Reconstruction of a fire regime using MODIS burned area data: Charara Safari Area, Zimbabwe

by
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DECLARATION

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ABSTRACT

Current efforts to address Zimbabwe's decade long veld fire crisis has partly been hindered by a lack of financial resources and fire data. This study illustrates the potential of using the MODIS burned area product as an alternative cost- and time-effective method for reconstructing historical fire records in Zimbabwe. Two MODIS burned area products were evaluated, namely the MCD45A1 and WAMIS (Meraka Institute's MODIS burned area product). Both products yielded similar levels of accuracy despite the difference in algorithms. However, it is assumed that at certain thresholds, either in tree cover or fire intensity, WAMIS ceases to map fires as accurately as the MCD45A1. Ten years of fire data for Charara Safari Area (CSA) was extracted from the MCD45A1, and used as a basis to establish six parameters: fire incidence, extent, seasonality, fire size, frequency and fire return interval (FRI). It was observed that approximately 50% of CSA burned annually, with an average of 132 fires occurring every year. Although there was no overall increase or decrease in the extent of area burned over the 10 year study period, an increasing trend in fire incidence was noted. Through an assessment of effective fire size, it was established that more fires in CSA were gradually becoming smaller in size, while the extent of area burned remained fairly constant. Hence, the increase in fire incidences and lack of a corresponding increase in area burned. This study was also used to identify areas in the fire regime that may be a potential ecological risk to the miombo woodland in CSA. Three points of concern were revealed: firstly, a high prevalence of late season fires was observed in the northern bounds of CSA. Secondly, 64.2% of the total area burned in CSA burned between 6 and 10 times over the 10 year period, and lastly, 85% of the total area burned over the period 2001 and 2010 had a FRI of less than 2 years. The combination of late season fires, high fire frequency and short FRI in CSA is indicative of possible alterations in the state of the miombo woodlands, which may have negative socio-economic implications on CSA and its surrounding communities. This study has demonstrated that the MCD45A1 is a useful source of much needed fire information for Zimbabwe. Therefore, the possibility of integrating methods employed in this study into the current collection of fire data should be given due consideration.

Key words: *MODIS burned area product, miombo, fire regime, MCD45A1, WAMIS.*

OPSOMMING

Huidige pogings om Zimbabwe se dekade lank veldbrand krisis aan te spreek is gedeeltelik belemmer deur 'n gebrek aan finansiële hulpbronne en vuurdata. Hierdie studie illustreer die potensiaal van die gebruik van die MODIS verbrande area produk as 'n alternatiewe koste-en tyd-effektiewe metode vir die rekonstruksie van historiese vuurrekords in Zimbabwe. Twee MODIS verbrande area produkte is geëvalueer, naamlik die MCD45A1 en WAMIS (Meraka Instituut se MODIS verbrand area produk). Beide produkte het soortgelyke vlakke van akkuraatheid opgelewer ten spyte van die verskil in die algoritmes. Dit word egter aanvaar dat op sekere drempels, óf in die boom bedekking, of brandintensiteit, WAMIS brande minder akkuraat karteer as die MCD45A1 produk. Tien jaar van vuurdata vir Charara Safari Area (CSA) is uit die MCD45A1 data onttrek, en gebruik as 'n basis om ses parameters vas te stel: vuurvoorkoms, omvang, seisoenaliteit, vuurgrootte, frekwensie en tyd tussen die terugkeer van vuur na ‘n spesifieke plek (nl. FRI). Dit is waargeneem dat ongeveer 50% van die CSA jaarliks gebrand word, met 'n gemiddeld van 132 brande wat elke jaar voorkom. Daar was nie 'n algehele toename of afname in die omvang van die totale verbrande area oor die 10 jaar studietydperk nie. Maar 'n toenemende neiging in die vuurvoorkoms was wel opgemerk. Deur middel van 'n assessering van effektiewe vuurgrootte, is daar vasgestel dat meer kleiner brande in CSA voorkom, terwyl die omvang van die verbrand area redelik konstant gebly het. Dus was daar ‘n toename in die aantal vuurvoorvalle al was daar nie 'n ooreenstemmende toename in die totale verbrande oppervlakte was nie. Hierdie studie is ook gebruik om gebiede in die vuurregime te identifiseer wat 'n potensiële ekologiese risiko vir die miombobosveld in CSA inhou. Drie punte van kommer word geopenbaar: eerstens, 'n hoë voorkoms van laatseisoen brande is waargeneem in die noordelike grense van CSA. Tweedens, 64,2% van die totale verbrande oppervlakte in die CSA brand tussen 6 en 10 keer bine die 10-jaar periode. Laastens, 85% van die totale verbrande oppervlakte oor die tydperk 2001 tot 2010 het 'n FRI van minder as twee jaar. Die kombinasie van laatseisoen brande, hoë vuurfrekwensie en kort FRI in CSA is 'n aanduiding van moontlike veranderinge in die toestand van die miomboveld, wat negatiewe sosio-ekonomiese implikasies op die CSA en die omliggende gemeenskappe kan uitoefen. Hierdie studie het getoon dat die MCD45A1 'n nuttige bron van broodnodige vuur inligting vir Zimbabwe is. Daarom, moet die moontlikheid van die integrasie van die metodes wat gebruik word in hierdie studie in die huidige versameling van vuurdata behoorlike oorweging gegee word.

Sleutel woorde: *MODIS gebrande area produk, miombo, vuur regime, MCD45A1, WAMIS*.

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ACRONYMS

ATSR	Along-Track Scanning Radiometer
AVHRR	Advanced Very High Resolution Radiometer
BA	Burned area
CA	Consumers accuracy
CSA	Charara Safari Area
DMSP	Defense Meteorological Satellite Program
EC	Error of commission
EMA	Environmental Management Agency
EO	Error of omission
FRI	Fire return interval
GES-DISC	Goddard Earth Sciences Data and Information Services Center
GIS	Geographical Information Systems
GOES	Geostationary Operational Environmental Satellite
GPCP	Global Precipitation Climatology Project
LP DAAC	Land Processes Distributed Active Archive Centre
MIR	Middle Infrared
MODIS	Moderate Resolution Imaging Spectroradiometer
MRT	MODIS Reprojection Tool
NASA	National Aeronautics and Space Administration
NFPS	National Fire Protection Strategy
NIR	Near Infrared
NOAA	National Oceanic and Atmospheric Administration
OLS	Operational Linescan System
PA	Producers Accuracy
QA	Quality Assessment
RMS	Root Mean Square
SANSA EO	South African National Space Agency Earth Obsrvation
SDS	Science Data Sets
SEVIRI	Spinning Enhanced Visible and Infrared Imager
SWIR	Short wave infrared
TOVAS	TRMM Online Visualization and Analysis System
TRMM	Tropical Rainfall Measurement Mission
UTM	Universal Transverse Mercator

VI	Vegetation index
VIRS	Visible and Infrared Scanner
WAMIS	Wide Area Monitoring Information System
WGS	World Geodetic System
WRS	Worldwide Reference System
WWF	World Wildlife Fund

CHAPTER 1: GENERAL INTRODUCTION

1.1 BACKGROUND TO THE STUDY

Fire in Africa has been a common and natural occurrence for millions of years, and is considered to have significant influence on the maintenance, distribution and function of fire-prone systems such as savanna ecosystems (Bond & Keeley 2005). Fires are deemed to be a necessary occurrence in the savannas as they are crucial in maintaining the tree-grass coexistence that typically characterises savanna ecosystems (Bond & van Wilgen 1996). It is estimated that 1.7 million km^2 (17% of the land area) of sub- Saharan Africa burns every year (Chigurah & Jerie 2010), and most of these fires are considered to be anthropogenic in nature (Bond 1997, Laris 2002, Eriksen 2007). According to Chidumayo (1997), the current use of fires today has changed very little from those of our ancestors. As was done centuries ago, fires today are used to clear bushes and undergrowth around homesteads, prepare land for cultivation, collect honey and make charcoal (Chidumayo 1997), to encourage grass growth for livestock, manage pests such as ticks, and also in hunting (Frost 1996).

African savannas have been observed to be fire tolerant due to their capacity to regenerate through coppicing (Trapnell 1959) and burying their seeds under soil during the fire season (Madwecka-Kornas 1980). However it is suggested that frequent burning in the savannas may lead to the destruction of vegetation (Bucini & Lambin 2001), land degradation and loss of biodiversity (Laris 2005). In addition, the large scale nature of burning in African savannas results in the production of large amounts greenhouse gases and aerosols which have a significant impact on global atmospheric chemistry and climate (Crutzen & Andrea 1990). As a result, there has been a growing need for the rapid and low cost provision of data on fire occurrence around the globe. To date, remote sensing systems offer the only means to meet this demand (Laris 2005). And to that effect, remotely sensed data has been integrated into the mapping and monitoring of fires over the years (Eva & Lambin 1998a, Barbosa et al. 1999a, Justice et al. 2002, Silva et al. 2005). Global burned area data sets have also been produced and made available online, for example, the Moderate Resolution Imaging Spectroradiometer (MODIS) burned area product (MCD45A1) (Roy 2005), GlobCarbon (Plummer et al. 2006) and L3JRC (Tansey et al. 2008).

As a result, remote sensing data, combined with Geographical information Systems (GIS), has been employed to map and quantify different aspects of a fire regime (fire frequency, intensity, severity, seasonality, etc.) in various studies (Heinl 2005, Syphard et al. 2009, Hongxiao 2010). As explained by

Archibald et al. (2010a), mapping fire regimes is useful on a number of levels. It provides information for land managers and policy makers, particularly with regards to issues such as ecological sustainability and climate change (Zhan et al. 2002). It is also valuable to researchers who aim to understand the impact of human population and land use practices on fire occurrence, and also in supplementing and testing existing knowledge in fire patterns.

1.2 PROBLEM STATEMENT

Fires in Zimbabwe, as in many other countries in sub-Saharan Africa, are a common occurrence throughout the rural areas. These fires burn from the onset of the dry season, around May and June, up to November as the rainy season sets in (WWF 2001). Fires are considered to be necessary in maintaining the natural diversity of Zimbabwe's vegetation, and are generally used to manage natural resources such as wildlife and forests in gazetted and communal lands (Shumba 2001, WWF 2001). However, it has been reported that since around 2000 there has been a general increase in the incidence of uncontrolled veld fires in Zimbabwe. Phiri et al. (2011) have cited land use changes brought about by the Fast Track Land Reform Program in 2000 as being one of the main causes for the upsurge in veld fire incidences. It is suggested that the sudden influx of people in areas previously sparsely populated led to the increase in anthropogenic activities which consequently resulted in an increase in fire incidences. Many of the fires were set while clearing land in preparation for cultivation and also by poachers using fires to trap animals (Matamera 2011). Other factors that have been attributed to the outbreak of uncontrolled fires in Zimbabwe include: the lighting of fires on the roadsides by resting travellers, careless throwing cigarettes into the bushes, makeshift rubbish dumps around homesteads, hunting camps and mines, the escape of fire during honey collection and annual land preparations (WWF 2001).

According to Tsiko (2006), veld fires have become the most widespread ecological disturbance in Zimbabwe leading to the loss of lives, property and wildlife. Uncontrolled fires in Zimbabwe have resulted in the destruction of forest plantations, particularly in the Eastern Highlands (Nkomo & Sassi 2009). As reported by Runyowa (2012), fires destroyed 9586 ha of timber, constituting 12% of Zimbabwe`s pine population, between July and November in 2011. The extent of the area affected was equivalent to what would have been harvested over a three year period. Pastures crucial for the cattle restocking program have also been greatly reduced by fires. According to allafrica (2006), veld fires burned nearly 10 925 351 hectares of land in 2004 and more than 11 504 947 hectares in 2005. Reports of veld fires destroying maize and wheat fields suggest that food security may also be compromised in the long term. Given the extent of damage caused by the increased occurrence of veld fires in Zimbabwe, it is generally agreed that these fires are a significant threat to the country`s economic recovery (Runyowa 2012).

While it has been acknowledged that the increased frequency of uncontrolled fires is a cause for concern, much attention has been focused on the immediate economic impacts, such as loss of crops, livestock and property. Little consideration has been given to the possible long-term, ecological ramifications of these fires on natural ecosystems such as the miombo woodlands. The miombo is a type of southern African savanna woodlands Africa that is dominated by the tree species *Brachystegia*, *Julbernadia* and *Isoberlinia* (Frost 1996). Though the miombo holds little value in terms of commercial timber, it significantly contributes to the livelihoods of rural communities throughout Zimbabwe, through the provision of firewood, building material, foliage for grazing, medicinal plants, honey, wild fruits and mushroom (Shumba 2001). The miombo woodland also covers approximately 45.2% of the country`s total land area, in which 27 of Zimbabwe's small protected areas are located (Olsen et al. 2001). While there is a general agreement that the miombo is a fire dependent and tolerant ecosystem, some researches have shown that an altered fire regime i.e. increased fire frequency, intensity, extent and late season fires, has the capacity to alter the composition and structure of the miombo, as well as convert the woodland to grassland, (Nefabas & Gambiza 2007, Gandiwa & Kativu 2009, Masocha et al. 2011, Ryan & Williams 2011). Therefore, it is worthy to investigate the potential risks the veld fire crisis in Zimbabwe may have on the miombo.

In response to the veld fire crisis, the Ministry of Environment and Natural Resource Management launched the National Fire Protection Strategy (NFPS) in 2006 (Phiri et al. 2011), which resulted in a series of fire awareness campaigns across the country, while the Environmental Management Agency (EMA) of Zimbabwe was tasked with the responsibility of fire detection and rehabilitation of fire affected areas. Part of the focus of the NFPS is to collect fire statistics; however, it was found by Chigurah & Jerie (2010) that there were large discrepancies in the reporting of fire incidents by EMA staff. This was attributed to immobility of EMA staff due to the limited availability of motor vehicles and short fuel supplies. It was also observed that fires which occurred in inaccessible areas were left unrecorded, while there was a general tendency by members of the public not to report fire incidences to EMA offices. The lack of fire records thus diminished the effectiveness of the fire management initiatives, as they were based on inaccurate accounts of fire incidences.

The nature of the veld fire crisis, coupled with the lack of financial resources and ineffective collection of fire statistics, creates a need for the development of a timeous and inexpensive system that can monitor fire occurrence in Zimbabwe, as well as fill in the gaps in existing fire records. Therefore, this study proposes the use of remotely sensed data in addressing the current need for fire statistics in Zimbabwe.

1.3 JUSTIFICATION OF THE STUDY

The main motivation of this study is to introduce an alternative method of collecting fire data to meet the urgent demand for fire information in Zimbabwe. The few studies that have analysed fire occurrence in Zimbabwe have been based, primarily, on fire statistics derived from a synthesis of information extracted from questionnaires targeted at key informants such as local chiefs, village heads, police officers and EMA, Forestry Commission and agricultural extension officers (Svotwa et al. 2007, Nkomo & Sassi 2009). The accuracy of such data may be greatly limited, by either inaccessibility of fire areas or an unwillingness to report fires. A study analysing veld fires using MODIS active fire data, MOD14A2 and MYD14A2 (Giglio et al. 2003), was also carried out by Chigurah & Jerie (2010) to demonstrate the advantages of using remote sensing and GIS in fire assessments. However, as explained by de Klerk (2008), the MODIS active fire data is designed to map actively burning fires during the period of satellite overpass and is therefore incapable of detecting all the fires that may occur over a given area, although it does provide valuable information on the spatial and temporal distribution of fires. However, the MODIS burned area product used in this study detects burned areas through a time series of daily surface reflectance data, and is therefore able to monitor fires that occur throughout the day in a given area (Justice et al. 2006). Also, the MODIS burned area product offers the opportunity to not only assess fire incidence, but also provides information on the extent of area burned, thus allowing other aspects of a fire regime such as fire extent, seasonality, fire size and fire return intervals to be explored (Archibald et al. 2010a). This in turn will facilitate the theoretical investigation of the potential risks posed by specific aspects of the fire regime towards the sustainability of the miombo.

The MODIS burned area product was considered a suitable alternative source of fire data in this study because it is available to the general public online at no cost, and therefore ideal, given the financial constraints affecting current fire management initiatives in Zimbabwe (Nkomo & Sassi 2009). The MODIS burned area product is particularly attractive because it is the only currently available global data set that map fires at 500m spatial resolution (Roy 2005), and therefore maps burned areas with better accuracy than the coarser 1km spatial resolution products e.g. the GBA-2000 burned area product by Tansey et al. (2004).

1.4 AIM OF STUDY

The aim of this study was twofold: firstly to explore the possibility of using remotely sensed data and GIS applications as an alternative cost- and time-effective method for reconstructing a 10 year historical fire record. And secondly, to identify potential ecological threats to the miombo woodland, posed by specific elements in the fire regime. In order to achieve these two main goals, a number of specific objectives were outlined.

1.5 OBJECTIVES

- Evaluate the mapping capability of two MODIS burned area products against burn scars mapped from Landsat imagery, and decide on the appropriate product to use for the study.
- Identify individual fires extracted from burned area data.
- Establish and analyze the fire regime focusing on fire frequency, extent, seasonality, fire size and fire return interval.
- Identify areas in the fire regime which are a potential risk to natural state of the miombo in the study area.

The research design employed to achieve the objectives of this study is illustrated in Figure 1.1.

