977). More detailed ecological parameters have included different timing of burns, different fire intensities and irregular fire behaviour patterns. While the early research concentrated on the impact of fires on the environment, more recent studies have also concentrated on managing fire regimes for grazing (eg Savadogo *et al.,* 2007), for wildlife and conservation, and on designing improved fire policy (Whitehead *et al.,* 2005; Dyer and Smith 2003; Bond and Archibald, 2003; Dyer *et al.,* 2001).

1. Fire experimentation in northern Australia

Australian savannas occur at the drier end of the savanna spectrum with frequent fires and woody plants characterised by eucalyptus. The emphasis of research into fire impact and management has been predominantly in northern Australia (Bradstock *et al.*,2002). An overview of fire experiments in the 2m km $^{-2}$ of tropical savanna is given in Williams (R.J) *et al.* (2003), and there are numerous research sites and studies reflected in published literature and in reports from the research stations. Controlled experiments of international significance have been sited at Munmarlarly, Kidman Springs and the well known Kapalga sites in the Kakadu National Park. Munmarlarly ran from 1973 to 1996 in a eucalypt-dominated savanna woodland, with 4 replicated 1-ha plots designed to investigate the interactions between fire, landscape and biodiversity. Lonsdale and Braithwaite (1991) in a statistical analysis of earlier studies concluded that the different fire regimes resulted in negligible changes in the vegetation, suggesting long term stability. In a later detailed assessment, Russell-Smith *et al.* (2003a) explored relationships between fire regimes and savanna responses, and detailed the variations between the types of woodland savanna, different timings for burning and associated effects on some soil properties. Frequent burns at low or moderate intensities produced a structurally stable system, though the authors suggest that no matter how rigorous the fire plot design, the results are limited for fire management purposes in view of the idiosyncratic nature of natural fire regimes over multiple temporal and spatial scales. The Kidman Springs experiment examined the productivity and composition of the herbaceous layer in an effort to better inform pasture management. The Kapalga experiment in the Kakadu National Park ran from 1989 to 1995 with a range of fire treatments similar to those at Munmarlarly but with very large plots of 10-20km^2 in sub-catchments and can claim to be the world's largest fire experiment. There have been numerous published studies from Kapalga and the wider National Park, covering a gamut of ecological and environmental issues (see for instance Edwards et al., 2003 looking at fire regimes and biodiversity; Price *et al.,* 2005, studying fire heterogeneity; Prior *et al.,* 2006 examining the biotic and abiotic factors influencing tree growth rates); Liedloff *et al.,* 2001 who modelled fire and grazing interactions). The well documented results from Kapalga are summarised in the book by Andersen *et al.* (2003) and subsequent articles (e.g. Andersen *et al.,* 2005). The main results showed that, with the exception of riparian vegetation and associated streams and also small mammal populations, much of the savanna biota showed little or no response to burning, even with the most extreme fire regimes and is therefore extremely resilient to fire. Where fire did have a marked effect it was felt between burnt and unburnt areas rather than high versus low intensity fires and this emphasised the importance of analysing fire frequencies.

Less well known Australian studies were sited in the Bowling Green bay National Park near Townsville, Queensland (Williams,P.R *et al.,* 2003a), where 3 fire regimes were studied between 1997 and 2001 over an area of 10ha. The abundance of most species was stable irrespective of the fire regime although several sub-shrubs, ephemeral and perennial forbs and grasses did increase in abundance particularly following late season fires. The density of trees increased markedly in the absence of fire for 4 years. Fire patterns and frequencies were compared over a 4-year period by satellite imagery for two areas in northern Queensland by Felderhof and Gillieson (2006), where the Mount Isa area in semi-arid conditions showed patterns and frequencies similar to those reported elsewhere in arid parts of Australia, whereas the Cape York Peninsula more closely resembled more mesic savannas. At the Bradshaw research station (NT) Gill *et al.* (2003) also examined fire and patchiness. Other studies have looked at the depth of fire penetration into the ground (Williams, PR, 2004), fire effects on seedbanks and seedlings (Radford et al., 2001; burning and the breaking of seed dormancy (Williams, PR *et al.,* 2003a.b); alien grass invasions (a major topic elsewhere as in Brazil) (Rossiter, 2003); important contributions to the debates on fire and productivity (e.g. Beringer *et al.,* 2007; Bennet *et al.,*2003); or the long history of fires in Australia and the role of the aboriginal activities (Bowman *et al.,* 2004; Bowman and Prior, 2004; Russell-Smith *et al.,* 2003b; Preece, 2002). There are also valuable accounts of fire impacts on wildlife which cannot be covered in this assessment but are summarised in publications such as Whitehead *et al.* (2005) or Woinarski *et al.,* (2004).