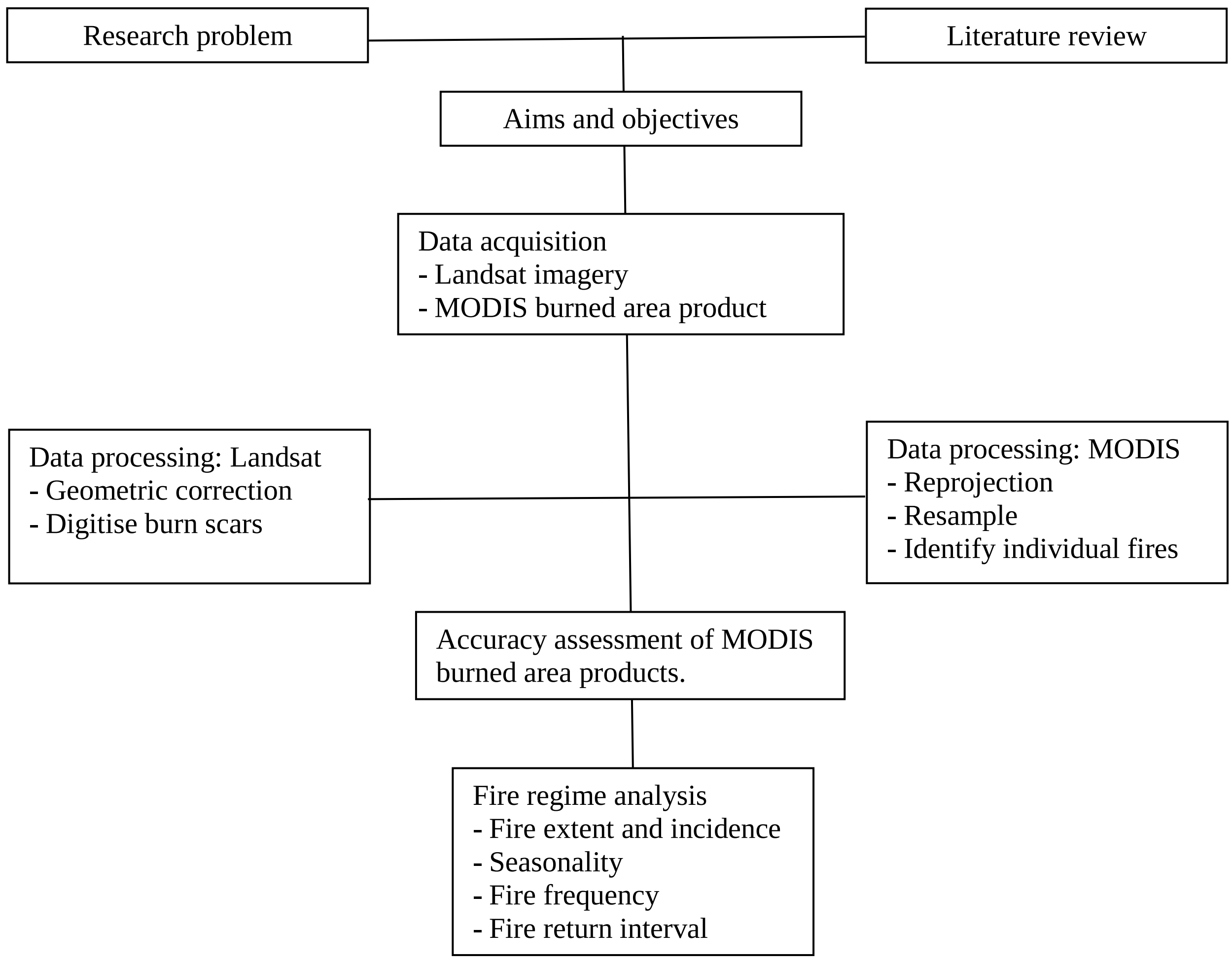

Figure 1.1 Research design

1.6 STRUCTURE OF THE STUDY

This document is comprised of six chapters: Chapter 1 is the introductory section of the study. This is followed by Chapter 2, which is a review of various literature concerning three central aspects of the study i.e. miombo ecology, fire regimes and burned area mapping. Chapter 3 consists of a detailed description of the study area as well as the data and methods utilized to carry out the study. Chapters 4 and 5 are a discussion of the results pertaining to the two main goals of the study. Chapter 6, which is the final chapter, gives a summary of the findings from the study and recommendations on key areas that require further study.

CHAPTER 2: LITERATURE REVIEW

2.1 MIOMBO ECOLOGY

2.1.1 Distribution and composition of miombo woodland

The term miombo is widely used to describe the savanna woodlands of Southern Africa that are dominated by the genera *Brachystegia*, *Julbernardia* and *Isoberlinia* (Ryan 2010). Miombo woodland is the most widespread vegetation type in the Zambezian phytoregion, covering a total of ten countries in central and southern Africa, lying between 3^0 and 26^0 South, and extending over 377 million hectares of land (White 1983). Miombo woodlands are also the most extensive woodland type in Zimbabwe covering most parts of the country`s watershed (Olsen et al. 2001).

The composition and structure of the miombo woodlands looks relatively uniform over large regions, this is due to the similarities in the physiognomy of the dominant canopy trees, which typically reflects their origins in the *Caesalpinioideae* family (Frost 1996). Woody plants account for 95 to 98% of the above ground biomass in undisturbed stands, while the herbaceous layer makes up the remainder (Martin 1974, Malaisse 1978). The woodlands typically consist of pagoda- or umbrella shaped trees in the upper canopy; a few scattered layer of trees in the sub-canopy; an intermittent mixing of shrubs and saplings in the understory; and patchy layers of grasses, forbs and suffrutices (Frost 1996). Differences in the composition and structure of the miombo are however more apparent at local scales, and have been attributed to a number factors. These include: past and current land uses and other anthropogenic disturbances (Robertson 1984), the effects of fire (Kikula 1986); wildlife impacts (Anderson & Walker 1974), geomorphic changes in landscape (Cole 1986) and edaphic factors, particularly soil moisture and soil nutrients (Campbell et al. 1988).

A number of sub-types, based on the dominant species, are found in the miombo woodland. Following Shumba (2001), the most widespread sub-type is the *Brachystegia spiciformis,* found in association with *Julbenardia globiflora* and *Brachystegia boehmii.* However, on the Kalahari sands the *B. spiciformis* is commonly associated with *Baikiaea plurijuga* and *Pterocarpus angolensis*. The second most widespread type is the *B. boehmii,* which usually occurs on escarpments at higher altitudes and is mainly associated with the *Afzelia quanzensis, Kikia acuminata* and *Acacia* species under warmer and drier conditions, while it merges with mopane woodlands at lower altitudes. The third type is *J. globiflora* which thrives at a wide range of altitudes and is often found as forest stands, however it also occurs in association with *Colophospermum mopane, K. acuminata* and *Sclerocarya birrea* at lower altitudes. *Parinari curatellifolia* is the fourth type; it occurs as pure unstratified stands on sandy soils with a high water table. It suggested that the whole central plateau of Zimbabwe was once covered by *P. curatellifolia* which was then invaded by *B. Spiciformis* and has generally been degraded to grasslands as a result of clearing and burning. The fifth type is known as the *Uapaca kirkiana* which occurs as pure stands commonly situated on well-drained soils in frost-free areas (Shumba 2001).

Miombo is often divided into wet and dry miombo based on the 1000 mm mean annual rainfall isohyets (White 1983). Wet miombo occurs where rainfall typically exceeds 1000 mm per year. It stretches over eastern Angola, northern Zambia, south-western Tanzania and central Malawi. The wetter miombo is floristically rich and almost all the miombo dominants, such as *B. floribunda, B. glabernima, B. taxifolia, B. wangermeeana, and Marquesia macroura*, are found to be present (Frost 1996). Canopy height in the wet miombo is often greater than 15 m as a result of deep moist soils, while the associated vegetation includes dry evergreen forest and thicket, swamp forest, evergreen riparian forest and wet dambos. Dambos are broad, grassy depressions which can cover up to 40% of the miombo landscape (Desanker et al. 1997).

Dry miombo occurs where the mean annual rainfall is less than 1000 mm, and is mainly found in Malawi, Zambia and Zimbabwe. This miombo is floristically poor with *B. floribunda, B. glabernima, B. taxifolia, B. wangermeeana*, and *M. macroura* absent or very local in occurence, with a canopy height of less than 15 m (Frost 1996). *B. speciformis, B. boehmii* and *J. globiflora* tend to be the only dominants present, while associated vegetation includes dry deciduous forest and thicket, deciduous riparian forest and dry dambos (Desanker & Prentice 1992).

According to Goldberg (2001), the miombo ecoregion falls under the seasonal tropical climate, where the average annual rainfall ranges between 800 and 1 400 mm. Most of the rainfall is concentrated in the hot summer months from November through March or April, while the following months are characterised by an intense winter drought that can last up to six months (Werger & Coetzee 1978). Depending on the elevation the mean maximum temperatures range between 21°C and 30°C, while the mean minimum temperatures range between 15°C and 21°C, the hottest temperatures are experienced in the lowland areas and the region is virtually frost-free (Goldberg 2001). Miombo trees are intolerant to absolute minimum temperatures of less than -4°C, this may be the factor that limits miombo woodland to the Zambezian phytoregion (Werger & Coetzee 1978). Following Barnes (1998), most of the miombo region is found at elevations ranging between 1000 and 1500 m, and is characterized by flat or undulating plains, dominated by intrusive granites and gneisses that frequently rise up above the woodland as dwalas or inselbergs. Miombo occurs on well drained soils that are derived from the African and post-African planation surfaces that form the Central African plateau (Hopberg 1986). The soils are highly weathered, acidic, and nutrient poor (Barnes 1998), and consequently unsuitable for cultivation, which accounts for why miombo regions have been relatively under populated in the past. In recent years, however, rural populations have expanded into the miombo regions, and with the exception of protected areas, there is little undisturbed miombo left (Chenje & Johnson 1994). Soil texture in miombo woodland is sandy loam, sand clay loam and sand clay, the clay content increases with depth due to the eluviation process. Soil colour varies from shades of brown in the top soil (0-30 cm) in well drained sites, to reddish orange in the bottom soil in poorly drained sites, (Chidumayo 1993). Shallow stony soils are common along the escarpment and around inselbergs (Barnes 1998). Under alkaline (pH>7) and water logged conditions, miombo is succeeded by mopane or munga woodland and swamp or riparian forest (Chidumayo 1997).

2.1.2 Miombo woodland regeneration.

Regeneration in miombo has three phases: (i) Seed production, dispersal and germination (ii) Seedling development (iii) Vegetative regeneration (Fanshawe 1971, Grundy et al. 1994, Chidumayo 1997).

Seed production, dispersal and germination: Fruit and seed dispersal in miombo woodland is concentrated in the late dry season from August to November, but in a few species such as *lsoberlinia angolensis*, seed dispersal continues into the early rainy season. Dispersal mechanisms among miombo trees vary. For example, the pod is the most common fruit type among the canopy species (*Brachystegia, Isoberlinia and Julbernadia species*) and these seeds are consequently dispersed by an explosive pod. Seeds from the *Albizia* and *Pterocarpus* species are dispersed by wind, while animals, mainly birds, are the common dispersal agents for fleshy fruit which is common in understorey and shrub species (Chidumayo 1997).

Seedling development: The majority of the miombo trees and shrubs have seeds that germinate immediately after dispersal, given that there is adequate water supply (Ernst 1988, Chidumayo 1991, 1992a). Droughts, water stress and fires can increase tree seedling mortality, however, different species resist these environmental stresses to varying degrees (Trapnell 1959, Strang 1966, Ernst 1988, Chidumayo 1991, 1992a, 1992b). For instance, on the one hand *U. kirkiana* seedlings die from drought, while *I. angolensis* seedlings survive both drought and fire. Likewise *B. speciformis* seedlings are fire intolerant, even in the second year of growth, while majority of *J. paniculata* seedlings are resistant to fire. When fire kills the above ground shoots of the seedlings, the seedlings respond by producing sucker shoots through their roots the following growing season. During the development phase most of the seedlings in the miombo woodland experience a prolonged period of successive shoot die-back. Shoot- die back is a result of water stress or fire during the dry season (Trapnell 1959, Boaler 1966, Chidumayo 1991, 1992a, 1992b). However shoot die back does not always lead to seedling death, if the root survives, a new shoot will be produced the following growing season. This is why seedlings of miombo trees allocate more photosynthetic biomass to root than shoot growth during the establishment phase. Such a mechanism is an important adaptation in an environment that is prone to regular annual fires and drought (Chidumayo 1997).

Vegetative regeneration: In almost all miombo trees, once the above ground parts have been removed or killed, the stumps and roots respond by producing sucker shoots (Trapnell 1959, Lees 1962, Boaler & Sciwale 1966, Hood 1972, Strang 1974, Banda 1988, Chidumayo 1989). Sucker shoots emerge from buds which develop on the roots and stem bases. Stumps produce many sucker shoots, but during the establishment phase the number of shoots is reduced by intershoot competition, therefore only dominant shoots contribute to the next generation of regrowth miombo. Consequently, stem density per plant declines slowly with age of regrowth. Fire, however, may either slow or accelerate this domination process. When a destructive fire occurs before dominant shoots achieve safe height to escape mortality the sucker shoot domination process reverts back to initial stage, and stumps respond by producing an equal or larger number of replacement shoots (Chidumayo 1988).

2.1.3 Fire in miombo woodlands.

Fires in the miombo region are mainly fuelled by the senesced grass layer which can burn throughout the dry season, with an average fire return interval of 2-4 years (Scholes et al. 1996, Barbosa et al. 1999b, Mouillot & Field 2005). Fire has been used for a wide range of purposes, including hunting, land clearing, rejuvenating grazing lands, and clearing paths around houses to reduce wildlife hazards since as far back as the Iron Ages (Chidumayo 1997, Eriksen 2007, Stronach 2009). The composition and structure of the miombo woodland is largely determined by the occurrence of fires. According to a number of fire experiments carried out in Zimbabwe (Furley et al. 2008) and Zambia (Trapnell 1959), the exclusion of fires in the miombo woodlands led to the formation of a closed canopy woodland and eventual development of a forest, while annual fires resulted in the eradication of all woody vegetation.

The majority of miombo woodland fires occur during the hot, dry season. A study carried out by Chidumayo (1993) to monitor the occurrence of fire at four dry miombo sites in central Zambia from

1990-1993, showed that out of the 13 fires that occurred at the sites two were recorded in August, five in September and six in October. Long term data on forest plantation fires in the Zambian Copper belt also supports the observation that the hot dry season, which stretches from August to October, has the highest fire frequency in the miombo region (Chidumayo 1997). Fires are generally more frequent over the months: August to September due to the high temperatures, windy conditions (WWF 2001), and large amounts of dried grass and fine woody material (Desanker et al. 1997).

2.1.4 Miombo adaptation to fire

According to (Chidumayo 1997), the effect of fire on miombo vegetation is dependent on the season and frequency of burning and on the amount of fuel load available. Miombo woodland plants have a number of adaptations to fire. These include the scheduling of phenological phases when the risk of fire is lowest (Malaisse 1974), the ability to resist fire by having vital tissues insulated from high fire temperatures, and the capacity to recover vegetatively when fire has damaged plant tissues (Frost 1996)

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The scheduling of active phenological phases when the fire risk is lowest increases the survival rate and productivity of miombo trees. Many herbaceous, and some woody plants, in miombo lie dormant during the dry season when the risk of fire is greatest (Chidumayo 1997), because actively growing or reproducing plants are more susceptible to fire damage than dormant ones, and so plants schedule growth and reproductive activities when the risk of fire is lowest (Frost 1985).

As observed by Madwecka-Kornas (1980), annual miombo plants usually survive fire as seeds buried in the soil. However, the majority of the trees in miombo do not have soil seed banks, but rather disperse their seeds on the soil surface and in the litter (Ernst 1988), making them more susceptible to damage by fire than buried seeds. On the other hand, some miombo trees protect their seeds from the direct effects of fire by bearing them in indehiscent fruits. For example, the woody fruit of *Gardenia subacaulis*, protects its` seeds from fire and only releases them during the rainy season after the fruit has decomposed (Madwecka-Kornas 1980).

The leaf flush is a very important phase in miombo trees, and its timing determines plant growth and productivity. Most of the dominant tree species in miombo, as well as other trees and shrubs are deciduous and shed their leaves during the dry season (Chidumayo & Frost 1996). The leaf flush phase mainly occurs in October and November ie: early rainy season. However, in a few species, such as *B. speciformis*, *I. angolensis*, *J. globiflora* and *Protea* species, leaf flush starts in August. Species with an early flush are more susceptible to fire damage throughout the hot dry season than those that flush late. For instance, August fires have little effect on *I. angolensis* although September fires reduce the productivity of this species. On the other hand, both August and September fires have little effect on the productivity of *U. kirkiana* because it flushes late (Chidumayo 1993). Therefore, repeated late dry-season fires selectively favour trees that flush late and damage trees that flush early (Chidumayo 1997).

A thick bark and or high wood moisture content contribute to the ability of the miombo to resist fires. Bark thickness is frequently cited as a protective measure against fire (Frost 1985, Stott 1988). A thick bark protects vital inner tissues, such as the meristems and vascular organs, more effectively than a thin bark. Bark thickness among miombo trees varies between species and over time. Large stems have a bark that is 3-5 times thicker than that of small stems, this may explain why saplings and small poles are more vulnerable to fire damage than large stems. *B. africana*, *J. globiflora*, *Monotes species* and *Pseudolachnostylis maproneifolia* have thin bark during the sapling stage and are therefore more likely to be fire sensitive than the other species (Chidumayo 1997).

Based on wood moisture content, miombo trees can be divided into three groups: those with high moisture content (>90%), those with moderate moisture content (60-90%) and those with low moisture content (<60%) (Chidumayo 1997). Miombo trees commonly have moderate moisture content. Since the burning potential of miombo biomass depends on moisture content, species with high moisture content, such as *Protea species*, *Uapaca species* and *Syzgium guineense macrocaarpum*, are less likely to be affected by fire than species with low moisture content, such as *Dichrostachys cineria* and *Swartzia madagascariences*. *D. cineria* is an invasive bush that thrives in the absence of fire, fires in the miombo region therefore control such bush encroachment (Chidumayo 1997).

Most miombo trees have the capacity to recover plant tissue damage caused by fires through coppicing. When the above ground parts of the tree are damaged by fires, the trees are known to coppice from dormant buds located in the branches, stem, or roots (Frost 1996, Kanschik & Becker 2001). The ability to resprout depends on the extent of damage to the apical meristems and the condition of the dormant buds. Absolute top-kill results in vigorous resprouting, however, resprouting is poor in very old or large trees or those with partial top-kill. There are no trees in the miombo woodland that have a combination of thick bark, late leaf flush and high moisture content, which would define a completely fire tolerant species. Nor are there trees with a combination of thin bark, early leaf flush and low moisture content, which would be a typical example of an extremely fire-intolerant species. The only combination that characterizes fire intolerant species in miombo woodland is low wood moisture content and either a thin bark or early leaf flush. These species include *D. cineres* and *S. madagascariensis*. All other species have either a structural or phenological adaptation to fire (Chidumayo 1997).

According to Shumba (2001), the miombo woodland in Zimbabwe generally holds little commercial timber, except for small areas in demarcated forests such as Mafungautsi Forest Reserve. However, the miombo woodlands have a diverse range of uses from watershed protection, provision of soil fertility, grazing and browsing, firewood, edible fruits, mushrooms, caterpillars and timber. Unfortunately, because most of the forests in Zimbabwe have been converted into intensive agricultural areas, the miombo woodland is now very limited in extent and has generally been degraded to grasslands and savanna as a result of clearing and burning, hence it is difficult to locate pristine miombo woodlands in Zimbabwe (Shumba 2001).

2.2 FIRE REGIME

2.2.1 Definition of a fire regime

A fire regime is defined by Gill (1975) and Bond & Keeley (2005) as the frequency, intensity, severity, seasonality, fuel consumption and spread pattern of fires at a given locality. The fire regime is a vital component of miombo ecology because fires maintain the structure and species composition of the woodland. Fires achieve this through the removal of above ground biomass and creation of conditions necessary for regeneration and a tree-grass coexistence (Masocha et al. 2011, van Wilgen et al. 2010). Several authors refer to such fire dependent ecosystems as pyrophytic vegetations (Madwecka-Kornas 1980, Scholes 1997, Trollope & Everson 1999).

2.2.2 Fire and the physical environment

In order to discuss the fire regime further, it is important to first form an understanding of the natural occurrence of the fire itself. The ignition, spread and dying of fires in the semi-arid region is largely dependent on the interaction of four environmental factors: combustion, climate and weather, and vegetation (Hely & Alleaume 2006).

Combustion is a process whereby energy, initially stored in vegetation during photosynthesis, is released as heat when material such as leaves, grass or wood are burnt (Whelan 1995). For combustion to take place, activation energy from an external source e.g. lightning or a human induced source, is required. Trollope (1984) suggests that there are three stages in the process of combustion: (i) preheating, in which fuel (grass or ground litter) is heated, dried and partly pyrolysed before the flames develop; (ii) flaming combustion, which is when flammable hydrocarbon gases, released from the preheated fuel, burst into flames; and (iii) glowing combustion, during which the remaining charcoal burns as a solid without flame, leaving a small amount of residual ash. Flaming combustion results in plant mortality, while glowing combustion is primarily responsible for duff consumption and seedbed preparation.

Topography has a major effect on the flaming combustion phase and on fire ignition. Lightning ignited fires are more common on the hilltops than in valleys or flat landscapes. The mid and upper slope areas are more flammable because they have drier soil conditions and drier vegetation, as a result of better water drainage. Combustion of the dry vegetation will require less water evaporation, therefore, less energy before ignition (Hely & Alleaume 2006). Moreover, fire burning uphill will propagate faster than fire spreading downhill or on flat terrain (Figure 2.1). Alexander (1982) explains that when fire burns uphill, the radiation energy released by the flames augments the preheating of the adjacent fuel which is in close proximity to the flame, due to the steep slope angle. This increases the rate at which the fire spreads.

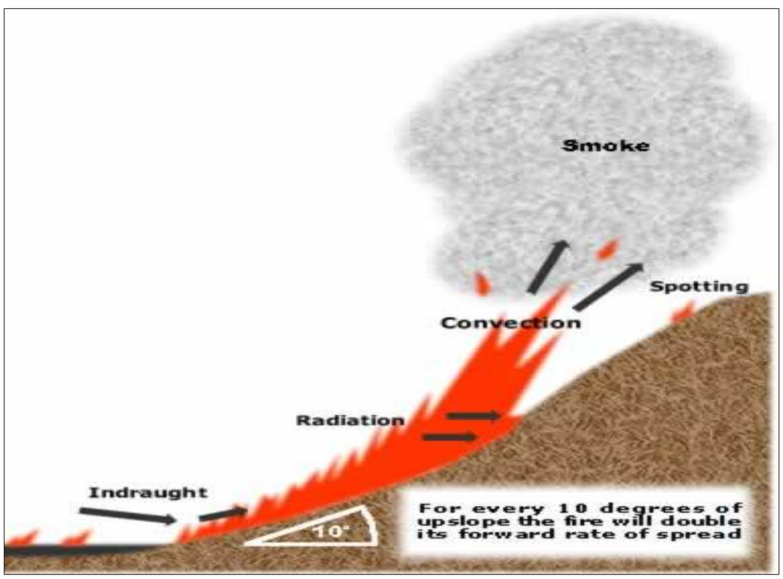

Figure 2.1 Fire burning upslope

Source: Charles Darwin University: Learnline http://learnline.cdu.edu.au/units/sbi263/index.html

Climate and weather have long term and short term effects on fire, respectively. Climate is a significant aspect of the fire regime because it determines vegetation type, fire season and fire frequency. Weather, particularly precipitation, determines the quantity and quality (moisture content) of vegetation and therefore, fuel production (van Wilgen & Scoles 1997, Hely et al. 2003b). Weather also influences fire behavior through the absence or presence of wind. As a result, the combination of topography, weather and climate determines the rate of fire spread (Alexander 1982).

Vegetation is the fuel component within the fire environment. It is directly associated with precipitation quantity, ie: periods of high precipitation induce maximum vegetation production and consequently increases fuel load (van Wilgen & Scholes 1997, Hely et al. 2003b). In the miombo region, vegetation growth corresponds with the rainy season (October to March/April) and starts to dry a few weeks into the dry season, fires may then occur if the minimum fuel load is achieved (Hely & Alleaume 2006). The minimum fuel load in sub-Saharan savannas is approximately 250 to 300 g/m^2 (van Wilgen & Scholes 1997, Hely et al. 2003a). The size of the vegetation matter is also important. Fine vegetation matter will burn more easily than thick woody biomass. In the initial propagation of a fire, larger trunks and boles are usually not considered as fuel, because they tend to be too moist and the bark too thick to ignite quickly. However, if the fire is spreading slowly or burning against the wind, the flames may burn long enough to ignite larger debris, which can burn several hours or days after the fire has died, leaving white ash in place of the consumed trunk or bole (Hely & Alleaume 2006).

2.2.3 Components of the fire regime

The fire regime is described by several spatial and temporal aspects. The spatial aspects are embodied in the fire type and fire extent/area, while the temporal components include fire frequency, fire return interval and the fire season. The spatial and temporal aspects are then combined with fire spread and intensity to produce a fire regime (Gill 1975, Bond & Keeley 2005, Hely & Alleaume 2006).

2.2.3.1 Fire type

Fire type is generally defined by the layer, in the vertical structure of the vegetation, in which the fire is spreading (Hely & Alleaume 2006). For instance, surface fires are fires that burn just above the ground, and are induced by fuels that are close to the surface, such as grass, litter and small shrubs with light fuel load. Surface fires are a common feature in the semi- arid region. On the other hand, crown fires burn in the tree and shrub canopies. These fires occur in areas where there is a high fuel load, allowing the flame to reach the canopy (WWF 2001). Another fire type, that is not particularly common in the dryland region, is the ground fire. Ground fires are slow spreading fires, that burn over long periods within the rich organic layers of soil, usually without flaming combustion. These fires are common in the Okavango delta of Botswana, where water logging prevents the growth of trees but gives rise to peatland build up, which is rich in organic matter. Other fire classifications include backfires that burn upwind and/or downslope and head fires which burn downwind and/or upslope (Bond & Keeley 2005, Hely & Alleaume 2006).

2.2.3.2 Fire extent and area

Fire extent is the most commonly used variable in measuring the impact of fire on vegetation, it is defined as the ratio of the surface area burned by a fire within a given area. Fire area is, however, the total surface area burned within a given area, and is usually expressed in hectares. When analyzing fire extent in the reconstruction of a fire history, it is crucial to select a study area that is larger than the biggest fire extent that has ever occurred within the area. Fire extent is determined by the vegetations` moisture content and fuel quantity, as well as the connectivity between fuel stands and the overall heterogeneity of the area (Hely & Alleaume 2006). Weather also has an impact on fire extent. In a homogenous landscape windy conditions may cause a fire to form a narrow oval shaped burn scar. Under wind free conditions, the burn scar will form a circular perimeter, with the center being the point of ignition (Alexander 1982). However, fire breaks such as roads, humid areas, rivers tend to alter such defined fire extents and shapes (Hely & Alleaume 2006).

2.2.3.3 Fire frequency and fire return interval

Fire frequency is defined by Bond & Keeley (2005) and Hely & Alleaume (2006), as the number of fires that occur within a given area and period of time. While, fire return interval (FRI) or fire cycle is described as the time taken for a fire to reoccur at any one site (Agee 1993, Hely & Alleaume 2006). Fire frequency is determined by (i) the time required to build a load of available fuel from the last fire, and (ii) the amount of potential ignitions. In a natural fire ecosystem, ignition and fuel load variability is almost entirely dependent on the weather prevailing within each fire season (Whelan 1995). The average fire frequency can be estimated and positively related to the amount of vegetation cover, for example, in infertile wet miombo found in Zambia, grasslands may burn annually due to heavy rainfall and high grass fuel load (Chidumayo 1997). At the same time fire frequency can be negatively related to the total amount of precipitation. For example, in the semi-arid savanna of Namibia fires may burn every five years due to high inter-annual variability in rainfall. In such regions, several years may pass before there is sufficient precipitation to build up the minimum fuel load required to sustain a fire (Hely & Alleaume 2006). Despite the scantiness of information on fire frequency, the FRI for the miombo ecoregion has been estimated to be approximately three years by Frost (1996). Other researchers (Scholes et al. 1996, Barbosa et al. 1999b, Mouillot & Field 2005) have also estimated a similar FRI of two to four years.

The ecological importance of fire frequency or FRI is that any alteration in the frequency may lead to shifts in vegetation composition and ecosystem functioning. Changes in fire frequency are largely propagated by human activities. Increased fire frequency in savannas has been attributed to an increase in human induced fires (Trapnell 1959, Trollope 1982, Bond & van Wilgen 1996). In addition, a fire frequency experiment carried out in Marondera`s miombo woodland by Ryan & Williams (2011), found that under a 1 year FRI, the portion of woody biomass within the study was completely eradicated. It was then concluded that regardless of fire intensity, the woody biomass in the study area did not have the capacity to sustain annual burning. In such extreme cases, conversion of woodland to grassland will indeed be observed. On the other hand a decrease in fire frequency or complete protection from fire alters the species composition and vegetation structure of the miombo. The reduction or absence of fire in miombo woodland would result in the recruitment of dominant canopy species, such as the *brachystegia*, into larger class sizes, leading to the development of a closed canopy forest and, consequently, a reduction in grass population (Govender et al. 2006, Ryan & Williams 2011). An increase in fire frequency will result in the invasion of fire tolerant species, some which may be alien to the vegetation community e.g. *Lantana Camara* and *Jacaranda mimosifolia* (Masocha et al. 2011).

2.2.3.4 Fire season

Fire season defines the season in which most fires occur. This season usually occurs during the dry season in semi-arid regions and during periods of drought in more humid areas. The natural fire season pattern has been altered by anthropogenic activities, mainly through the ignition of fires outside the normal fire season (Bond & Keeley 2005, Hely & Alleaume 2006). The length of the fire season tends to be shorter than the actual dry season. This is due to the fact that the early dry season vegetation carries a significant amount of moisture, and therefore the fuel is not dry enough to ignite and sustain a fire. For example, in the African savanna, fire season is concentrated towards the end of the dry season when lightning, from convective storms, ignites extensive areas of dry vegetation (Booysen & Tainton 1984, Stott 2000). However, due to human induced ignitions and land management practices, the current fire season in African savannas spans throughout the dry season (Stott 2000) as illustrated in Figure 2.2.

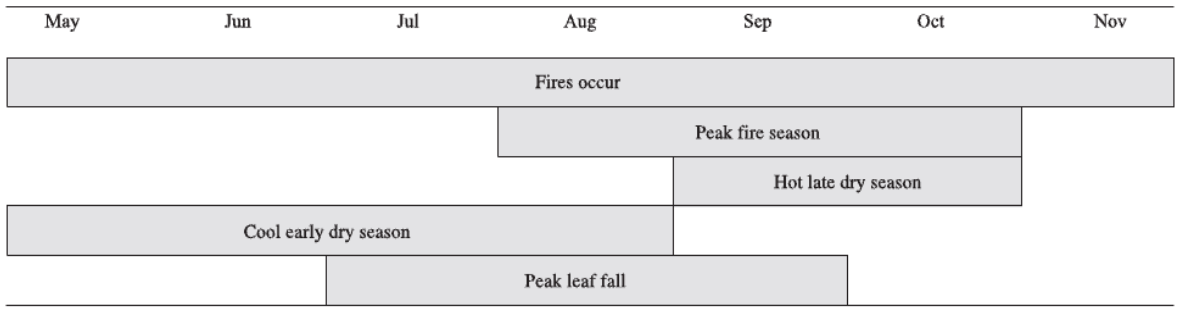

Figure 2.2 Fire season in miombo woodlands

Source: Pereira et al. (2004), modified from Haanpaa (1998).

In Zimbabwe the fire season is classified into two categories: (i) Cool, early season fires and (ii) hot late season fires (WWF 2001). Cool, early season fires occur in the early dry season between the months of May and July, when there is still moisture in the vegetation. Because of the partially wet fuel and benign weather conditions, these fires tend to be cooler (less intense) and slow burning (WWF 2001, Ryan & Williams 2011). In the savanna ecosystems, farmers and pastoralists burn most grasses in the early season (Bucini & Lambin 2001). For example, in Senegal, early season fires are used to initiate regrowth in the perennial grasses (Mbow et al. 2000). The other benefit of early season fires is that they create small patches of burned areas that later act as fire breaks when hotter fires occur in the late dry season (Eriksen 2007). In Mali, herders and hunters use early season burning to create a landscape of burnt and unburned patches, in order to reduce the destructiveness of late dry season fires. Therefore, early season fires can be used as both protective and preventative measures in the savannas` fire environment (Laris 2002). Frequent early season fires result in the formation of a closed canopy woodland, with thickets of less fire tolerant species, which in turn suppresses grass growth in the understory and consequently reduces the rate of fire incidence (Frost 1996).

Hot, late season fires burn towards the end of the dry season from August to October, and are considered to have a damaging effect on the savanna ecosystem (Laris 2002). These fires burn more intensely, and cover vast areas due to the increased availability of dry fuel and windy conditions. In Zimbabwe fire intensities are as high as 500 to 5000 Wm^{-1} during the late dry season, and range between 100 to 300 Wm^{-1} in the early dry season (Robertson 1993). Late season fires are particularly destructive because they tend to be a combination of both ground and crown fires, and due to their high intensity, they have the capacity to completely burn large woody trees in the woodland (WWF 2001, Ryan &Williams 2011). A study carried out in Ndola showed that the highest mortality rates, under a late fire season in miombo woodland, belonged to the mature dominant species e.g *I. angolensis* and *B. spiciformis*, the sapling regeneration of these trees was also greatly reduced (Trapnell 1959). Frequent late season fires eventually convert woodland into open, tall grass savanna with patches of isolated, fire tolerant canopy trees such as the *P. angolensis*, combined with scattered understory trees and shrubs (Frost 1996).

2.3 MAPPING BURNED AREAS

2.3.1 Overview of developments in burned area mapping

Burned area mapping is one of the tools that are used in assessing the impact of fire on vegetation. It facilitates the reconstruction of a fire history within a given area, by giving the time, location and extent of fire damage over an extended period of time. Information derived from burned area mapping allows one to establish fire frequency and FRI prevailing in the area under study (see Heinl 2005, Archibald et al. 2010a). Mapping of burned areas is commonly carried out through the use of field mapping, especially in protected areas, e.g. national parks and reserves (Trollope 1993, Gandiwa & Kativu 2009). However, field mapping of burned areas (also known as fire affected areas) is generally costly and time consuming. Furthermore, field rangers tend to have difficulties in manually measuring the full extent of large fire scars (de Klerk 2010). On the other hand, a wide range of remote sensing applications for measuring burned areas have been developed over the last forty years (Giglio et al. 2009), these techniques will be investigated in the discussion below.

Growing concern over greenhouse gas emissions and climate change has prompted an increasing demand for long term fire data at global, regional and local scales. In response, using various coarse and medium resolution sensors, a number of satellite-based fire observations have been developed over the years (Giglio et al. 2009, Archibald et al. 2010a). This has resulted in the global production of both active fire data and burned area data. The development of active fire data characterized the initial stages of fire mapping via remote sensing. The first satellite based fire data was derived from the Advanced Very High Resolution Radiometer (AVHRR) (Dozier 1981, Matson & Dozier 1981). Ensuing satellite-based active fire observations, included the Geostationary Operational Environmental Satellite (GOES) Imager (Prins & Menzel 1992), the Defense meteorological Satellite Program (DMSP) Operational Linescan System (OLS) (Elvidge et al. 1996), the Along-Track Scanning Radiometer (ATSR) (Arino & Rosaz 1999), the Visible and Infrared Scanner (VIRS) (Giglio et al. 2000), the Moderate Resolution Imaging Spectroradiometer (MODIS) (Justice et al. 2002), and the Spinning Enhanced Visible and Infrared Imager (SEVIRI) (Roberts et al. 2005).