2. Fire experimentation in southern Africa

Fire has always been to the forefront in consideration of African savannas (eg Booyson and Tainton, 1984). In southern Africa, the research carried out in the Kruger National Park stands out in terms of its large scale, high level of replication, long-term studies and the extensive range of associated botanical, wildlife and environmental concerns (over 70 publications on different aspects of the research). As Higgins *et al.* (2007) have commented, few long-term data sets are available anywhere to document the extent to which fire can structure savannas. The Kruger research provides invaluable information on fire impacts and also offers a yardstick of comparison for the neighbouring Zimbabwe experiments described later. The National Park at Kruger was established in 1926 and the long-term, plot-based fire experiment started in 1954 in order to provide a scientific basis for fire management. The large area covers a range of rainfall from 350mm in the north to 750mm in the south with extended wet and dry periods significantly affecting fire occurrence, principally through the combustible fuel load (Van Wilgen *et al.,* 2003). Four representative woody savannas are dominated by Acacia nigrescens and Colophosperum mopane over basaltic soils, Terminalia sericea and Combretum spp. on granite soils. Natural fires typically break out in the late dry season with a mean fire return interval (1941-96) of 4.5 years. The fire experiments comprised burns at varying return intervals and seasons in a series of 7ha plots in the 4 types of savanna vegetation. The plots have been open to wildlife grazing and it has been suggested that this may have influenced some of the results. A valuable overview is presented in du Toit *et al.* (2003) and useful evaluative commentaries are given in Higgins *et al.* (2007) and Woods *et al.,* 2002. Van Wilgen *et al.* (2007), give a helpful summary table of fire effects on vegetation, wildlife and a number of environmental properties. As far as the effects of fire on woody plants is concerned, the key findings were that fire frequency, fire season and complete exclusion from burning did not affect the size of tree populations and that there was no decrease in tree density with increasing fire frequency. This apparently unlikely result was attributed to the ability of the woody (and other species) to re-sprout from the base and immediate sub-surface. Re-sprouting increased with fire intensity, which scorched aerial parts of the trees (Nefabas and Gambiza, 2007; Kennedy and Potgieter, 2003; Bond and Midgely, 2001; Trollope *et al.,*1995, Trollope, 1984, 1982). While constant fires lowered the size of individual species they rarely killed them. Nevertheless fire had significant effects on the size, structure and biomass of the tree populations, and studies on fire intensity reveal the interaction of numerous controlling factors particularly seasonality (influencing fuel moisture) and past fire frequency (Govender *et al.,*2006). Enslin *et al.* (2000) and Shackleton and Scholes (2000) showed that while there had been no significant changes in woody species from 1954 to 1998, density decreased on biennial but increased on triennial treatments. Control plots where fire was excluded often had dominant larger trees, whereas the fire-affected plots possessed numerous stunted individuals. The influence of fire exclusion was more obvious in areas of higher rainfall resulting in higher biomass figures. The timing of the fire season was also significant (Govender et al., 2006) with intensities being lower in summer and highest in the winter fires, though intensities did not vary between annual, biennial triennial or quadrennial intervals.

As far as the impact of fires on the herbaceous components is concerned, current research appears to indicate that fires in the dormant season have little effect on community composition, but wet season fire and fire exclusion does have a significant effect (Van Wilgen *et al.,* 2007). Fire is known to be important in influencing herbaceous plant communities in savanna woodlands, particularly where there are effects on the seasonal availability of water (Chidumayo, 1997). Post-fire succession in the Okavango, Botswana was shown to be a reflection of the plant species characteristics, the seedbank and plant mobility in early successional stages and by competition and niche differentiation at later stages (Heinl *et al.,* 2007). After 50 years of burning, grass species richness at Kruger was greatest where fire was excluded or burnt during the wet growing season - again more marked in areas of higher rainfall (Van Wilgen *et al.,* 2007). These findings suggest that in the Kruger experiment the manipulation of fire regimes is not critical for the maintenance of diversity in the herbaceous component. In a similar vein, Hudak *et al.* (2004) concluded that increased fire occurrence promoted landscape heterogeneity whilst fire exclusion did not. Mycorrhizae and root architecture are believed to be strongly affected by fire (Hartnett *et al.* 2004) and the extensive mycorrhizal network underpinning savanna vegetation, particularly in low nutrient conditions, represents a valuable strategy for maximising resources under stress and disturbance.

3. Fire experimentation in South America

South and Central American savannas are amongst the most humid of global savannas and tend to occur over highly weathered, infertile soils. Work on long term fire experiments in savannas of South America has been mostly conducted in Brazil and to a lesser extent in Venezuela (Miranda *et al.,* 2002; San Jose 1991, 1997; Mistry, 1988). One of the best known long-standing fire experiments is located at the Ecological Reserve of the Brazilian Institute of Geography and Statistics (IBGE) and in the adjacent University Agua Limpa Reserve near Brasilia. This IBGE Reserve has sections that have been protected from fire since 1972, and contains an array of study plots established on different *cerrado* physiognomies that are subjected to different fire regimes (Santos *et al.,* 2003). There are even older fire experiments set up in Brazilian Cerrados (Coutinho, 1982, 1990), although they do not present rigorous information on the initial environmental conditions before the trials commenced (Moreira, 2000). More recently, work has been conducted both on the changes in structure and composition on a fire-protected cerrado sensu stricto in NE Brazil (Roitman *et al.,* 2008), and on the interaction between of soil and vegetation under different fire frequencies in study sites located in the Emas National Park (Silva and Batalha, 2008). Research on the impact of exclusion of fires on savanna vegetation in Venezuela has been conducted since 1960 at the Llanos Biological Station in the central part of the country and has been the subject of long-term monitoring as shown in successive studies published by San Jose and Fariñas (1983, 1991). Research in this region was subsequently extended with a view of studying the impact of various regimes of land use and fire frequency on the ecosystem (Silva *et al.,* 2001).

A common pattern observed in all the studies mentioned above is that of the increases in density of woody species in *cerrado* sites protected from fire. Moreira (2000) conducted research at the IBGE Ecological Reserve comparing five different savanna physiognomies that were burnt every 2 years to other five savanna counterparts that were excluded from fire for over 18 years. The study shows that fire exclusion favoured fire sensitive species and led to an increase in woody elements in all physiognomies studied although the effect of fire was shown to have a stronger effect on open savannas than on intermediate physiognomies. This work along with previous studies in the *cerrados,* concluded that if sites were to be protected for long periods of time it would encourage the more wooded (*cerradão)* physiognomies (Oliveira and Ratter, 2002; Hoffman and Moreira, 2002: Furley, 1999; Ratter, 1992; Coutinho, 1990, Eiten, 1972). Roitman *et al.* (2008) also showed that the structure and composition of a fire-protected savanna underwent significant changes (see also Hoffman, 2002; Hoffman and Solbrig, 2003; Hoffman *et al.,* 2003). Studies carried out in the Venezuelan llanos also showed a highly significant increment of the woody component and a change in floristic composition of the herbaceous layer after exclusion of fire and cattle grazing (San José and Fariñas, 1983 and 1991; Silva *et al.,* 2001). An increased occurrence of fires in the *cerrado* formations has also been evaluated with the aid of matrix model projections (Hoffmann, 1999), showing reductions in the density and size of woody plants as a result of increases in the number of fire events. The persistence of woody shrub elements in the more open cerrado reflects their ability to resist burning, through the development of resistant underground lingo-tubers (Medeiros and Miranda, 2008).