Active fire data is derived from hotspot detection algorithms. These algorithms record the time and location of actively burning fires at the moment of satellite overpass (Roy et al. 2002). Therefore, seasonal and daily patterns of fires can be determined. Fire active data spanning over the past decade are available from sensors such as MODIS and SEVIRI. The MODIS sensor on the Terra and Aqua platforms provides eight day summaries of active fire data at a 1 km resolution. The SEVIRI sensor, on the other hand, records active fires every 15 minutes at a 5 km resolution (Archibald et al. 2010a). Limitations in fire active data do exist however. For instance, active fire algorithms only detect actively burning fires and do not measure the extent of the fire affected area. Furthermore, fires burning before or after the satellites` overpass go undetected. The presence of clouds or thick smoke may also obscure the detection of an actively burning fire (Roy et al. 2005a).

Due to the temporal bias and spatial inadequacy in active fire data, various remote sensing techniques have been developed to map the extent of burned areas. The application of land cover analysis on a range of satellite imagery has been widely used to map burned areas. For instance, Mbow et al. (2000) used NOAA AVHRR satellite imagery to map the spatial and temporal distribution of fires over a period of three years in Senegal. Hudak & Brockett (2004) mapped fire scars from 1972 to 2002, using historical Landsat imagery. While Bucini & Lambin (2001), used the short-wave infrared (SWIR) and thermal bands of ATSR images to map areas affected by burning. However, these remote sensing techniques require a significant amount of processing and expertise. They may also prove to be costly in terms of acquiring satellite imagery and the necessary software for analysis (de Klerk 2008).

Nevertheless, ready-to-use burned area products may be used as a surrogate for manual generation of burned area data. Some of the current burned area products are available from GLOBSCAR, GBA-2000 and MODIS (Giglio et al. 2009). The GLOBSCAR burned area product, initiated in 2001, was designed to produce global monthly maps of burnt areas for the year 2000, using daytime data from the ATSR-2 instrument onboard the ESA ERS-2 satellite (Simon et al. 2004). The GBA-2000 burned area product, designed by Tansey et al. (2004), is derived from SPOT VGT 1 km imagery and consists of global monthly burned area data for 2000. The MODIS burned area product (MCD45AI) was developed from an algorithm designed to map burned areas by Roy et al. (2002). It is based on the application of a bidirectional reflectance model-based algorithm on the MODIS near infrared (NIR) and SWIR bands. The MCD45A1 provides global daily 500 m records of burned areas from 2000 onwards (Roy 2005). Other burned area products include the GlobCarbon (Plummer et al. 2006) and the 1km L3JRC modified from the GBA-2000 algorithm (Tansey et al. 2008). All the burned area products mentioned above are freely available to the public online.

2.3.2 Detectability of burned areas in African savannas

The detection of burned areas is based on the condition of the post-fire landscape. Spectral contrast between unburned areas and burned areas is determined by three aspects: the presence of charcoal and ash resulting from burning, the reduction in photosynthetic vegetation (Robinson 1991) and an increase in land surface heat (Eva & Lambin 1998b). However, the spatial and temporal variability in the spectral response of vegetation in burned areas, affects the accuracy of burned area mapping products (Laris 2005).

The accuracy of burned area products depends on the spatial pattern of burned scars (Silva et al. 2003). A study carried out by Silva et al. (2005) assessed the accuracy of 13 Landsat scenes versus SPOT VGT data to estimate burned area in the African savannas. The authors concluded that small or complex shaped burn scars, were less accurately estimated at the SPOT VGT 1 km resolution. Eva & Lambin`s (1998a) comparison between burned area estimates derived from 30 m resolution Landsat TM data and 1 km resolution ATSR-based maps gave corresponding results. The study showed that the coarser resolution data underestimated burned areas in landscapes dominated by fragmented burn scars. On the contrary, the burned area in landscapes with larger and more condensed patches of burn scars was overestimated. These errors are a function of low resolution bias, which results when burn scars occur at a finer spatial pattern than the satellite image resolution (Boschetti et al. 2004). Therefore, when using coarse resolution data for estimating burned area in savanna woodland, errors are expected to occur in either an early fire season which results in patchy, fragmented burn scars, or in a late fire season where the burn scars are larger and more dense (Laris 2005).

Improving burned area mapping accuracies in the savanna woodland requires higher resolution data, such as the MODIS 500 m burned area product or fine resolution data such as Landsat TM (Roy & Boschetti 2009). However, the detectability of burned areas also depends on the degree of spectral contrast with the background. Burn scars surrounded by dense woody vegetation or shadows are more difficult to detect accurately than those surrounded by dry grasses (Eva & Lambin 1998b). The time since fire also has an impact on burned area mapping. As burn scars fade with time the spectral contrast between burned and unburned vegetation is reduced. This could be a function of prevailing weather conditions and the vegetation recovery rate. It should also be noted that vegetation in the savannah starts to dry with the onset of the fire season, therefore it becomes harder to distinguish burned vegetation from senesced vegetation as the dry season progresses (Laris 2005).

A general overview of burned area mapping shows that limitations exist in both the active fire product and the burned area product. The active fire product has a non-optimal overpass time and a low revisit frequency of the satellite, while the burned area product is significantly affected by low bias resolution, and the variability of the vegetation’s spectral response to fire (Laris 2005). The type of product used is subject to the users needs. For example, active fire products have been used for mapping fire frequency at the regional scale and for real-time fire detection (Justice et al. 2002), while burned area products have been used to produce maps of burned scars (Eva & Lambin 1998a). Both products have also been combined to increase the accuracy of burned area estimates (e.g. Roy et al. 2005a).

CHAPTER 3: RESEARCH METHODS

3.1 STUDY AREA

3.1.1 Location

Charara Safari area (CSA) is located in Hurungwe District in the Mashonaland west province of Zimbabwe. The CSA is surrounded by communal lands, namely, Kanyati, Nyaodza and Vuti, on its eastern boundaries, while its south west boundary lies on the shores of Lake Kariba (Figure 3.1). The CSA is a state owned hunting area safari (Vorlaufer 2002), with several fishing and safari camps along its shores, that covers an area of approximately 166 354 ha (Surveyor General 1978).

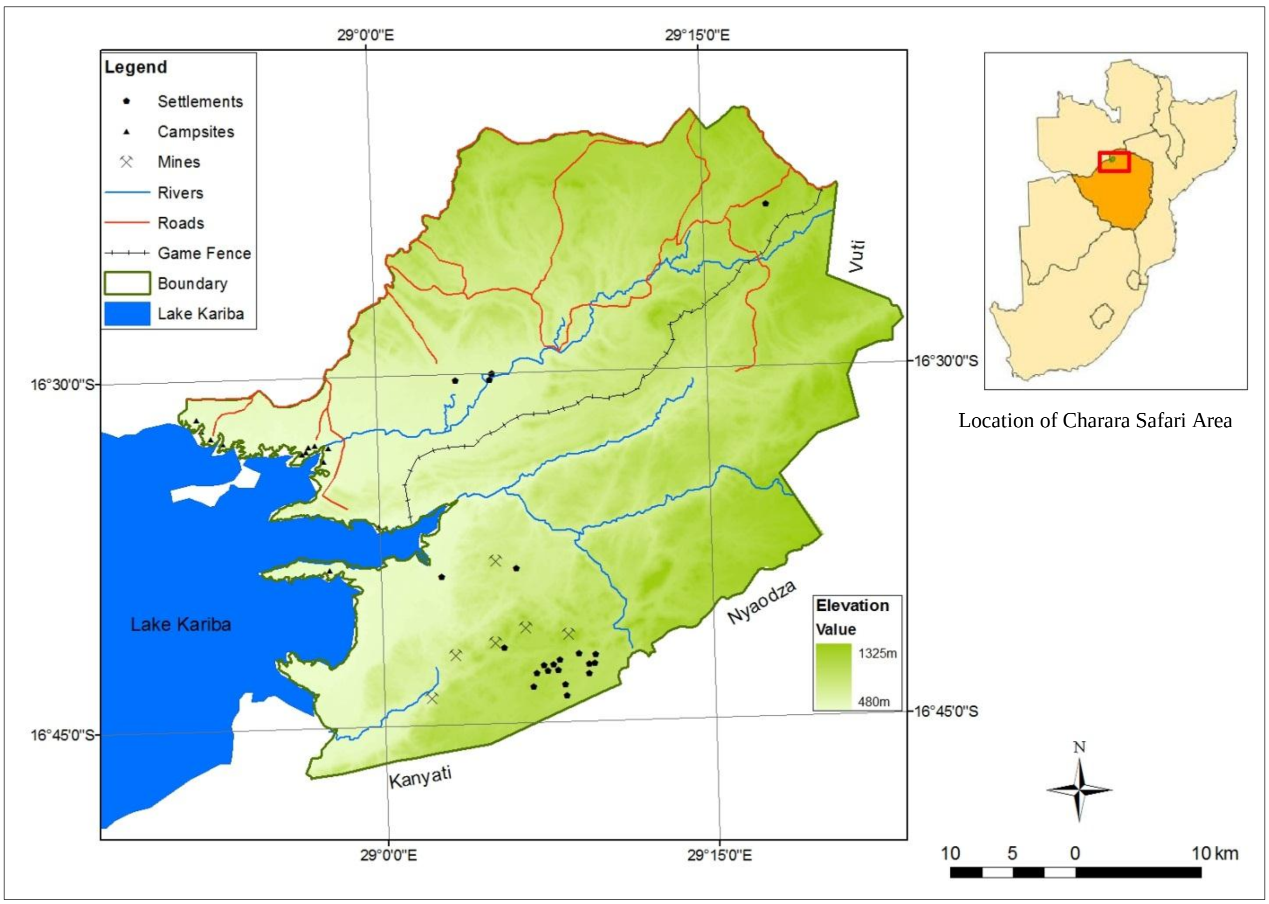

Figure 3.1 Map of study area (CSA) adopted from Surveyor General (1978)

3.1.2 Geology

According to Ashton et al. (2001), CSA is mainly characterised by underlying gneisses, and Karoo Supergroup rocks in the area adjacent to Lake Kariba. The soils are very shallow (<25 cm deep) and gravelly lithosols derived from phillites and quartzites (Department of Research and Specialist Services 1979). The area is generally defined by undulating plains with a series of inselbergs (round shaped hills). (Silber 2011).

3.1.3 Climate

The climate over CSA is generally hot and dry, with a rainy season spanning from November to April. The CSA cuts across the of Zimbabwe`s natural regions: III, IV and V (Figure 3.2). Therefore, the rainfall in CSA forms a pattern of three vertical bands of different ranges of rainfall. The area adjacent to the lake shore is in natural region V, and is characterised by rainfall of less than 650 mm. The section in the middle of CSA falls under natural region IV, and has an annual rainfall of around 450 to 650 mm. The last portion of CSA covering the area adjacent to the eastern border, covering natural region III, has infrequent heavy rainfall of about 650 to 800 mm per year (Surveyor General 1984). The average mean temperature in CSA is approximately 31^{0}C, with minimum temperatures of about 26^{0}C in July and maximum temperatures in October of about 36^{0}C (Silber 2011).

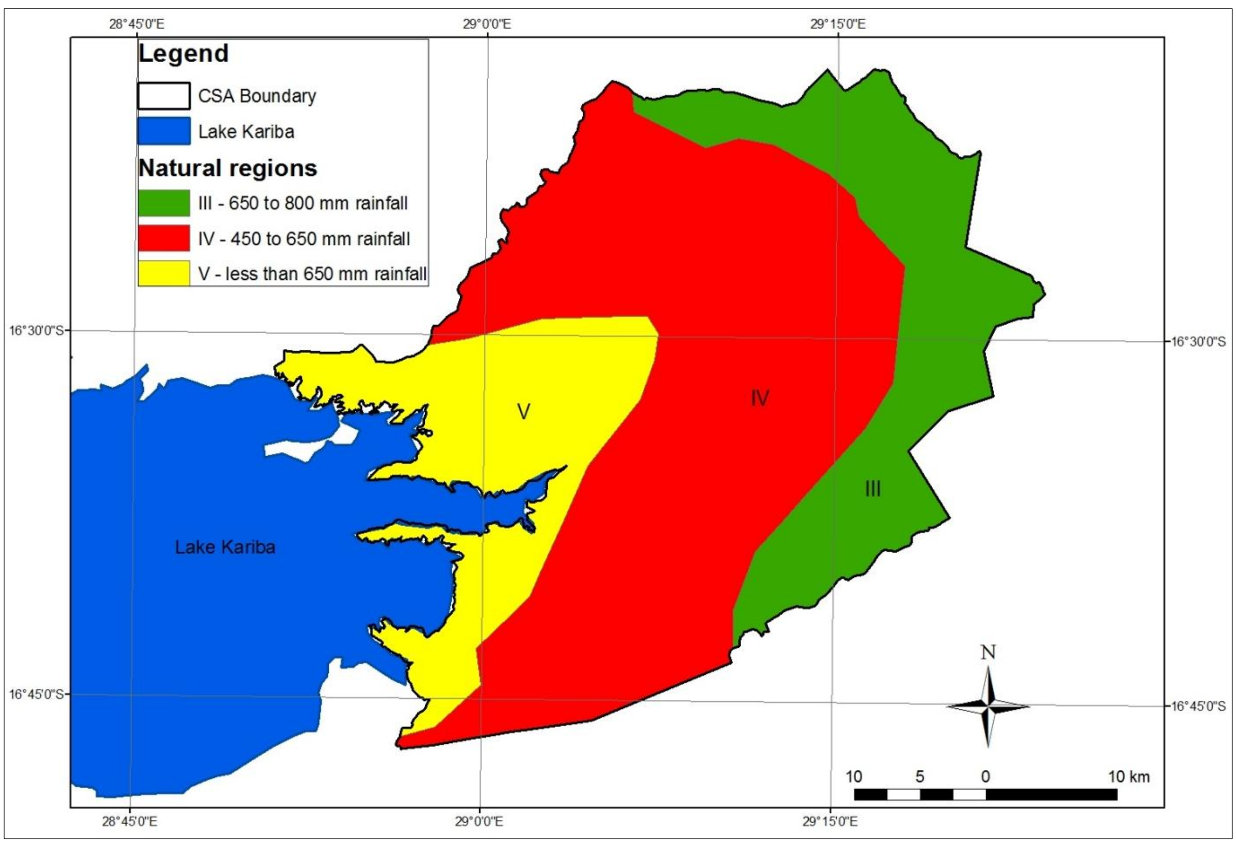

Figure 3.2 Natural regions covering CSA adopted from Surveyor General (1984)

3.1.4 Vegetation

According to (Olsen et al. 2001), CSA is mainly characterised by miombo woodland. However, *Colopohospermum mopane* (*C. mopane*) woodland dominates the area directly adjacent to lake shore. The mopane woodland is located on deep grey to brown sandy clay loam to clay soils formed from the underlying Karoo rocks. *C. mopane* can grow to a height of up to 12 m, while the shrub layer which is usually not dense, is 1 to 6 m high (Silber 2011). The grass layer is not well developed and generally consists of short, annual grasses and therefore provides little fuel for fires. Therefore, fires tend to be less abundant in mopane woodlands compared to miombo woodlands (Silber 2011). On the other hand, the miombo woodland covering CSA is defined as drier Zambezian miombo, dominated by *B. spiciformis* and *J. globiflora* (White 1983). The grass layer in the miombo woodland consists of perennial grasses, which are thicker than that of the mopane, and thus provide high fuel loads for the frequent occurrence of fires (Silber 2011).

3.1.5 Motivation for selecting site

The CSA was specifically selected for this study for a number of reasons. Firstly, it is located in one of the most fire affected provinces in Zimbabwe (Bara 2012) and was therefore deemed suitable as it would be an area guaranteed of fire occurrence. Secondly, due to its heterogeneity in vegetation cover, CSA was considered an ideal study area to map fires. As mentioned in the previous chapter, the aim of this research was to specifically assess the patterns of fire occurrence in the miombo woodland. Fire behaviour is greatly influenced by vegetation type, therefore, an area with a combination of several vegetation types would yield misleading results.

3.2 DATA COLLECTION

For the purposes of this study three sets of data were collected (Table 3.1): two MODIS Burned Area products (MCD45A1 and WAMIS) were used to extract various characteristics of the fire regime, while Landsat imagery was used to evaluate the mapping capabilities of the MODIS Burned Area products.

Table 3.1 Summary of data collected

Product	Source	File format
MCD45A1	NASA Reverb ECHO https://reverb.echo.nasa.gov	Hierarchical data format (*.hdf)
WAMIS	WAMIS, Meraka Institute www.wamis.co.za	Geotiff (*.tif)
Landsat	South African National Space Agency Earth Observation (SANSA EO) http://catalogue.sansa.org.za/.	FAST (*.fst)

3.2.1 MCD45A1 Burned Area Product

The MCD45AI is a MODIS Level 3 Monthly Tiled 500m Burned Area Product. This dataset has a temporal coverage from April 1ST 2000 to present day (Roy et al. 2008). As with all other MODIS products, the MCD45A1 is identified by its alphanumeric code of three letters and two numbers: MCD indicates that both Terra (morning) and Aqua (afternoon) satellites are used to collect the data and 45 is the identifier of the burned area product (Boschetti et al. 2009). MCD45A1 data are presented in the standard MODIS Land tile format in sinusoidal equal area projection (Hongxiao 2010). Each tile has a fixed area of approximately 1200 × 1200 km (10 degrees × 10 degrees in size). The horizontal (h) coordinates range from 0 – 35 and the vertical (v) coordinates range from 0 – 17, hence the tile in the upper left corner would be designated h0,v0 (Boschetti et al. 2009), see Figure 3.3.