Several studies have suggested that burning is the main determinant factor explaining the differences in vegetation between fire-protected and unprotected sites given a lack of other significant factors, such as differences in soil properties (Moreira, 2000; Roitman *et al.,* 2008). Whilst some of the work carried out in *cerrados* has reflected this absence of relationship between soil properties and the different cerrado formations, this may result from the flux of vegetation over the landscape with time, and other studies have indicated stronger links between soil fertility and a gradient of *cerrado* physiognomies (Furley, 1996; Lopes and Cox, 1977ab; Goodland and Pollard (1973). This is likely to relate to the increased organic matter found in the more wooded formations and in part to local differences in parent material. Uptake of soil water was also shown to be greater for protected areas than for burned areas (Quesada *et al.,* 2008).

Numerous studies illustrate how the spectrum of *cerrado* physiognomies reacts to and is resilient to fire and Felfili *et al.,* (2000) and Roitman *et al.,* (2008) show how fire acts as an intermediate disturbance factor impeding the ability of the *cerrado* formations to reach 'carrying capacity'. Overall, there seems to be a common view shared by all long term fire exclusion studies, in South American studies and elsewhere, that an extremely complex interaction exists between fire and the dynamics of woody cover in savanna communities and that burning results in constant disequilibrium.

III. Achievements of long term experimentation on savanna vegetation

The results from global long-term experiments afford many valuable insights into the effects of burning (Miao *et al.,* in press). Early ideas considered fire to be entirely harmful, leading to land degradation with woodland or forest retreat and savanna advance. The role of fire has subsequently been explored in much greater detail and precision, and has shown that regular low intensity fires at critical seasons can reduce fuel load, lower pest infestations, return nutrients to the ground and encourage plant diversity. The influence of fire behaviour, intensity and frequency, the timing of burns, and the overall impacts on vegetation structure, species composition and species resilience in varying conditions of rainfall, soil fertility, herbivory and human interaction have been shown to be intricately interwoven and to some extent site-specific.

The degree to which fire influences savanna vegetation is still controversial (see Scholes *et al.,* 2003; Van Wilgen *et al.,* 2003) but plant species have indisputably evolved fire-tolerant and fire-persistent traits (Allen, 2008). A number of general observations stand out as a result of prolonged burning:-

i. vegetation structure is strongly affected, lowering the proportions of trees to shrubs and herbaceous plants and reducing ground cover; irregular fire behaviour leads to patchiness; ground layer plants change from perennials to a dominance of annuals.

ii. tree height decreases inversely with fire frequency; total above-ground biomass is severely reduced.

iii. tree and woody shrub density remains fairly constant through seedling recruitment and resprouting, though numbers of individual species numbers may be affected.

iv. overall plant composition is little affected, although there may be feedback behaviour between fire and biodiversity.

v. once trees reach a certain height (~3m+), they are less susceptible to burns.

vi. burning has relatively little impact on subsurface rooting structures and, except with extraordinary intensities, does not penetrate far below the ground surface; however surface soil organic matter (0-5cm) can respond rapidly to changes in fire regime (viz. Williams *et al.,*2008; Bird *et al.* 2000).

vii. burning has significant effects on soil nutrient dynamics (which in turn can influence plant growth) and greenhouse gas exchange (Rees *et al.,* 2006).

Further characteristic features include:

viii. fire has a more marked effect on the structure of woody vegetation when it occurs in higher rainfall areas (over ca. 600mm y^{-1}).

ix. the burn season and timing affects its impact on the vegetation, mediated by factors such as fuel moisture, fire weather and whether the vegetation has senesced.

x. fire intensity and thus severity varies with fuel load, moisture content and meteorological conditions.

xi. characteristic plants found over the spectrum of savanna formations may be fire-sensitive or fire-tolerant and the proportions vary with fire frequency, intensity and behaviour.

xii. micro-environmental variations (such as soils or moisture availability) can have a distinct impact on burning and recovery.

xiii. the fire return interval in savannas is typically ~3 years but can in places be annual.

Overall however, the ability of the vegetation to recover from severe fires relates to favourable climate and local factors such as bark thickness, coppicing, suckering and resprouting from basal stems or rootstocks and the supporting role of mycorrhizae (Högberg, 1992; Hartnett *et al.,* 2004). Taller woody plants tend to survive many of the fires by growing higher than the flame zone (Gignoux *et al.,*1997; Caldwell and Zieger, 2000). The role of key environmental and ecological pressures, amongst which are soils and drainage, and patterns of herbivory, have also been added into dynamic models of savanna response to fire.

The present objective is to assess the information from global fire experiments with particular reference to the impact of long term fire trials on vegetation, and to illustrate the inherent concepts in a long-term fire experiment in Zimbabwe.

IV. Fire trials in Zimbabwe

As in many parts of Africa, there is evidence of sustained use of fire throughout the country for thousands of years and for a multitude of purposes, and natural fires have occurred since long before man appeared. In what is now Zimbabwe, extensive fires caused by human activity over prolonged periods have resulted in a manmade landscape over most if not all of the country, and it has been suggested that the savanna woodlands as well as the grasslands are the end result of human disturbance (Austen, 1972; West, 1972; the fire climax of Trapnell, 1959 and Lawton, 1979). Stem mortality is especially noticeable in the more semi-arid districts, such as the savannas on Kalahari sands in the west of the country (Holdo, 2005), where late dry season fires can severely damage woody plants (Gambiza *et al.,* 2005). The savanna woodlands are crucial to the livelihoods of many rural people and a simulation of woodland dynamics under different management systems is assessed by Gambiza *et al.,* (2000).

The two best known and most detailed studies are located at the Grassland Research Station at Marondera in dry woodland savanna (*miombo*) to the west of the country, and at Matapos research station in the south-west with a tree-bush vegetation cover. At Matapos, early small plot experiments were started in 1947 to study different management strategies and the results of the trials were later reported by Kennan (1972). Two typical areas of *thornveld* (Acacia-dominated tree-bush savanna on moderately fertile red or black soils derived from epidiorite and schist), and *sandveld* (Terminalia-Burkea tree-bush cover on relatively infertile gneiss and granite). The plots on the *sandveld* (started in 1947) measured 100x90feet and those on the *thornveld* (90x75feet) but otherwise all the plot experiments were identical (12 randomised treatments, replicated three times). While grazing by domestic animals was excluded, the sites were open to wildlife and therefore grazed. Despite considerable care, the results have been difficult to interpret, though the point quadrat and tree count data yielded useful information. Basal cover was affected by fire frequency, which also reflected a succession of stages from more open to more woody conditions. There were also interesting comments made on the susceptibility or resilience to fire by different grass, shrub and tree species and the also on the effects of burning at different times of the year, with the *sandveld* vegetation being on the whole much more resistant to burning than the *thornveld*. Additional trials were also carried out at Nyamandhlova Station (hotter and drier) with similar results to the *thornveld* over a 17 year period of analysis, and at Tuli (low veld with very high temperatures and low rainfall) but the latter was abandoned because of the extreme irregularity of rainfall (Kennan, 1972).

The other major long term experiment was situated at Marondera, and the remainder of this contribution will outline and analyse the preliminary results from this set of fire trials.

V. The Marondera fire trials from 1953 onwards

A long term burning experiment was set up in 1953 at the Marondera Grassland Research Station southeast of Harare. The station is situated within one of the largest blocks of *miombo* vegetation in southern Africa and lies at an altitude of 1610m. There are two distinct seasons – hot and rainy from November to April and then a cool and dry season. Some rain occurs in every month, with a maximum from December and February. The driest months of the year are July and August. The mean annual rainfall is around 900mm, but there are wide variations between years. There have been a number of very dry years (linked to El Nino events), with <500mm total annual rainfall, including 1990/1 and 1991/2 towards the end of the fire experiments. Above average rainfall was received in several years, including 1953 when most of the treatments began. Over the period 1941-1997, the maximum total annual rainfall recorded was 1700mm in 1980/1. Mean monthly temperatures range from 11^0 C in June to 19^0 C in November. Temperatures appear to have been slightly higher than average from 1993 to 1997. Occasional night frosts in the winter period over elevated ground restrict the growth of plants.

The *miombo* is a mesic to dystrophic savanna woodland (Campbell *et al.* 1995, 1998; Chidumayo 1997). It is a diverse formation with an estimated 8500 species of higher plants of which 334 are trees (Frost, 19960. It is dominated by Brachystegia, Julbernadia and Isoberlinia, (Fabaceae, sub-family Caesalpinioideae). In the central plateau of Zimbabwe, which incorporates the study area, the natural vegetation is described as deciduous *miombo* woodland with B. spiciformis and J. globiflora as dominants. Other common associated plants are Albizia antunesiana, Combretum molle, Faurea speciosa, Strychnos cocculoides and Vangueria infausta. There is considerable variation in the composition and biomass of the herbaceous layer, and includes the grasses Hyparrhenia, Andropogon, Loudetia, Digitaria, Eragrostis (Frost, 1996), and also Brachiaria brizantha, Craspedorhachis rhodesiana and Rhynchelytrum nyassanum. The shrubs typically comprise the genera Dolichos, Eriosema and Indigofera.

The soils in the Marondera region are mostly Ultisols, but over the experimental plots are Alfisols (orthoferrallitic group, Thompson and Purves, 1978, Nyamapfene, 1991). They are generally coarse-grained sandy soils derived from granite, characterised by a kaolinitic clay fraction and free sesquioxides. Less than 5% consists of weatherable material and the soils are heavily leached. They are acidic and nutrient–poor, with low levels of exchangeable bases, nitrogen and extractable phosphorus. The weak cation exchange capacity results from low levels of organic matter and the low-activity clay fraction.

The objective of the original study was to investigate the effectiveness of fire in controlling coppice re-growth of the woody vegetation (Barnes, 1965; Strang, 1974). Originally the area was covered by dense woodland and, before the beginning of the experiment, the vegetation was cleared manually (in 1948) by cutting the woody component at ground level. However, there were no measurements made either of the vegetation or the soils prior to the application of the burning regime. At the start of the experiment, the dense coppice that had grown since 1948 (6-15ft high with a few large trees) was cut to ground level and the first of the fire treatments started in 1953 (Barnes, 1965).

The fire experiment consisted of a 5x2-split plot design plus control plots with replicates in two randomised blocks. Each plot (36x60m) was subdivided into two equal parts of 18x60m. One part was cleared by mattock and both parts were included in the fire frequency trials. The implications of the mattock + burn treatments are not considered here. The fire treatments established five burn frequencies. These were burning annually, burning every second year, every third year, every fourth year, and no burn (control plots). Replications were assigned to the longer intervals between fires so that at each frequency, there was a plot burnt each year. Fires were started late in the dry season, before the spring rain, to achieve as hot a burn temperature as possible. The plots were surrounded by a 3 m wide fire- break and grazing was excluded from the area.

The treatments are described and numbered as follows:

burnt annually, started in 1953 (F1)
burnt biennially, started in 1953 (F2)
burnt biennially, started in 1954 (F2)
triennial burn, started in 1953 (F3)
triennial burn, started in 1954 (F3)
triennial burn, started in 1955 (F3)
four-year burn, started in 1953 (F4)
four-year burn, started in 1954 (F4)
four-year burn started in 1956 (F4)
control, complete protection from fire (F0)

These plots were laid out in the two parallel blocks as illustrated in Fig 1 and lie on a north-facing, gentle slope (ca. 3 degrees). The fire treatments continued until 1963 when they were temporarily abandoned because of severe damage from cockchafer grubs (Strang, 1974). The fire experiment was finally concluded in 1991 and species abundance was analysed for data collected in 1987, 1988 and 1990. Further measures of basal area and tree height were undertaken later in 2007, 16 years after the last fire treatments.