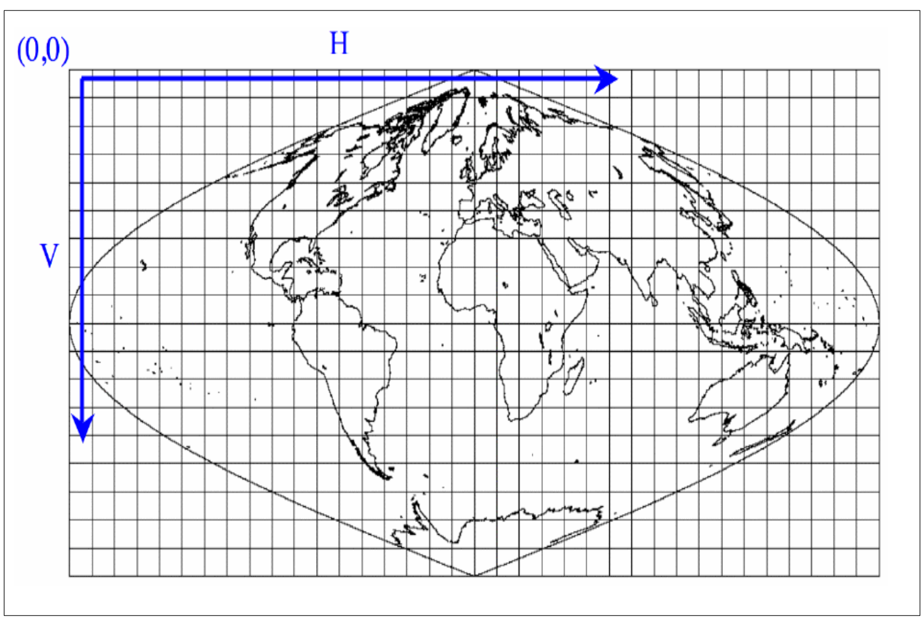

Figure 3.3 MODIS tiling scheme (Justice et al. 2006)

MCD45A1 data are produced at a 500m resolution using a bi-directional reflectance (BRDF) model-based change detection algorithm (Justice et al. 2006). It is based on the principle that burned areas are characterized by charcoal and ash deposits, removal of vegetation, and alteration of the vegetation structure (Roy et al. 1999). The algorithm is applied to each individual geolocated 500m pixel, to map the spatial extent of the burned pixels and the approximate day of burning, fires that occurred in previous seasons or years are not mapped (Roy et al. 2008). The algorithm identifies the approximate day of burning by locating rapid changes in the daily surface reflectance trend (Roy et al. 2008). Reflectances sensed within a window of a fixed number of days are used to predict the reflectance for the subsequent day. A statistical measure is then applied to quantify the difference between the predicted BRDF and observed BRDF. Large discrepancies between the predicted and measured BRDF values indicate a significant change; this change is further examined by using a temporal consistency constraint to differentiate between temporary changes considered as noise and persistent changes of interest indicating burned areas (Roy 2005, Justice et al. 2006).

The MCD45A1 data were downloaded from the National Aeronautics and Space Administration (NASA) website http://wist.echo.nasa.gov. However, as of 2012 the website for download has been changed to https://reverb.echo.nasa.gov. The MC45A1 data are distributed as one file per tile per month in hierarchical data format (*.hdf), and data downloaded for the 10 year period 2001 to 2010 totaled 120 *.hdf files occupying 450 Mb storage space. The name of each file downloaded provides information about the spatial and temporal coverage of the product.

For example, **MCD45A1.A2000306.h16v07.005.2006360205845.hdf** can be decoded as:

- **MCD45A**1 = MODIS level 3 Monthly tiled burned area product in HDF.
- **A2000306** = year and Julian date of the starting day of the month covered by the product: 306 is the Julian date of Nov 1, hence 2000306 means that the product covers November 2000.
- **h16v07** = spatial extent: the file covers tile h16v07
- **005** = version identifier. 005 indicates Collection5
- **2006360205845**= processing day and time (year 2006, Julian day 360, 20h58'45")

This naming convention guarantees a unique name for each file downloaded: if a tile is reprocessed the last number, specifying the day and time in which the file was processed, will be different, thus avoiding any confusion with obsolete data (Boschetti et al. 2009).

Each MCD45A1 file consists of the following eight Science Data Sets (SDS`s) i.e. layers: Burn date, Burned area (BA) pixel Quality assessment (QA), Number of Passes, Number Used, Direction, Surface Type, Gap Range 1 and Gap Range 2. This research makes use of the 2 byte integer burn date data set, which provides the approximate Julian day of burning from eight days before the beginning of the month to eight days after the end of the month. A code indicating unburned areas, snow, water, or lack of data is also integrated into the data set:

- 0 - unburned
- 1-366 - approximate Julian day of burning
- 900 – snow or high aerosol 9998 - water bodies (internal)
- 9999 - water bodies (seas and oceans)
- 10000 - not enough data to perform inversion throughout the period

(Boschetti et al. 2009)

3.2.2 WAMIS Burned Area Product

The Wide Area Monitoring Information System (WAMIS) Burned Area Product is MODIS-derived 500m burned area data produced by the Meraka Institute in South Africa (http://www.meraka.org.za/) in 2009 and is based on the Giglio et al. (2009) algorithm (de Klerk et al. 2011). The algorithm used to produce the WAMIS burned area product is based on the assessment of changes in vegetation index values and the integration of active fire data. The Giglio algorithm (Giglio et al. 2009) looks for a large, rapid decrease in the vegetation index (VI) that varies from the normal trend shown by a burn-sensitive VI compiled for the same time series of images. Persistent changes in the VI time series and measurements in temporal texture are then summarized to produce composite imagery. The composite imagery is further combined with 1km MODIS active-fire observations to identify burn-related and non-burn related changes within the scene. This information is used to derive probabilistic thresholds suitable for identifying burned and unburned pixels in the scene. The combination of active-fire and vegetation reflectance data enables the algorithm to adapt regionally over varying pre- and post-burn condition and across multiple biomes (Giglio et al. 2009).

The WAMIS burned area product is available for download from Meraka Institute’s Wide Area Monitoring Information System (WAMIS) website (www.wamis.co.za). It is distributed in either Geotiff (*.tif) or Shape (*.shp) format in Albers Conical Equal Area projection, World Geodetic System (WGS) 1984 Datum, with one file per month covering the whole of Southern Africa. For each month two files were available, one for each SDS: Burn date (2 byte) and BA pixel QA (1 byte). The BA pixel QA SDS expresses the level of confidence in the detection of pixels using values 1 to 4, from the most confident to the least confident respectively. However, burned area data for 2008 only covered the months January to February, therefore data for the rest of the year was supplemented from the WAMIS site. The remaining burned area data for the period 2008 to 2010 was downloaded from www.wamis.co.za in *.tif format, only the burn date SDS was available for this data.

3.2.3 Landsat images

Central to any accuracy assessment is the need for reference data which is essentially more accurate than the map under evaluation (Boschetti et al. 2006). Hence, in evaluating the mapping accuracy of the two MODIS products used in this study, high resolution Landsat imagery was utilised. The use of Landsat imagery as reference data in validating global burned area products is widely acknowledged in various scientific literature (Giglio et al. 2009, Roy & Boschetti 2009, Siljander 2009, Tsela et al. 2010). Due to a higher 30m spatial resolution, Landsat has a greater ability than MODIS to detect smaller and spatially fragmented burned areas. Therefore, it can be assumed that burned area maps extracted from Landsat imagery are a close representation of actual burned areas on the ground and thus be used as reference data (Boschetti et al. 2006). Further advantages to using Landsat in this retrospective evaluation exercise include:

1. Landsat`s historical data record, which can accommodate the time period under study in this research.
2. Wide spatial coverage per scene (approximately 170 x 183 km) ensures a limited amount of images required to map burns within the study area
3. Acquisition of imagery at relatively low to no cost.

In evaluating the mapping capabilities of the algorithms used in the MCD45A1 and WAMIS burned area product, Landsat imagery were ordered from the South African National Space Agency Earth Observation (SANSA EO) catalogue http://catalogue.sansa.org.za/ SANSA EO gives an allowance of 10 free Landsat images to each research student. However, more than ten images, spanning over the ten year study period, were required because the study area did not fall into one scene. For every year, two scenes had to be obtained to cover the entire study area (Figure 3.4). Each scene was referenced by Landsat`s unique Worldwide Reference System (WRS-2) path and row coordinate. Path171-Row71 covered the northern part of the study area, while Path171-Row72 covered the southern art of the study area.

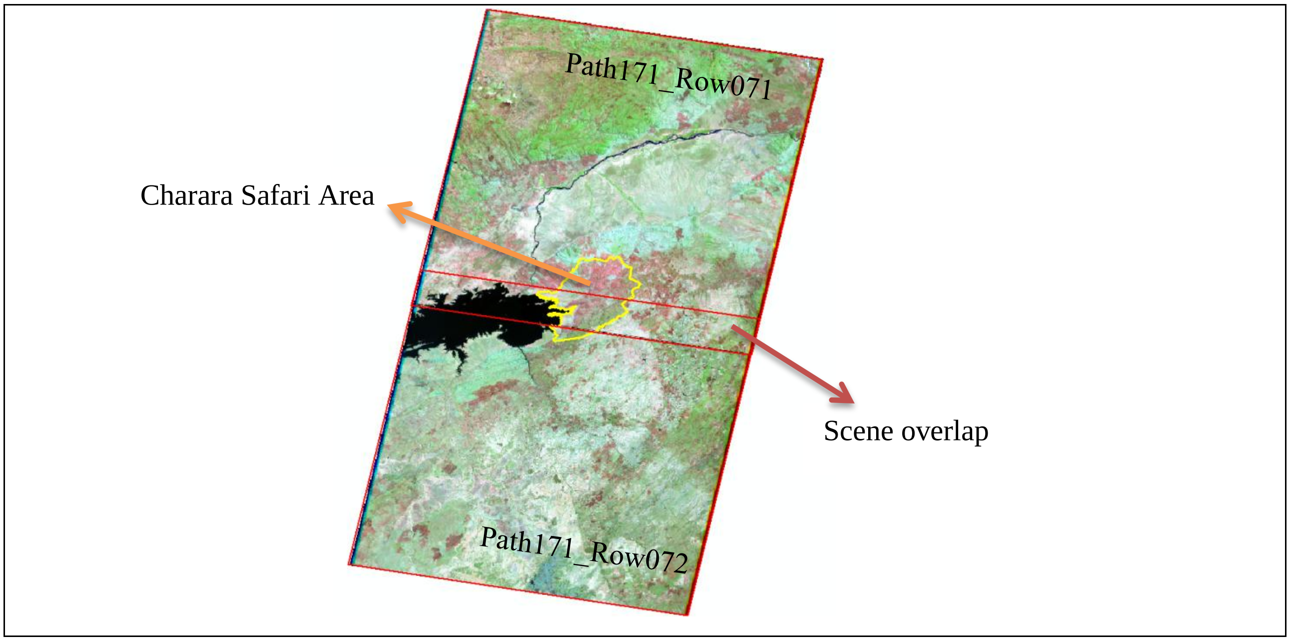

Figure 3.4 Two Landsat scenes (red outline) from the same day joined to cover the study area (yellow outline)

A total of twelve Landsat TM images and two Landsat ETM images were acquired for the period 2001-2010. However, cloud-free images for the years 2002, 2003 and 2010 failed to extract from the SANSA database, and therefore were not available. The fourteen scenes were received in FAST (*.fst) format and projected in the Universal Transverse Mercator (UTM) coordinate system. Images with acquisition dates at or closer to the end of the fire season (November) were selected where possible. As explained by Boschetti et al. (2006), depending on the development of the post burn surface in a given ecosystem, the spectral signatures of fire affected areas may remain detectable for weeks to years. In the savanna, which characterises the study area, burn scars may persist throughout the fire season, but will eventually disappear before the following fire season. Therefore, to ensure the detection of the bulk of burned area for the entire season, the end of the fire season was considered an ideal time period to select the Landsat images. It was not possible to acquire images near the end of the fire season for all the years however, namely 2007, 2008 and 2009 (Table 3.2).

Table 3.2 Landsat images acquired from SANSA EO with number of month covered in the entire fire season (May- November)

Path	Row	Date	No. of months covered	Sensor
171	071	24-Sep-2001	5	ETM
171	072	24-Sep-2001	5	ETM
171	071	26-Oct-2004	6	TM
171	072	26-Oct-2004	6	TM
171	071	29-Oct-2005	6	TM
171	072	29-Oct-2005	6	TM
171	071	14-Sep-2006	5	TM
171	072	14-Sep-2006	5	TM
171	071	16-Aug-2007	4	TM
171	072	16-Aug-2007	4	TM
171	071	02-Aug-2008	4	TM
171	072	02-Aug-2008	4	TM
171	071	17-May-2009	½	TM
171	072	17-May-2009	½	TM

3.3 PRE-PROCESSING OF DATA

3.3.1 MCD45A1 and WAMIS

The MCD45A1 data were converted into an ArcGIS compatible form from *.hdf format to *.tif using the MODIS Reprojection Tool (MRT). The MRT tool is available for download from the NASA Land Processes Distributed Active Archive Centre (LPDAAC) website: https://lpdaac.usgs.gov/tools/modis_reprojection_tool (Devineau et al. 2010). This tool enables the user to resample images, assign a projection and subset data. The MRT tool was used to reproject the MCD45A1 data from Sinusoidal projection to UTM zone 35S, and remove the seven SDS`s that would not be utilised in the study. A raster from the WAMIS data was selected as a reference layer (master grid) to facilitate accurate alignment between both the MCD45A1 and WAMIS data. Using the nearest neighbour technique, all MCD45A1 data were resampled to 424.388 m pixel size to match the master grid selected from the WAMIS data. WAMIS burned area data were also reprojected from Albers Conical Equal Area to UTM zone 35S and the datum set to WGS 84, resampled and re-aligned to the master grid using ArcGIS 9.3.

3.3.2 LANDSAT imagery

In preparing the Landsat images for the extraction of burned areas intended as reference data in the accuracy assessment of the two MODIS products, two pre-processing steps were taken. The initial step was to convert Landsat imagery format into a file format compatible with ArcGIS and ERDAS. For this purpose, ENVI was used to convert the images from *.fst format to *.tif. Before proceeding to manually digitising the fire scars, the images were also georefernced using ERDAS IMAGINE AutoSync.

3.3.2.1 Geometric correction

Landsat images collected from SANSA are L1G products, i.e. radiometrically and geometrically corrected (using systematic values) according to user specified parameters such as map projection, image orientation, and pixel cell size. As a result, the acquired Landsat images are free from both sensor and satellite related distortions (USGS http://landsat.usgs.gov/). Nonetheless, in order to ensure perfect alignment between the images, Band 1 from the 2001 (171-071) image was used as a reference to coregister the rest of the bands in the remaining images. An initial set of several tie-points were manually selected to run the automatic point matching function in Autosync, which generated more tie-points between the reference and input images. An RMS error of ±1 was achieved for all the images. Composites of bands 1 to 5 and 7 for each image were subsequently created in ArcGIS 9.3 for further processing.

3.3.2.2 Atmospheric and topographic correction.

No atmospheric and topographic correction was applied to the images. As explained by Mather (2004), atmospheric correction is particularly important in the case of comparing data derived from a number of images acquired at various times. However, the Landsat images in this study were not used for a multi-temporal analysis of burned areas, but rather for the sole purpose of manually digitizing burn areas, and therefore would not require atmospheric correction. In addition, due to the simple nature of manually delineating the burned scars from the images, it was deemed unnecessary to mosaic them, a process which would have otherwise required atmospheric correction. On the other hand, it is generally accepted that topographic correction does not yield any significant improvement in detecting burn scars from satellite imagery. For example, Mitri & Gitas (2004) found topographic correction to improve accuracy in mapping burns in the mountainous region of Thasos in Greece, by only 1.16%. Therefore, topographic correction is commonly left out when pre-processing of images for burned area mapping (Petropoulos et al. 2010, Bastarrika et al. 2011), and as such was not be employed in this study.

3.4 PROCESSING MODIS BURNED AREA DATA

3.4.1 Removal of non-relevant dates

Following the pre-processing of the MODIS burned area products, subsequent processing of the data was performed in ArcGIS 9.3. As described earlier, files from the MODIS burn data set, covering a period of a month, consist of a series of pixels described by five sets of information, defined by a specific set of values: Julian burn day, unburned, snow, water and invalid data (Boschetti et al. 2009). Values pertaining to burned area (1 to 366) were the point of interest for this study, therefore other values describing all other information (i.e. 0, 900, 9999 and 10000) were considered as non-relevant values. However, each file had an excess of 16 days, 8 days each from the preceding and consecutive month, therefore these excess dates were also classified as non-relevant values. All non-relevant values for both MODIS burned area products were consequently removed from each file in the 10 year data set using the raster calculator in ArcGIS 9.3.

3.4.2 Identifying individual fires

Having isolated the relevant burn values, all the files were converted from rasters to vectors (Appendix A), thus creating monthly vector files consisting of an amalgamation of 424.4× 424.4 m polygons, each with their respective burn dates. Each burned polygon was subsequently assigned of a numeric identifier in order to establish individual fires from the data. Following Archibald & Roy (2009), an 8 day/1 km algorithm was applied to classify each individual fire. This algorithm took account of the 8 day temporal accuracy of MODIS data and used a distance of 1 km to accommodate possible spotting of fires in the study area (Diaz-Delgado & Pons 2001). Therefore, all neighbouring polygons which burned within 8 days and 1 km of each other were assigned with the same fire ID. This was achieved by using a buffer to group burned polygons that occurred within a kilometre of each other. Polygons within each buffer were then manually assessed to ensure that all the polygons located in the buffer had burned within 8 days of the minimum burn date found in the buffer. Polygons that fell within this criterion were subsequently given a unique ID, and so the process was repeated until all polygons had been assigned with a new fire ID (Figure 3.5). Once the 8day/1 km algorithm had been applied to the entire 10 year data set, the number of fires and extent of area burned was established to form the basis for reconstructing and analysing the fire regime in CSA.