The effect of burning on tree growth

After 9 years (1953 –1962), the growth of trees and re-growth coppice was slight and there was little difference evident between the burn frequencies (Table 1, left). Even with a four-year burn interval there was little sign of a growth in height. The plots that had complete protection from fire showed the greatest tree growth, height and canopy development by 1962. Clearly burning has had a dramatic impact, though it should be remembered that a half of each plot was cleared of woody vegetation before the start of the fire treatments and therefore there was less buffering effect. As previously reported (Campbell *et al.,* 1995), fire is the principal agent in preventing regeneration and crippling young stems. Grundy (1992) and Grundy *et al.* (1994) observed that Brachystegia and Julbernadia increased rapidly after cutting and thereafter more slowly, but there is little evidence from our data that there was much growth during the early years of the experiment. However, later studies (Campbell, 1995) and our observations from 2007 (Table 1, right), indicate that the 3 and 4 year burn plots had recovered with greater mean heights (14.8-15.6 m) than were present in the control plots (9.5 m) ($p<0.05$). These canopies had a similar species composition to the protected plots but a less evolved structure. The height data show no significant variation between treatments. This indicates that the few trees that did survive on the frequently burned plots were large specimens similar in stature to those on the unburned plots.

Brachystegia and Julbernardia dominated the plots in 2007. The number of large stems of these two species, as a fraction of the total number of large stems, was 0.89±0.16 for F3, 0.83±0.05 for F4 and 0.98±0.03 for F0. F1 had no large trees (all data refer to live trees >20cm dbh). There was no significant treatment effect (1 way ANOVA $p=0.89$). From the historic notes it appears that this dominance was sustained throughout the experiment. Trees were almost completely absent from annually burnt plots even after 16 years (without fire) from the end of the experiment, but were somewhat more abundant in 2-yearly and 3-yearly burnt plots, where they tended to occur as suppressed saplings. Brachystegia and Julbernardia tend to be fire-sensitive when young and increasingly tolerant when they grow up to greater canopy height.

The basal area of trees measured in 2007 and stocking density was highest in the quadrennial burn treatment, but due to the high variability within plots, these differences were only significant at $p<0.1$ (Figure 2). The long intervals between fire (triennial and quadrennial burns) allowed many trees to attain heights and bark thicknesses where they were less susceptible to fire damage and such observations are consistent with previous studies (eg. Higgins *et al.,* 2007; van Wilgen *et al.,* 2007). The relatively larger trees and biomass in the 4-year burn compared with the complete protection treatment may have reflected the response of trees to the release of nutrients resulting from the burning treatments. The small plots, coupled with the intrinsic variability of woodland vegetation structure, means that the replication is critical for detecting differences between treatments. The ANOVA table (both parametric and non-parametric) for plot basal area shows that the variation within treatments, although large, was smaller than between treatments. The 17 replicates were enough to control for this heterogeneity and credit must be given to those who designed this experiment for achieving sufficient replication. This is in contrast to many other long-term experiments in Africa, which were unreplicated.

The floristic composition of the plots was not affected by the burning treatments, with 80-100% of the trees being of the two dominant species. This is in contrast to other long-term fire experiments (eg Zambia Trapnell, 1959), where the protected plots in wetter miombo conditions were invaded by forest and evergreen species. This may be due to a lack of nearby forest as a source of seeds at Marondera.

Effects of fire on herbaceous plants

The composition of herbaceous plant communities was influenced by treatments during the course of the experiment with both increases and decreases in individual species. Some of the grass species flourished after burning, whilst others decreased.

For example, in relative terms, *Hyparrhenia filipendula, Heteropogon contortus,* and *Eriosema affin* showed significantly higher numbers in burned plots than in protected plots ($p<0.001$; $p<0.01$; $p<0.01$ respectively). There was no significant difference between the annual burn and the quadrennial burn. Figure 3a. On the other hand, burning significantly reduced the numbers of some grass species such as *Rhynchelytrum nyassanum, Digitaria gazensis* and *Eragostis racemosa* ($p<0.05$). Figure 3b. At the same time the numbers of other species increased significantly following burning *Crasspedhorachis qfricana, Andropogon spp* and *Trachypogon spicatus* but there was no apparent trend (Table 2). This matches some of the observations in Strang (1974), although the trends were not statistically significant. The numbers of sedges and further grass species (such as *Rhytichelytrum repens* and *Nidorela uricrata*) were not significantly affected by the fire treatments. However, whilst there is evidence that the sedges were reduced by fire, no apparent trend could be discerned for *R. repens* or *N uricrata* (Table 3). These results parallel conclusions elsewhere illustrating how fires influence the demography of the herbaceous strata (such as Garnier and Dajoz, 2001).

Overall, the preliminary evidence from this long-term experiment indicates that frequent burning (annual, biennial and to a large extent triennial fires) severely inhibits woody plant growth and has a mixed impact on the herbaceous cover. Interestingly, the tree growth in the triennial and quadrennial fire frequencies ends up by being greater than that for the control plots. The results suggest that regeneration following fire is slow and patchy but that once woody plants have reached a stage where they are relatively resistant to fire, miombo woodland can develop rapidly. Frost (1996) suggests that this is likely to occur when trees reach 2m or more. Other research suggests figures of 3m+, and Ryan and Williams (submitted) in nearby Mozambique, suggests that there is a step change in survivorship at 12cm dbh. The present research indicates that this occurs after 10 years. The patchiness may reflect the nature and distribution of the pre-fire trial vegetation and its subsequent re-sprouting, and may indicate small scale vegetation patterns and nutrient cycling related to factors such as past disturbances or termite mounds (Traore *et al.,* 2008; Gill *et al.,* 2001; Campbell *et al.,* 1988, 1995). This variability in the burning impact is also likely to be the result of increasing bark resistance, lower levels of combustible material and gradual reduction in grass cover (potential comestible load) and therefore less severe fires. Conversely, as the woody cover is removed, so the herbaceous layer is likely to increase (increased light and moisture) leading to a greater fuel load and risk of burning (Gambiza *et al.,* 2000).