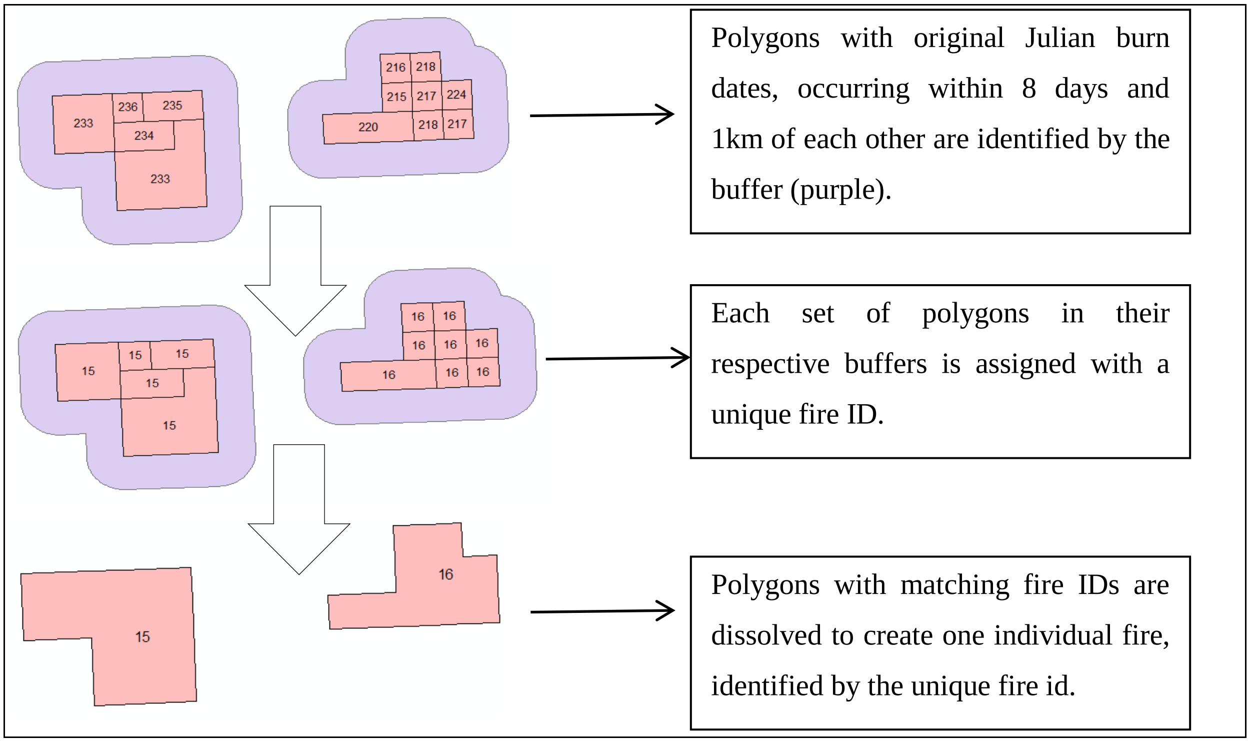

Figure 3.5 Process of identifying individual fires using the 8 day/1 km algorithm

3.5 PROCESSING LANDSAT IMAGERY

3.5.1 Digitising burn scars

Due to lack of ground truth data, it was decided that burn scars would be manually digitised from the Landsat imagery. Although the process is a tedious and time-consuming task, as observed by Peng and coworkers (2009), it is generally agreed that burn scars are relatively easy to delineate via visual interpretations (Roy et al. 2005b, Bastarrika et al. 2011). Also, it has been noted by Hudak & Brockett (2004) that manual digitisations of burn scar generally yield greater accuracies than burn scars derived through automatic techniques. Automated delineations of burn scars are largely limited by spectral confusion caused by the age of the burn scar (time elapsed since fire was extinguished), low combustion completeness, vegetation type and hindered detection of small fragmented fires. However, manual digitising can be particularly laborious when mapping small patchy fires across fragmented landscapes (Roy et al. 2005b, Bastarrika et al. 2011).

An extensive literature review was conducted to establish an appropriate band combination that would effectively discriminate between burned and unburned areas. Burn scars show high spectral contrast against unburned areas in the visible (0.4 – 0.7µm), near infrared (NIR) (0.7-1.5 µm) and middle infrared (MIR) (1.5 - 4.0 µm) regions of the electromagenetic spectrum (Fuller 2000). As explained by Eva &

Lambin (1998b), reflectance is greatly reduced in the visible channels when vegetation is burned due to the low albedo of the burn scars. The reduction of photosynthetic activity in green vegetation also induces a fall in the NIR. While, there is an apparent spectral distinction between burned areas and dry vegetation in the MIR region. Therefore, the 742 band combination was used to map the burn scars from Landsat in this study. According to Campbell & Wayne (2011), the 7-4-2 band combination utilises regions in the visible, NIR and MIR, as illustrated in Figure 3.6.

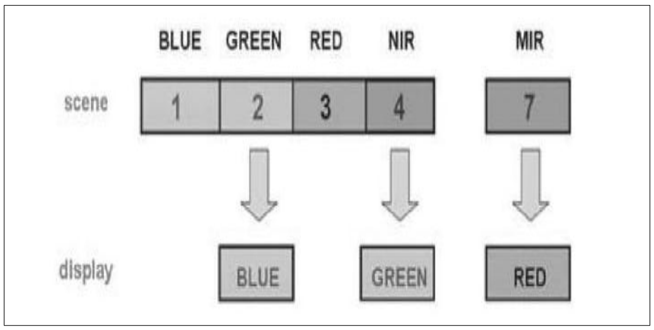

Figure 3.6 Band combination 7-4-2. Source: Campbell & Wayne (2011)

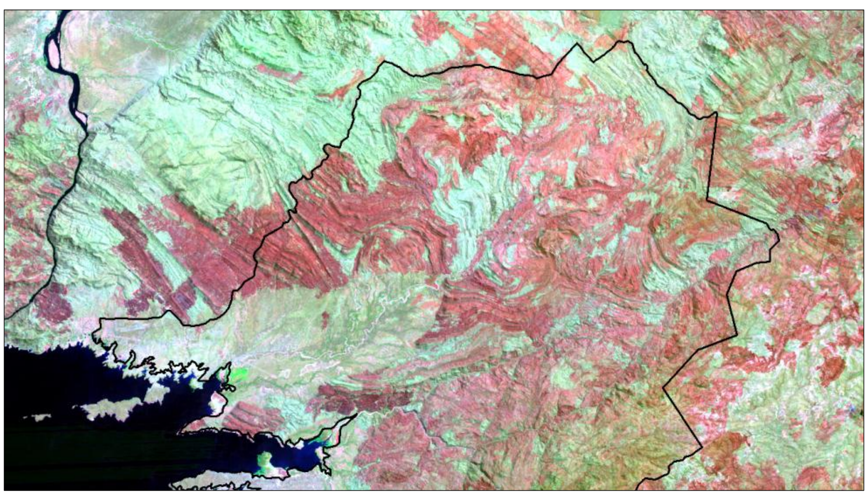

Figure 3.7 Band combination 7-4-2 as seen in the image over CSA, red portions signify burn scars

The 7-4-2 band combination is particularly useful because it gives a somewhat clear delineation of different features in the image (Figure 3.7). For example, healthy vegetation appears as bright green, while dry and sparse vegetation appears orange and brown, barren soils are generally pink, and open water is blue. Burn scars however, are identified by their deep reddish-brown colour (Peng et al. 2009), and were therefore easier to delineate during the digitising process (Appendix B). The burn scars derived from the

Landsat images were then used as the reference layer in the evaluation of the MCD45A1 and WAMIS (see Chapter 4).

3.6 DATA ANALYSIS

Following the processing and evaluation of the two MODIS burned area products, the burned area data set that proved most suitable for the purposes of this study (see Chapter 4) was selected for the reconstruction of CSA`s fire regime over the period 2001 to 2010. Reconstructing fire regimes is a method that has been used in a number of studies to establish past fire patterns for the purposes of determining the causes and ecological impacts of fires (Heinl 2005, Devineau et al. 2010). This method not only entails assessing the fire regime in CSA in terms of fire incidence over time as is common in fire assessments carried out in Zimbabwe (Section 1.3), but also facilitates the analysis of other aspects of the fire regime e.g. fire frequency, size and seasonality (Gill 1975, Bond & Keeley 2005). Statistics on the number of fire incidences and extent of area burned per month and annually in CSA between 2001 and 2010 were extracted from the MODIS burned area data set in ArcGIS 9.3 and imported to MS Excel for further analysis (Appendix C). Trends in fire incidence, extent, season, sizes, frequency and the fire return interval (FRI) were then established through a series of calculations in MS Excel and presented in the form of tables and graphs. Maps were also created in ArcGIS 9.3 to examine the spatial distribution of fire season, fire frequency and FRI in CSA over the 10 year study period (see Chapter 5).

CHAPTER 4: ACCURACY ASSESSMENT

In order to determine the reliability of the burned areas mapped by the two MODIS algorithms, and also select the product which will be used in reconstructing CSA fire history, an accuracy assessment in the form of a confusion matrix was employed (Appendix D). A confusion matrix, also known as an error matrix, is a geographic comparison of two maps: one map derived from remotely sensed data (map under evaluation) and the other from a different source of information (reference map). The confusion matrix measures the accuracy of the two maps on a point-by-point basis, and is also known as the standard method of reporting site-specific error (Campbell 2002). There are two types of errors that define site-specific errors: position error and thematic error. Position error occurs when the shape and size of a feature on a map is correct but its placement on the map is incorrect, while thematic errors are misclassifications of pixels on a map (Ned Horning http://www.biodiversityinformatics.cmnh.org), for example, the classification of actual burned pixels as unburned areas.

4.1 EVALUATION OF MCD4A1 AND WAMIS BURNED AREA PRODUCT

A confusion matrix was used to evaluate the accuracy of burned areas derived from the two MODIS burned area products: MCD45A1 (Roy et al. 2002) and WAMIS (Giglio et al. 2009), using fire scars digitized from Landsat as the reference data. Landsat images with acquisition dates closest to the end of the fire season (i.e. November) were selected for the assessment. As described previously in Section 3.2.3, some data was not available. As a result, burned area data for the following years were evaluated: 2001, 2004, 2006 and 2008. However, it is worthy to note that the Landsat images used in the evaluation do not cover the entire fire season i.e. May to November, and the months that are covered are detailed as shown in Table 4.1. According to Campbell (2002), when handling information that varies over seasons and time, it is important that information derived from the reference image matches that of the image to be evaluated, with regards to time. Otherwise, changes that would have occurred during the time that elapsed between the acquisitions of the two images may be misclassified as errors in the assessment. Therefore, to avoid the possibility of such errors, the burned area mapped by the MCD45A1 and WAMIS products in the months corresponding specifically to those covered by Landsat was also calculated, and used in the matrix. The construction and calculation of the confusion matrices was performed using ArcGIS, MS Access 2010 and MS Excel 2010.

Table 4.1 Landsat temporal coverage & burned area calculated for Landsat, WAMIS and MCD45A1

Landsat acquisition dates	**No of months covered (mths)**	**Burned Area Digitised from Landsat (ha)**	**Burned area mapped by WAMIS (ha)**	**Burned area mapped by MCD45A1 (ha)**
24-Sep-2001	5	112 852.75	78 430.38	77 337.50
26-Oct-2004	6	109 037.21	87 793.20	85 727.40
14-Sep-2006	5	104 932.49	89 142.95	88 868.95
2-Aug-2008	3	79 582.62	33 946.52	50 196.62

From the confusion matrices, five measurements of accuracy were derived for each of the four years: (i) Error of omission (EO), also known as false negatives, which indicates how much is missed by the classification; (ii) Error of commission (EC), or false positives, which indicates the level of over-estimation by the classification; (iii) Producers accuracy (PA), expressed as the proportion of reference area in a class correctly classified in the image being evaluated; (iv) Consumers accuracy (CA), defined as the proportion of the classification area in a class correctly classified in the reference layer; and (v) Kappa (k), which is a statistical measure of the degree of association between two data sets. PA and CA are specifically related to map production and in effect complement the EO and EC, respectively (Campbell 2002, Johnson & Ross 2008). A summary of the results are provided in Table 4.2 and explained in the following discussion. EO were higher than EC for both MODIS products, with levels of accuracy ranging between 33% and 73% for PA, and 78% to 89% for CA. Values for k also indicated a fair level of accuracy in both products, with the exception of calculations for WAMIS in 2008 (Table 4.2).

Table 4.2 Summary of evaluation: Landsat vs WAMIS & MCD45A1 (for Burned category)

	Error of Omission %		Error of Commission %		Producers Accuracy %		Consumers Accuracy %		Kappa (k)	
YEAR	WAMIS	MCD45A1	WAMIS	MCD45A1	WAMIS	MCD45A1	WAMIS	MCD45A1	WAMIS	MCD45A1
2001	37.68	38.80	10.33	10.17	62.32	61.20	89.67	89.83	0.40	0.39
2004	29.55	30.74	12.36	11.77	70.45	69.26	87.64	88.23	0.47	0.47
2006	30.71	31.21	18.55	18.87	69.29	68.79	81.45	81.13	0.40	0.39
2008	66.33	46.01	21.29	14.68	33.67	53.99	78.71	85.32	0.26	0.46

4.2 ERROR OF OMISSION

EO specifies the level at which both the MCD45A1 and WAMIS product missed or under-mapped fires which were identified in the Landsat image (Campbell 2002). Excluding 2008, the two MODIS products missed 29.6 to 38.8% (Table 4.2) of the fires mapped from Landsat. There is a difference of less than 1% between the EO calculated for both MCD45A1 and WAMIS for the years 2001, 2004 and 2006. However, in 2008, the omission error was much higher than that measured in the other three years. WAMIS missed 34% more fires than those missed in the last three years of evaluation, while the MCD54A1 fared better and missed approximately 12% more fires than had been missed in the previous years. Lastly, contrary to the other three years evaluated, WAMIS performed poorly versus MCD45A1 in 2008, under-mapping 20.3% more fires than the MCD45A1 product (Table 4.2).

Roy & Boschetti (2009) suggest that the MCD45A1 detects approximately 75% of burned areas mapped by high resolution imagery in Southern Africa, and therefore potentially misses about 25% of burned areas derived from Landsat imagery (Figure 4.1). The EO calculated for this accuracy assessment corresponds with this assertion, as both MODIS products missed around 33% of burned areas mapped from Landsat imagery in the years 2001, 2004 and 2006 (Table 4.2). MODIS burned products under-map fires for a number of reasons. Firstly, at 500 m resolution, burned areas smaller than a pixel (< 25 ha) have a lower probability of being detected (Staver et al. 2011). Also, low combustion completeness (i.e. the level at which the vegetation has been burned) may result in an insufficient change in surface reflectance to be classified as a burn, hence some fires remain unidentified (Roy et al. 2005a). Lastly, both persistent cloud cover and high tree canopy cover may also obstruct the satellite`s detection of fires burning on the surface, thus resulting in an omission error (Giglio et al. 2009).

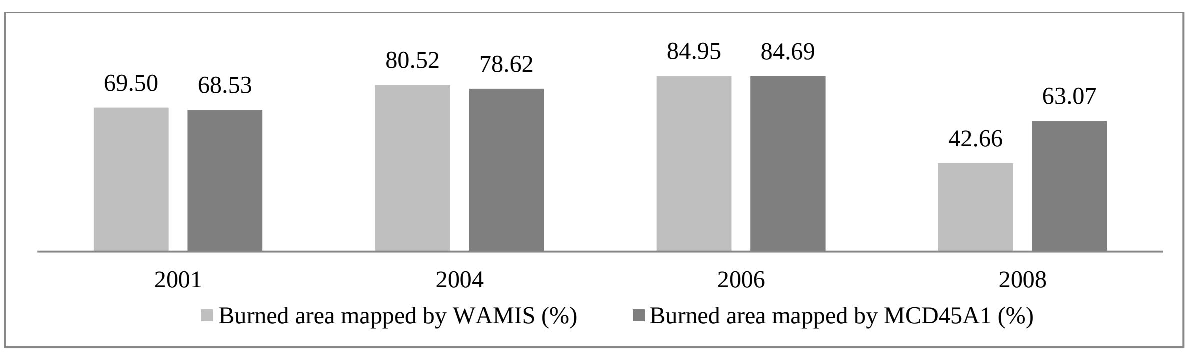

Figure 4.1 Proportion of burned areas derived from Landsat detected by WAMIS & MCD45A1

With reference to the anomaly in the level of omission revealed in both MODIS products in 2008, given that fires mapped for this year covered the early season i.e. May to July, fires burning over this period are generally small and fragmented due to the partially moist fuel load and mild weather conditions (Eriksen 2007). As mentioned by Giglio et al. (2009) and Staver et al. (2011), the MODIS burned area products miss the majority of small fires because of the coarse resolution, thus leading to high errors of omission. There are two other scenarios that may be considered, both related to a possible increase in rainfall. A time series of precipitation data was therefore downloaded from The Tropical Rainfall Measurement Mission (TRMM) Online Visualization and Analysis System (TOVAS) (http://gdata1.sci.gsfc.nasa.gov), a member of the Giovanni (Goddard Earth Sciences Data and Information Services Center (GES-DISC)). Using TOVAS, precipitation data for Charara Safari Area (CSA) was plotted from the Global Precipitation Climatology Project (GPCP) data set, Version 2.2 as shown in Figure 4.2.

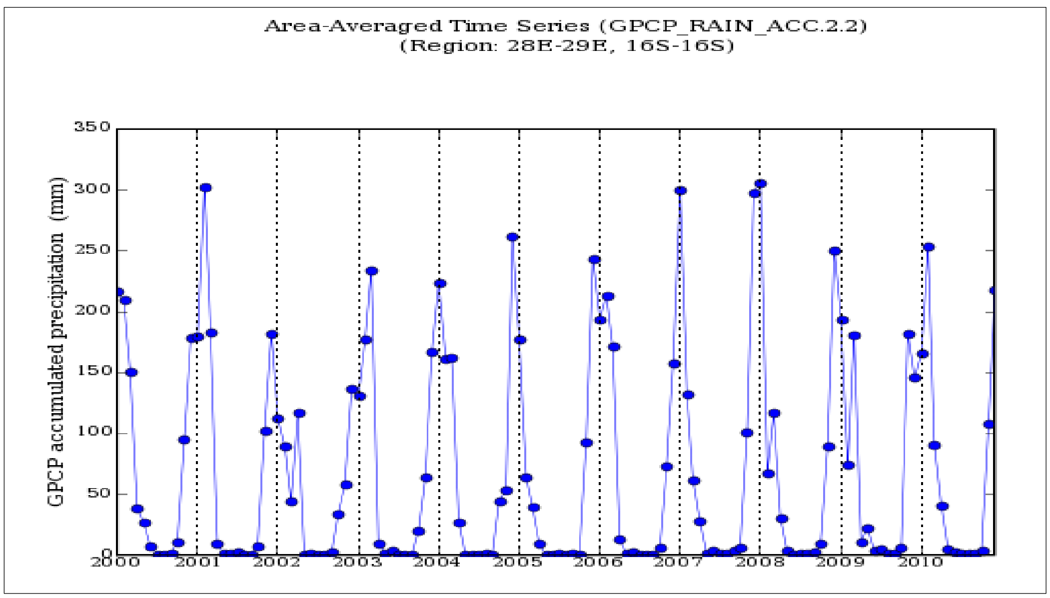

Figure 4.2 GPCP accumulated precipitation for Charara Safari Area for the period 2001 – 2010

Figure 4.2 portrays a peak in precipitation in the years of 2007 and 2008 (50 mm higher than peaks in preceding five years), which could possibly have led to a substantial increase in the amount of moisture content in the fuel load in 2008. It should also be noted that the data evaluated for 2008 consists of early season fires occurring between May to July, when temperatures are relatively low. As a result, due to partially wet fuel load and calm weather conditions, the fires would then be expected to be less intense (cooler) and slow burning (WWF 2001). Therefore, it can be assumed that the fires mapped in 2008 were predominantly low combustion fires. And as stated earlier (Roy et al. 2005a), low combustion fires may not present a noticeable change in surface reflectance to warrant detection by the MCD45A1 BRDF-based algorithm. Giglio and coworkers (2009) also allude to the under-estimation of large portions of areas burned by low combustion fires by the WAMIS algorithm. This is because the partial combustion of woody biomass tends to produce a negligible change in the VI used for the initial detection of possible burned areas by WAMIS. Hence, low combustion fires may be considered as part explanation for the large level of omission shown in 2008 by the two MODIS products.

Increased tree canopy cover caused by the surge in rainfall in 2007 and 2008 may also be considered as an explanation for the irregular omission errors recorded in 2008. Kanschik & Becker (2001) state that canopy cover in miombo varies between 20 and 100% depending on various ecological factors, rainfall being one of them. Findings from Giglio and coworkers (2009) in which the active fire based algorithm for mapping burned areas was assessed, reinforce the previous assumption. In their study, burned areas mapped by the active fire algorithm were evaluated against burned areas derived from Landsat imagery in Southern Africa under two vegetation covers: high tree cover (≥ 21%) and low tree cover (< 21%). Regions of high tree cover recorded an error of omission of 41%, while burned areas were only under mapped by 14% in low tree cover. The error of omission in this evaluation was 41% for WAMIS and 37% for MCD45A1 (Table 4.2), therefore both MODIS products correspond to the results calculated by Giglio et al. (2009) in the high tree cover region. Having established that the study area is quite probably influenced by high tree cover, it may be argued that the spike in omission errors in 2008 is possibly a reflection of increased canopy cover, resulting from the two previous years of high rainfall in CSA. Since fires in the Southern Africa are predominantly surface fires fueled by understory biomass, the assumed increase in canopy cover might have considerably reduced the detectability of the burn scars by both MODIS products (Giglio et al. 2009). Furthermore, Archibald et al. (2010a), who found evidence of negative feedback between tree cover and fire, states that at 40% tree cover in Southern Africa, fire incidences drastically decrease as a result of reduced fuel load in the understory. However, as will be illustrated in the following chapter, despite the assumed increase in tree cover in 2008, fires during this year recorded the largest amount of burned area in the entire 10 year study period. This then suggests that tree cover in 2008 ranged between 21 - 40%, and within this range the detectability of burned areas by both MODIS algorithms was severely compromised. WAMIS`s under-estimation of 20% more fires than MCD45A1 in 2008 (Table 4.2), suggests MCD45AI performs better than WAMIS under high tree cover conditions. A vegetation cover analysis spanning over the 10 years under study would be useful in validating this assumption, but will not be carried out in the course of this study due to time constraints.

4.3 ERRORS OF COMMISSION

EC reflect the level at which both MODIS products over-estimated the actual burned area identified in the Landsat image. The commission errors, ranging between 10 and 21%, are substantially lower than the omission errors (Table 4.2). This has been a typical conclusion in various validations of MODIS burned area data sets (Giglio et al. 2009, de Klerk et al. 2011). As explained by Laris (2005), coarse resolution data, such as MODIS, require thresholds to determine whether a pixel is burned or unburned. The threshold chosen results from a compromise between the number of commission and omission errors, while preserving a high level of valid pixels. Low EC and high EO shown in this accuracy assessment are indicative of the integration of high threshold values into the two algorithms under study. High threshold values are used to ensure the detection of large fires covering a greater proportion of a pixel, but this high threshold leads to smaller fires being missed, thus reducing EC. However, this does result in smaller fires below the given threshold going undetected leading to higher EO. In addition, EC are more readily identified and rectified in quality assessments and algorithm tuning procedures, and therefore tend to be lower than EO (Roy & Boschetti 2009). Nevertheless, spectral confusion is a prime factor in the over-mapping of burns by MODIS. WAMIS data particularly confuses recently cleared, unburned forest patches with burned areas because of the similarities in spectral signatures and temporal texture (Giglio et al. 2009). MCD45A1 data has been noted to show confusion under reduced infrared reflectance, as such a change in reflectance does not only signify burning, but is also indicative of a removal of vegetation, exposure of less reflective soils, flooding and vegetation senescence. Shadows cast by clouds and surface relief may also exhibit spectral changes similar to those caused by fires, leading to commission errors (Roy et al. 2005a). Both MODIS products measured very similar levels of commission, with a maximum difference of 0.6% in the years 2001, 2004 and 2006. In 2008 however, WAMIS over-estimated roughly 6% more fires than MCD45A1 (Table 4.2).

4.4 PRODUCERS ACCURACY

PA is defined by Campbell (2002) as a measurement of accuracy based on the analysts point of view, it informs the product producer (MCD45A1 or WAMIS) of the proportion of the reference image (e.g. Landsat) which was correctly classified in the image under evaluation. For example, in 2001 61.2% of the burned area mapped by Landsat was correctly classified as burned by MCD45A1 (Table 4.2). From the PA value, one can also then calculate how much of the fires mapped by Landsat were missed by the MODIS products. Therefore, it is evident there is a relationship between the PA and EO: the PA shows how much of the fires mapped in Landsat were correctly classified as burns by MODIS, and the omission error measures the level at which burned areas mapped by Landsat were missed by MODIS. Hence, the summation of PA and EO is always 100% (Campbell 2002). PA calculated for both MODIS products ranged between 61.2% and 70.5%, with a maximum difference of about 1% between the two MODIS products. Except for 2008 however, where a difference of approximately 20% in the PA was recorded between WAMIS and MCD45A1, WAMIS showing lesser accuracy (Table 4.2). Having demonstrated the relationship between PA and EO, it stands to reason that PA is also an indicator of the level of omission in both MODIS products. Therefore, factors that could have contributed to the level of PA calculated are the same as those that affect EO, and thus include: the effects of high tree cover in the detection of understory burns, limited identification of fires small fires (i.e.13-25 ha) due to coarse spatial resolution, persistent cloud cover, and low combustion completeness.

4.5 CONSUMER`S ACCURACY

CA is of significant importance in evaluating the mapping capabilities of both MCD45A1 and WAMIS algorithms. This parameter measures the reliability of the MCD45A1 and WAMIS algorithms as predictive devices (Campbell 2002). CA informs the user of how much of the portion identified as burned area by the two algorithms corresponds to the actual burned area on the ground i.e. Landsat. The CA calculated for both MODIS products showed good accuracy, with values ranging between 78 and 90% (Table 4.2). These results display close agreement with Roy et al. (2005b), where MCD45A1 detected 85% of the true burned area over the Southern African region. CA values further indicate that the chances of the MODIS algorithms mapping burned areas that are not actual burns on the ground are fairly low. The CA for both MCD45A1 and WAMIS measured quite similarly in all years except 2008, where WAMIS was shown to be less accurate by almost 6% (Table 4.2). As is the case with PA and EO, the summation of CA and EC is also equal to 100%. In some studies such as Johnson & Ross (2008), EC is actually calculated by subtracting the CA from the value 1 and expressed as a percentage. Hence, CA does not only illustrate the capabilities of the MODIS products in detecting true burned areas, but also the level at which these products can over-estimate burned areas. Based on the statistical relationship between CA and EC as well as the high CA calculated in this assessment, it may be concluded that spectral confusion in the detection of burned areas by the MODIS algorithms negatively influences the accuracy of the burned area mapped, but only to a small extent.

4.6 KAPPA COEFFICIENT

The Kappa coefficient is a measure of the difference between the observed agreement between the two maps (i.e. Landsat vs. MCD45A1/WAMIS) and the agreement that might be achieved solely by chance matching of the two maps. For instance, k = 0.40 can be interpreted to mean that the MODIS algorithm achieved an accuracy of 40% better than would be attained by a chance assignment of random burned pixels in MODIS to randomly selected burned pixels in Landsat. Because not all agreement can be credited to the success of the two algorithms in detecting burned areas, k is therefore calculated to provide an adjusted measure of agreement by removing the estimated chance of agreement. As the k moves towards +1, the effectiveness of the algorithm increases. Kappa values near zero suggest that the mapping capabilities of the algorithm yield no better results than would a chance assignment of pixels (Campbell 2002).The following equation was used to calculate k:

$$\mathrm{k} = \frac{observed - expected}{1 - expected} \quad \text{(Campbell 2002)}$$

where *observed* (equivalent to overall accuracy) is calculated by dividing the total number of correctly assigned pixels by the total number of pixels, and *expected* is based on the chance agreement between the map under evaluation and reference map, and is calculated as products of row and column values of a class in the confusion matrix.

A summary of the k values calculated for both MCD45A1 and WAMIS in 2001, 2004, 2006 and 2008 is shown in Table 4.3. Excluding 2008, both WAMIS and MCD45A1 show close agreement. In the years 2001, 2004 and 2006, k ranged between 0.39 and 0.47, suggesting 'fair to good agreement' as defined by Landis & Koch (1977), illustrated in Table 4.4. Therefore, with the exception of WAMIS in 2008, where k is 0.46 for MCD45A1 and 0.26, the Kappa values calculated in this assessment represent a reasonable level of accuracy in the burned areas mapped by both MODIS algorithms.

Table 4.3 Summary of Kappa statistics calculated for WAMIS and MCD45A1

Year	**Observed accuracy (%)**		**Expected accuracy due to chance (%)**		**k**	
	WAMIS	MCD45A1	WAMIS	MCD45A1	WAMIS	MCD45A1
2001	70	69	49	49	0.40	0.39
2004	74	74	51	50	0.47	0.47
2006	71	70	51	51	0.40	0.39
2008	64	73	51	51	0.26	0.46

Table 4.4 Summary of Kappa values as described by Landis & Koch (1977)

K	Level of agreement
< 0.4	Poor agreement
0.40 – 0.75	Fair to good agreement
> 0.75	Very good to excellent agreement

Excluding WAMIS in 2008, the Kappa values measured for both MODIS products correspond to findings by Giglio and coworkers (2009), where k was 0.49 under high tree cover (i.e. >21%) in Southern Africa. These results further highlight the assumption that tree cover significantly affects the mapping accuracy of MODIS burned area products in Southern Africa. More so, the low k value calculated for WAMIS in 2008 reinforces the suggestion that increased tree cover and low combustion fires resulting from two consecutive years of high rainfall might have heavily affected the detection ability of WAMIS.

4.7 RELIABILITY OF WAMIS AND MCD45A1

The purpose of this accuracy assessment was to determine the reliability of the MCD45A1 and WAMIS burned products in mapping burned areas in CSA, an area predominantly consisting of miombo woodland. With the exclusion of 2008, the results for both MODIS products, in the five parameters of accuracy calculated, differed by a very small margin of ±1% (Table 4.2). Therefore, although the two MODIS products use different algorithms to map burned areas, they nonetheless yield similar results when mapping fires over CSA. The anomalies in the calculations for 2008 do, however, suggest that the MODIS products may map burned areas differently at certain environmental thresholds. For instance, the 20% difference in the EO and PA calculated for WAMIS and MCD45A1 in 2008 (Table 4.2). As established earlier, MODIS products may be expected to miss roughly 30-40% of burned areas under conditions of high tree cover and low combustion completeness (Giglio et al. 2009, Roy & Boschetti 2009). The fact that WAMIS missed 66% of the area burned in 2008, a level of omission that is not only above average, but also does not tally with EO calculated for MCD45A1 (as in all the other three years), gives reason to assume that at an unknown level of tree cover, WAMIS begins to map burned areas well below MCD45A1. Another possibility is that at certain intensities of low combustion fires, WAMIS ceases to map burned areas similarly to MCD45A1. Further investigation into the relationship between rainfall, tree cover, and fire intensity versus burned area mapped by the two MODIS products would assist in establishing these supposed thresholds.

Based on the relatively low variability of the accuracies recorded by MCD45A1 between all four years, and the irregularities presented by WAMIS in 2008 (Table 4.2), MCD45A1 demonstrates more robust performance in mapping burned areas over varying environmental conditions. The MCD45A1 shows greater advantage over WAMIS in this regard, possibly because the specifics of the BRDF based algorithm used for this product facilitates consistent mapping of fires over varying physical conditions in the savannas. Furthermore, in terms of data collection, MCD45A1 product proved to be more readily available and provided sufficient data for the whole 10 year study period. On the other hand, WAMIS data are only available for download from 2008 to current day, with the bulk of fire data for 2010 missing, data prior to 2008 was provided by a personal source. Taking into account the general consistency in accuracies calculated over the four years evaluated and easy access to data, the MCD45A1 product proves to be a more reliable source of burned area data for this study, and will henceforth be used as the primary source of data in the fire regime analysis to follow.

CHAPTER 5: RECONSTRUCTION OF THE FIRE REGIME

Reconstructing a fire regime involves establishing a detailed historical account of elements that describe the nature of fires in a given area e.g. fire frequency, severity, size, pattern and seasonality (Gill 1975, Bond & Keeley 2005). Studies across various African vegetation types, such as Trollope (1993), Heinl (2005) and Devineau et al. (2010), have reconstructed fire histories in order to determine the drivers behind the occurrence of fire patterns and also assess their impact on ecosystems. The purpose of such an undertaking in this task is two-fold. As mentioned in Chapter one, the success of Zimbabwe's National Fire Protection Strategy, has been significantly limited by an inability to collect fire statistics, due to a lack of resources and inaccessibility of some areas affected by fires (Phiri et al. 2011). By quantifying various fire regime variables in this study, the potential benefits of integrating the use of remotely sensed data into the collection of fire statistics in Zimbabwe will be illustrated. Secondly, given the increase in uncontrolled fires since the launch of the Land Reform Program in 2000 (Phiri et al. 2011) and the potential consequent reduction in miombo due to increased fire frequency and or late season fires (Frost 1996, Ryan & Williams 2011), elements of the fire regime, which may be negatively impacting on the viability of the miombo woodland in Charara Safari Area (CSA), will be identified through the reconstruction of the fire history. As established in the accuracy assessment (Chapter 4), the MCD45A1 burned area product showed greater accuracy in mapping fires in this study area, and was therefore utilized in recreating CSA`s historical fire record for the period 2001 to 2010. From this data, information on the number of fire incidences, extent of area burned, seasonality, fire size, fire return interval (FRI) or frequency was compiled.

5.1 FIRE INCIDENCE AND EXTENT OF AREA BURNED

The number of fires and extent of area burned in CSA for every month within the fire season (May to November) was extracted from the MCD45A1 data, for each year between 2001 and 2010. The proportion of area burned for each year was also calculated and summarised alongside area burned, as shown in Figure 5.1.