While the herbaceous cover predominates under frequent fires, the species composition varies, with a number of characteristic grass species flourishing following fires whilst others are severely reduced. Similar changes in species composition have been reported elsewhere (eg Hoffman and Moreira, 2002). The economic significance of this, for instance with regard to palatability for livestock (Hassan et al., 2008: Lowry, 1995), would be worth further investigation.

VI. Conclusions

The value of long-term studies in environmental and ecological research is controversial but the general consensus is that there is much useful information to be derived from well-founded and rigorously controlled fire experiments (Brown *et al.,* 2001). While there are many examples of field studies over years or even a few decades, the number of longer fire trials is very low. Marondera represents one of the few studies that have been soundly established and systematically carried out over nearly four decades (1953-1991) and in the face of numerous difficulties. The plots have remained intact since the cessation of the fire trails and continue to provide valuable information on re-growth.

The preliminary results for this dry miombo savanna woodland are in general accord with other reported work with a few significant additions. Frequent fires (especially annual and biennial) have had a marked impact on the vegetation structure and tree density, reducing the plots to fire-resistant grasses with a scattering of shrubs. A distinct recovery was observable with fire return intervals longer than two years and re-seeding, re-sprouting and coppicing were evident, resulting in a large number of individuals in any of the woody species populations following a number of years without fire. With repeated burning at short return intervals, plant composition changed in that fire-tolerant species became more dominant. Extreme burning over a prolonged period had a persistent effect for long after the fire experiment had ceased. However, since seedlings are generally fire-resistant and regeneration is so effective, it is probable that a fire-free period will enable the species composition to mirror that reported elsewhere and remain relatively constant. This is being investigated at present with re-surveys following the termination of the fire trials. The control plots illustrate the likely form of the vegetation structure and plant composition with fire elimination although the greatest growth was reported in the less frequently burnt plots.

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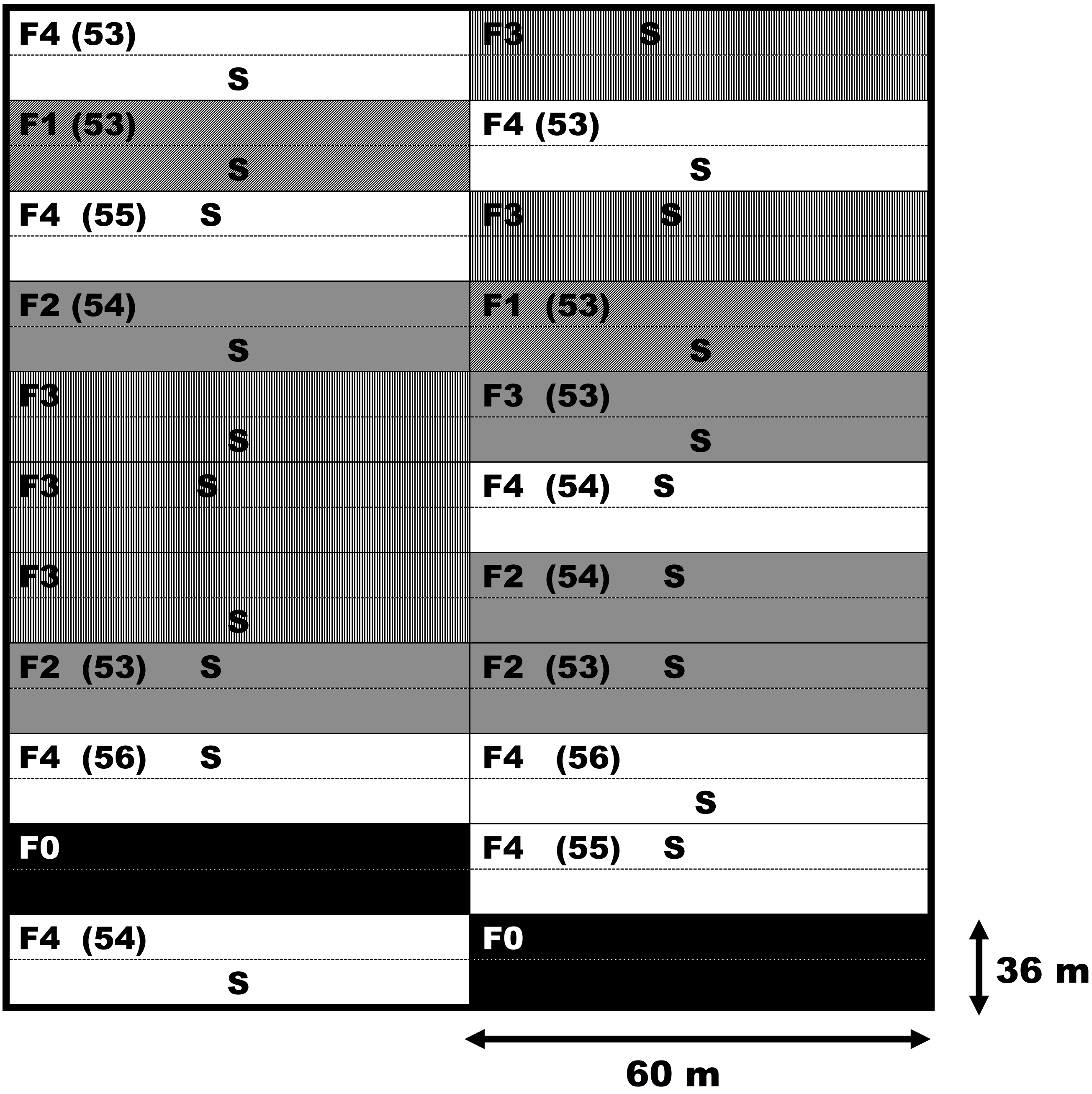
Figure 1 Marondera Site Plan



Key

FO	No burning
F1	Annual burn
F2	Burn every 2 years
F3	Burn every 3 years
F4	Burn every 4 years

S = Cut and burned

Figure 2 Basal area and stocking density in 2007 for live trees > 20 cm DBH

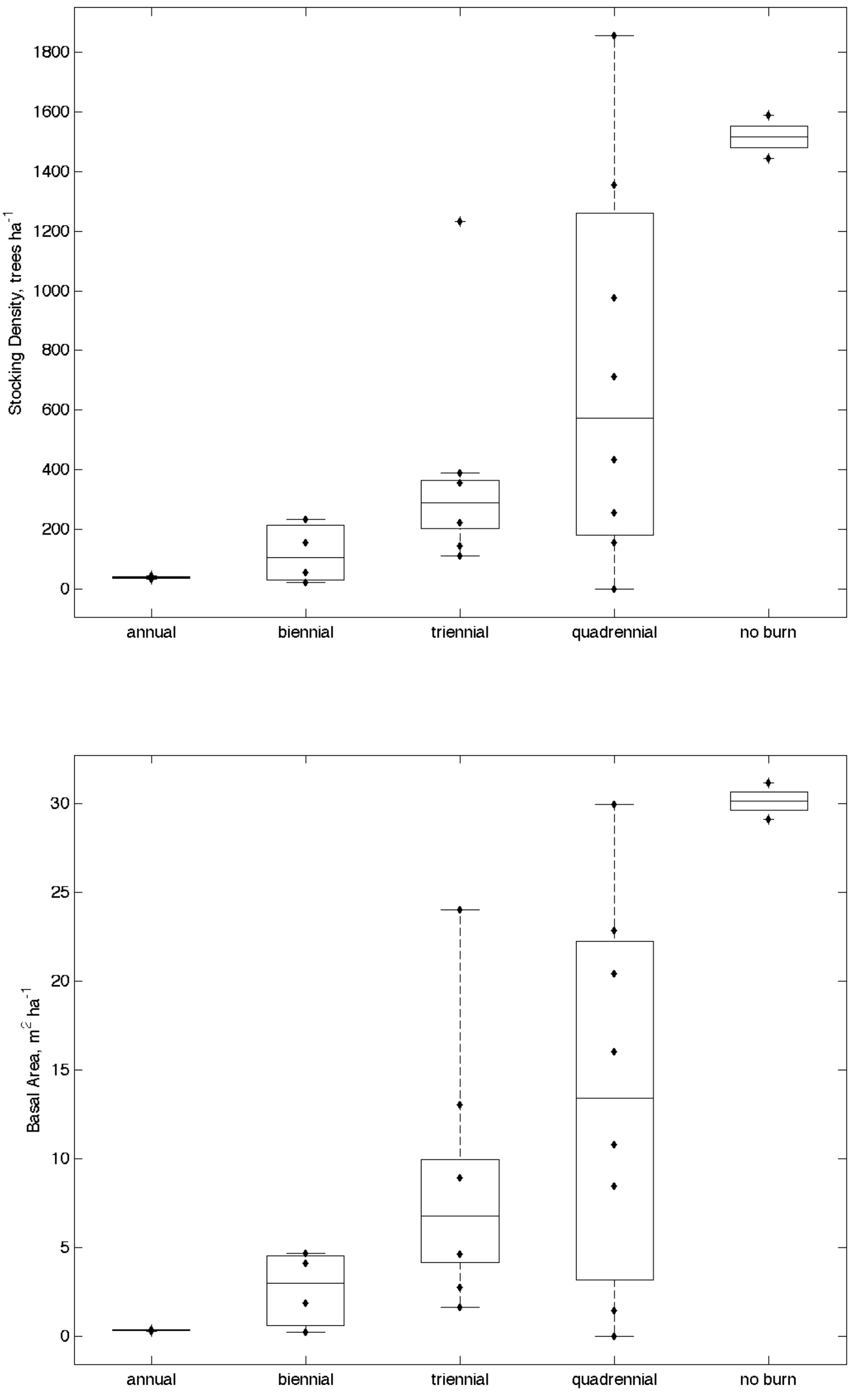


Figure 3

Fig 3 a

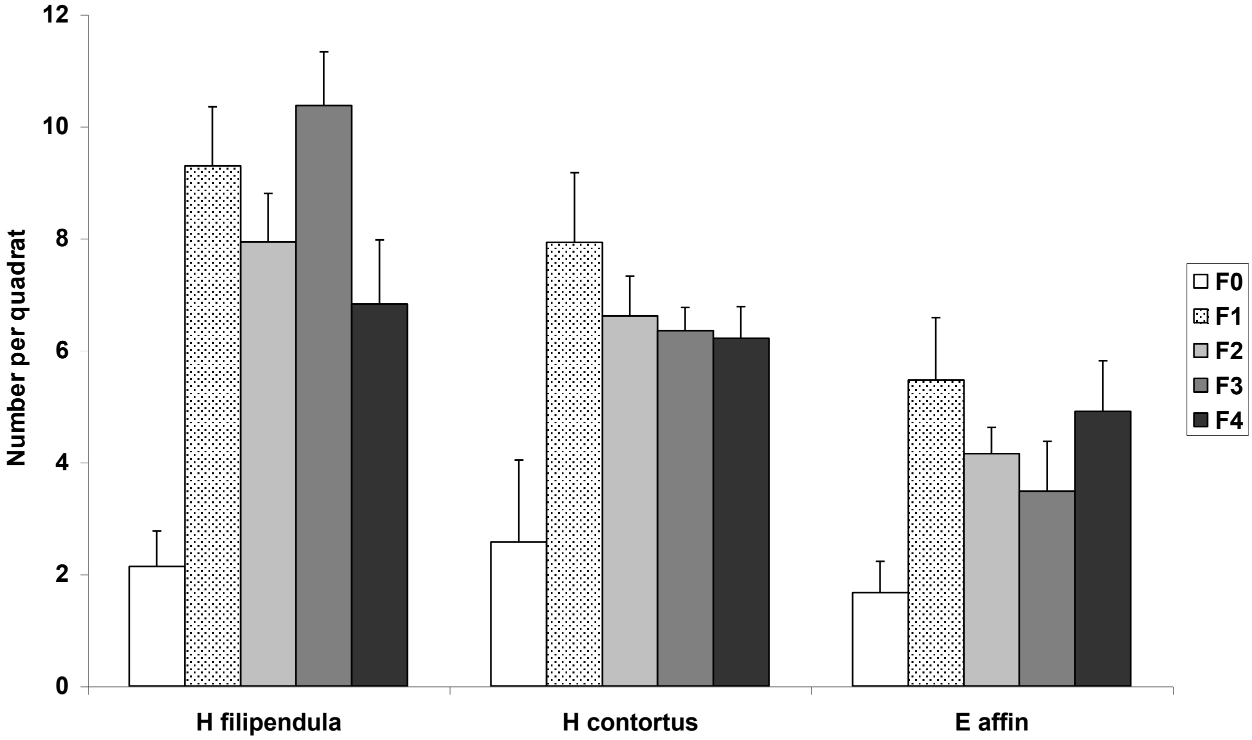


Fig 3 b

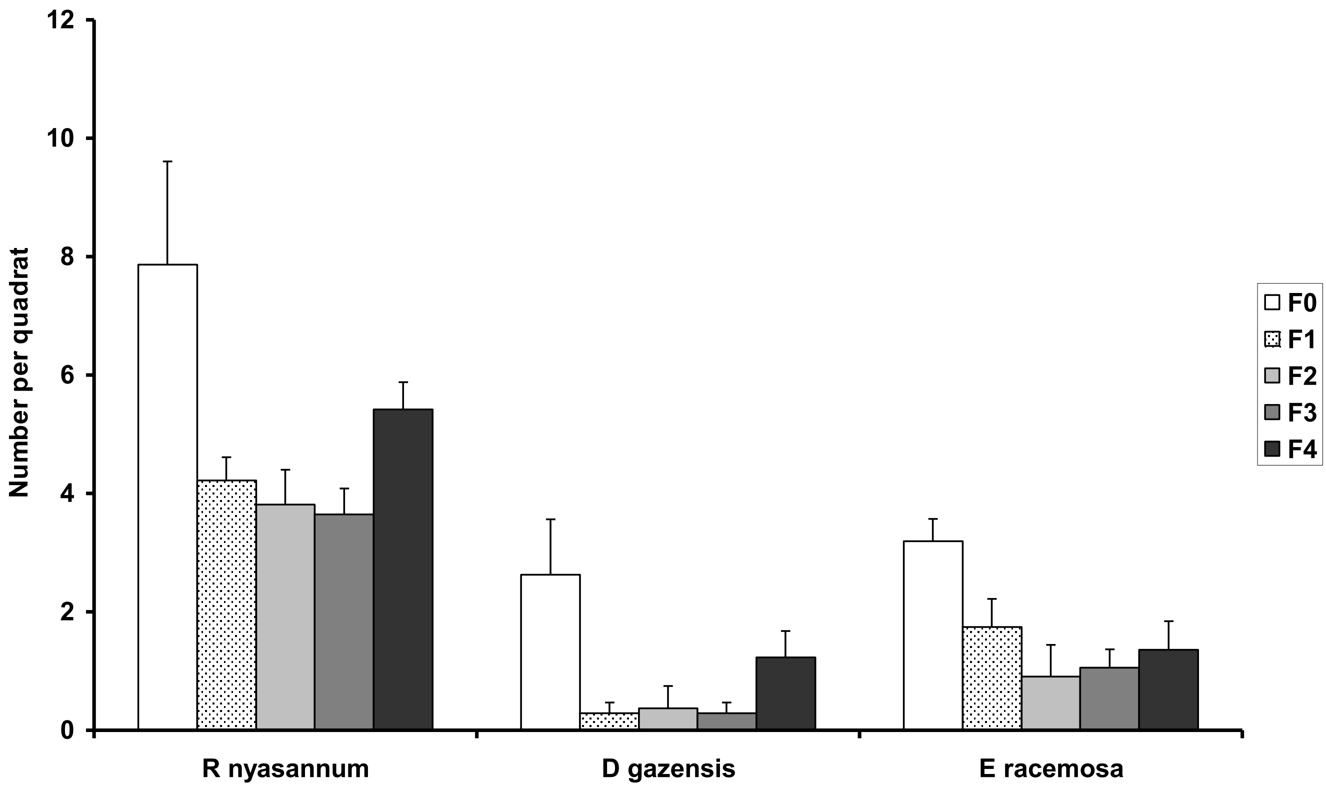


Table 1: Effect of frequency of burning on tree height in1962 and 2007

Fire Treatment	Average Tree Height (m)	
	1962	2007
F0 (protected)	2.83	15.0
F1 (annual burn)	0.18	6.7
F2 (biennial burn)	0.46	11.3
F3 (triennial burn)	0.88	11.2
F4 (quadriennial burn)	0.27	10.6
Significance	P<0.01	P=0.46

Table 2: Effect of frequency of burning on numbers of three grass species

Fire Treatment	*Trachypogon spicatus*	*Andropogon spp*	*Crasspedhorachis qfricana*
FO	1.13	1.01	2.56
FI	2.24	2.85	3.57
F2	1.02	1.60	7.96
F3	0.33	0.20	5.29
F4	1.93	4.82	3.74
Significance	*	*	*
SED	0.63	0.91	1.34
LSD	1.34	1.96	2.87

*significant at $p < 0.05$

Table 3: Effect of frequency of burning on numbers of three selected species

Fire Treatment	Sedges	*Rhytichelytrum repens*	*Nidorela uricrata*
FO	3.64	1.89	3.95
FI	0.69	1.93	1.87
F2	1.53	1.59	2.99
F3	1.12	2.74	4.75
F4	1.71	1.76	3.49
Significance	NS	NS	NS