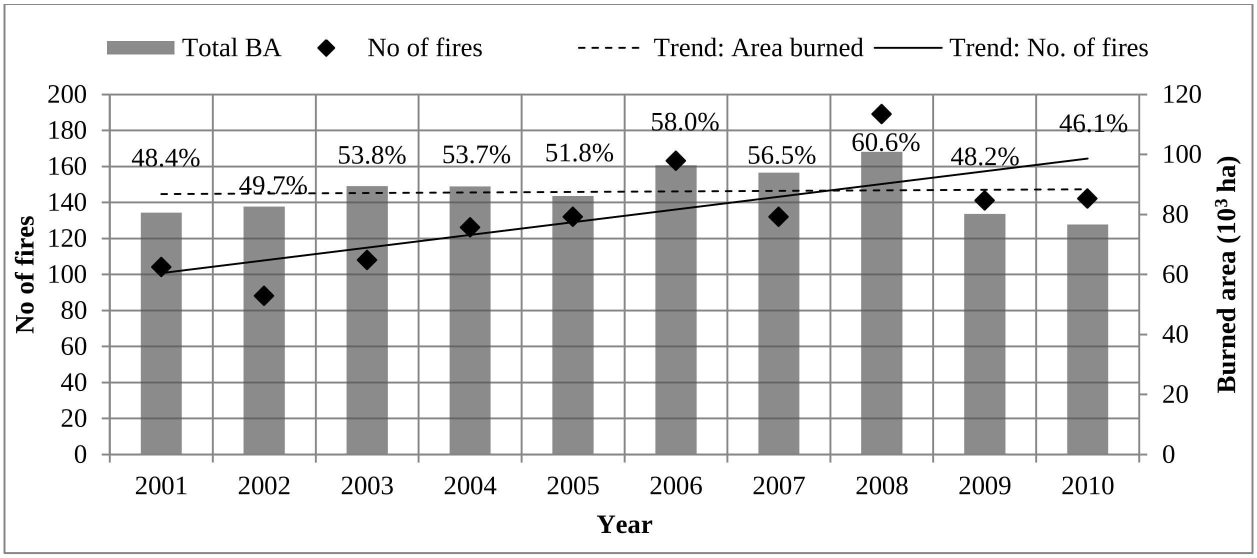

Figure 5.1 Annual number of fire incidences, burned area (grey bars) and percentage area of CSA burned (values shown above the bars) from 2001 to 2010

According to Figure 5.1, fires burned on average 52.7% of CSA annually over the 10 years under study, an equivalent of approximately 87 626 ha per year. The most fire affected year was 2008 where 60.6% of the study area was burnt, while 2010 showed to be the least fire affected year, burning 46.1% of CSA. The number of fire incidences ranged between 88 and 189 fires, with 2002 and 2008 recording the lowest and highest number of fires, respectively. Further assessment of Figure 5.1 indicates that there is no particular trend in the area burned in CSA to suggest a decline or increase in the extent of area affected by fires over the 10 year study period. However, it should be noted that 10 years might not be a sufficient amount of time to make conclusive statements about the trend in total area burned. With such a short fire record, it is difficult to determine whether inter-annual shifts in the extent of area burned may signify an actual departure from patterns in area burned in the past (Ryan 2011, pers com). As was found by Johansson (2011), when performing a time series analysis of area burned in Canada over 41 years, there were no particular trends in the area burned when the time period was separated into 11 years and 29 years, but an increasing trend in burned area immediately became evident when the whole 41 years were incorporated into the time series. Furthermore, according to Niklasson & Granstrom (2000), a general consistency in the percentage burned over time does not necessarily suggest a constant fire regime, while fluctuations in the amount of area burned over time may be a reflection of alterations in fire density and/or fire size. On the other hand, fire incidences in CSA display an increasing trend between 2001 and 2010. This suggests that an increase in fire incidence does not essentially signify an increase in the amount of area burned, but rather may be indicative of variations in fire sizes across the 10 year study period. Therefore, in order to observe meaningful trends in area burned and fire incidence, further analysis was carried out on a monthly basis (Figure 5.2) and also in the context of fire size, as will be discussed in the following sections.

5.2 SEASONAL PATTERNS OF FIRES IN CSA

Fire extent, date and location derived from the MCD45A1 product facilitated the establishment of seasonal patterns of burning over the 10 year study period in CSA. These patterns served as indicators as to whether or not the nature of fire occurrence in CSA may be posing a potential risk to the miombo woodland.

5.2.1 Fire seasonality

According to Archibald et al. (2010a) fire seasonality may be used in reference to the length of the fire season within a given area. Following Hongxiao (2010), fire occurrence in southern Africa runs parallel to the dry season from April to November, therefore the fire season is generally expected to be seven months long. Data on the number of fires and extent of area burned, derived from the MCD45A1 product, was utilised to describe fire seasonality in CSA as illustrated in Figure 5.2. The trend in the amount of area burned and number of fire incidences mapped over 2001 and 2010 in CSA generally corresponds to studies, such as Pereira et al. (2004), which describe burning in the miombo woodlands as one which begins in May, at the onset of the dry season and ends in November, at the arrival of the wet season, with peaks in fires expected between mid-July and October (Figure 5.2).

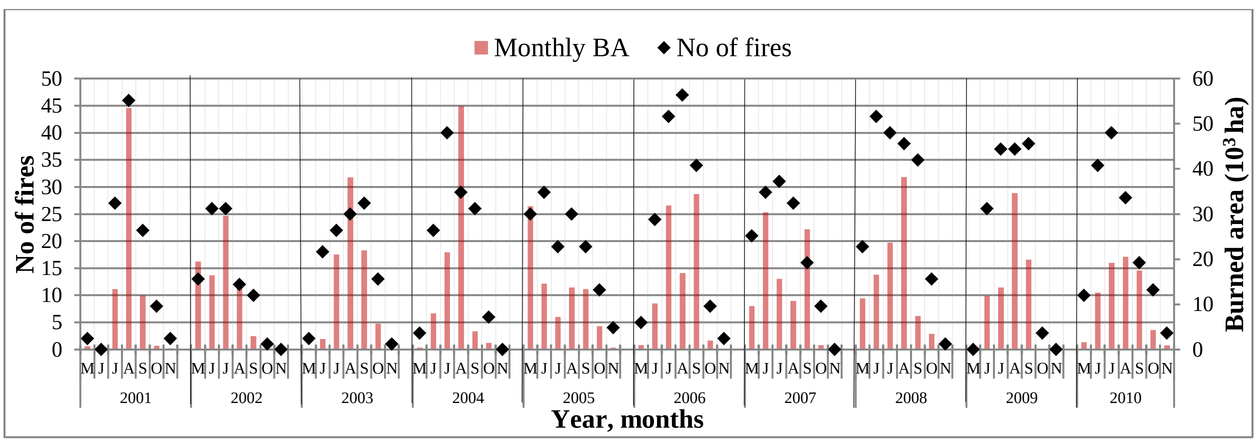

Figure 5.2 Number of fires and monthly burned area derived from MCD45A1 for Charara Safari Area

As displayed in Figure 5.2, the majority of the peaks in area burned occur in August; this is typical of late season fires which occur at high temperatures, when the fuel load is extremely dry. Fires which burn in the months August to October are suggested to have damaging effects on savannah ecosystems (Laris 2002), as they burn at high intensities and are usually a combination of both surface and crown fires, thus affecting the ability of woody vegetation in the miombo to recover from the fires (Ryan & Williams 2011). However, burned areas in the years 2005 and 2007 show patterns common to early season fires which burn mostly between May and June (Figure 5.2). Such fires are less intense and slow burning, and if they occur frequently may lead to the formation closed canopy woodland (Frost 1996). The occurrence of early season fires in CSA may be attributed to the escape of fires originating from neighbouring communal lands, induced by farmers to initiate growth in the perennial grasses (Mbow et al. 2000, Bucini & Lambin 2001), into CSA. Also, it is possible that in CSA, being a state-owned hunting area (Vorlaufer 2002), fires may be lit in the early season by game rangers to draw out or drive wild animals during the hunting season (Frost 1996). Secondly, early season fires may have been used in CSA as part of a prescribed burning program commonly used in protected areas to reduce the destructiveness of late dry season fires (WWF 2001). As explained by Eriksen (2007), early season fires fragment the landscape with small patches of burned areas which act as fire breaks later in the hot season. Figure 5.2 also shows that the bulk of the fires in CSA burn between May and August, indicating that these fires originate mainly from anthropogenic activities, as natural fires caused by lightning would be expected to occur closer to the rainy season in October and November (Heinl 2005, Archibald et al. 2010a).

Another important aspect in fire seasonality is the interplay between early and late season fires in a given fire season. As noted earlier, late season fires have been cited as potentially damaging to the miombo as they burn through larger areas, and at high intensities which reduce the regenerative ability of the miombo (WWF 2001). On the other hand, early season fires lead to the formation of a closed canopy, establishment of fire sensitive tree species and gradual suppression of grasses. However fire alone cannot disrupt the development of the miombo, but may retard it (Frost 1996). Therefore, a fire regime defined by a combination of both early and late season fires is essential to the maintenance of the miombo woodland. Reports of increasing fire incidences in Zimbabwe since 2002 (Matamera 2011), particularly in Mashonaland west province (Bara 2012), where CSA is located, gives reason to assume that there might have been a shift in the fire regime from early season fires to late season fires over the 10 year study period. As explained in Section 2.1.3, more fires tend to occur in the late season than in the early season. A comparison of fire data, prior to and after 2000, would enable a more conclusive investigation into the possible shift in the seasonal occurrence of fires in CSA; however fire records dating before 2000 were not available from MODIS. Nonetheless, the length of fire season (seasonality) and months of peak fire activity between 2001 and 2010 were summarised, as shown in Table 5.1, and seem to indicate that there was no gradual shift in the fire season i.e. early to late season fires, in the period 2001 to 2010 in CSA.

Rather, early and late season fires alternated without any particular trend, between the 10 years. Early season fires were identified by peak fire activity occurring between months: May to July, and late season fires were defined by peaks in the months: August to October.

Table 5.1 Summary of fire seasonality and peak fire activity. Shaded months represent late season fires

Year	**Length of season**	**Month of no fire event**	**Peak month- No. of fires**	**Peak month- Area burned**
2001	6 months	June (technical issue) *	August	August
2002	6 months	November	June	July
2003	7 months		September	August
2004	6 months	November	July	August
2005	7 months		June	May
2006	7 months		August	September
2007	6 months	November	July	June
2008	7 months		June	August
2009	5 months	May and November	September	September
2010	7 months		July	August

*No burned area data was available for June 2001 due to a technical fault on the MODIS satellite.

Further examination of Table 5.1, revealed that the number of fires and area burned in eight out of the ten years studied reached peak level in different months of the same year. It was also observed, that the length of the fire seasons across the 10 years generally conformed to that which is cited in Frost (1996), where fires in miombo woodlands burn for seven months, between May and November. In some years, however, fires did not occur in the month of November, most likely due to the onset of the rainy season around this period. In 2009, the fire season was much shorter spanning between June and October. It is assumed that fires in 2009 may have started later in June, due to high moisture content in the grass fuels and or cool weather conditions in May, caused by a prolonged rainy season (Adler et al. 2003), thus restricting the igniting of fires until June. As explained in (WWF 2001) the length of a fire season depends on the past rainy season and onset of current rainy season.

While the trend in peak fire activity over the 10 year study period indicates that there is no gradual shift from early to late fire seasons in CSA, a comparison of the amount of area burned and number of fires that occur in each of the two parts of the fire season gives added insight into the potential ecological risk posed by fires that burned over the 10 year period towards the viability of the miombo in CSA, as shown in Figure 5.3 and Figure 5.4.

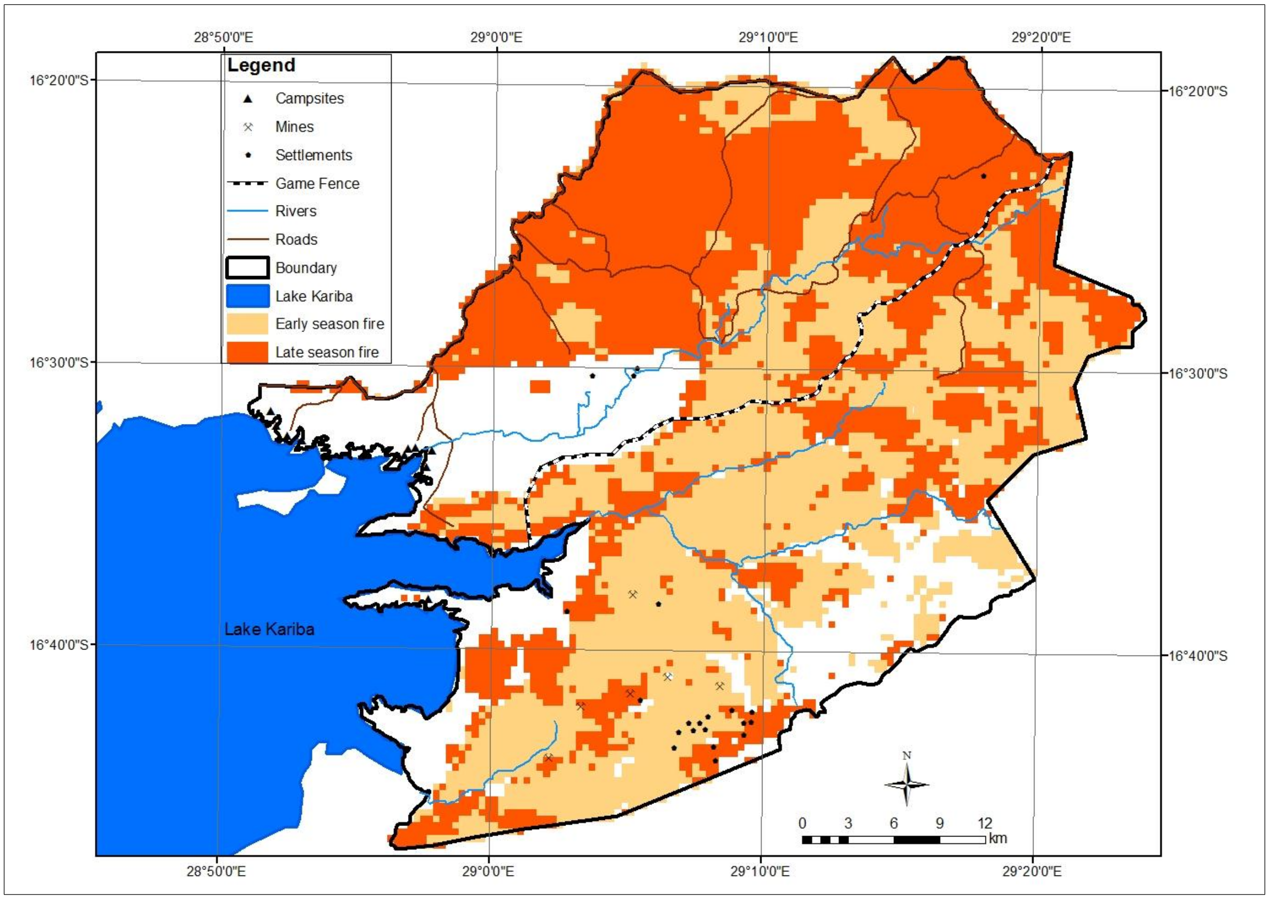

Figure 5.3 Proportion of total area burned by early season and late season fires in CSA from the period 2001 to 2010

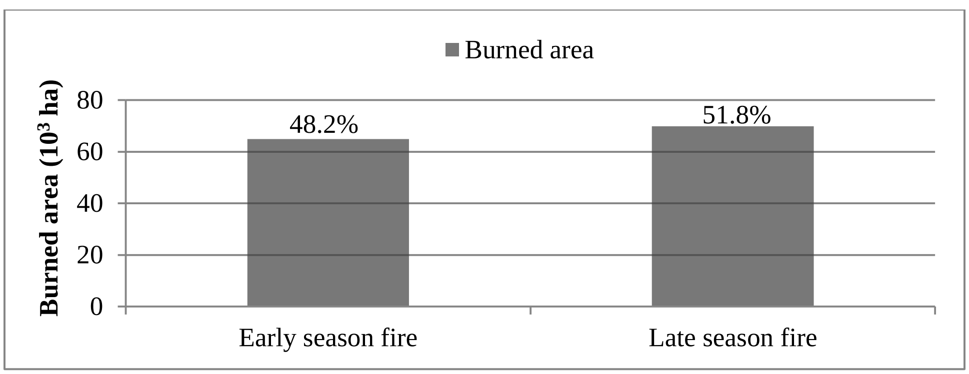

Figure 5.4 Proportion of area burned by early and late season fires between 2001 and 2010 in CSA

As illustrated in Figure 5.3 and Figure 5.4 both early and late season fires contributed almost similarly towards the total area burned throughout the 10 year study period, with early and late season fires accounting for 48.2% and 51.8% of total burned area respectively. However it was observed that the bulk of the late season fires occurred in the northern section of CSA, while early season fires dominated the southern region. The distribution of early and late season fires in CSA may be a result of the communal settlements and mines located south of the game fence, particularly communal farming, while the area in the north is generally uninhabited and reserved for wildlife. Such trends have been evident in studies such as Laris (2002) where rural communities use early season fires to initiate regrowth in grass biomass for their livestock and also as a protective and preventative measure against late season fires. Early season fires may also be evidence of prescribed burning in different parts of CSA. The presence of late season fires north of the game fence in CSA (Figure 5.3), could suggest that in the absence of human settlements the fire regime in CSA is presumably characterised by late season fires, as fires in Zimbabwe are generally known to occur under hot, dry conditions when fuel moisture is low (WWF 2001). Also, the lack of fires near the lake shore, south west of CSA may be attributed to the presence of mopane woodland closer to the shoreline. The mopane generally thrives on alluvial soils (Shumba 2001), and is characterised by a poorly developed grass layer consisting of short, annual grasses and therefore provides little fuel for fires (Silber 2011).

Figure 5.5 shows that the majority of the number of fires recorded between 2001 and 2010 occurred in the early season, as shown in Figure 5.5 (b) a greater proportion of the number fires burned in the early season in six out of ten years. When assessing the seasonal pattern of fire occurrence such a result may be expected, as early season fires in the savannas are characterised by many small fires, while late season fires have fewer but larger fires (Archibald et al. 2010a). Calculating the actual area burned by these fires occurring both in the early and late season may be of more significance, particularly in the context of area burned by late season fires, which are considered to be potentially damaging to the miombo woodland (Ryan & Williams 2011). Figure 5.5 (c) and (d) indicate that in six out of ten years, late season fires accounted for a larger proportion of the area burned in CSA. These late season fires burned as much as 66.5% of the area burned in 2001 to 32.6% in 2005. It is not unusual that the fire regime in CSA is dominated by late season fires as the majority of the fires in miombo woodlands are known to occur during the hot dry season i.e. August, September and October (Chidumayo 1993). However, as explained by Frost 1996, late season fires may lead to the conversion of miombo woodland into open, tall grass savannas. In addition to changes in vegetation structure, late season fires may also cause changes in vegetation composition. For instance, fire tolerant species such as *Pterocarpus angolensis* and *Erythrophleum africanum* are favoured by late season fires, and consequently increase in their presence, while the dominant miombo tree species (e.g: *Brachestygia spiciformis*, *Isoberlinia angol*ensis and *Julbernadia paniculata*), which are fire-tender, tends to decrease (Trapnell 1959). Therefore it is possible that the structure or composition of the miombo in CSA may have been altered in areas burned by late season fires over the 10 year study period.

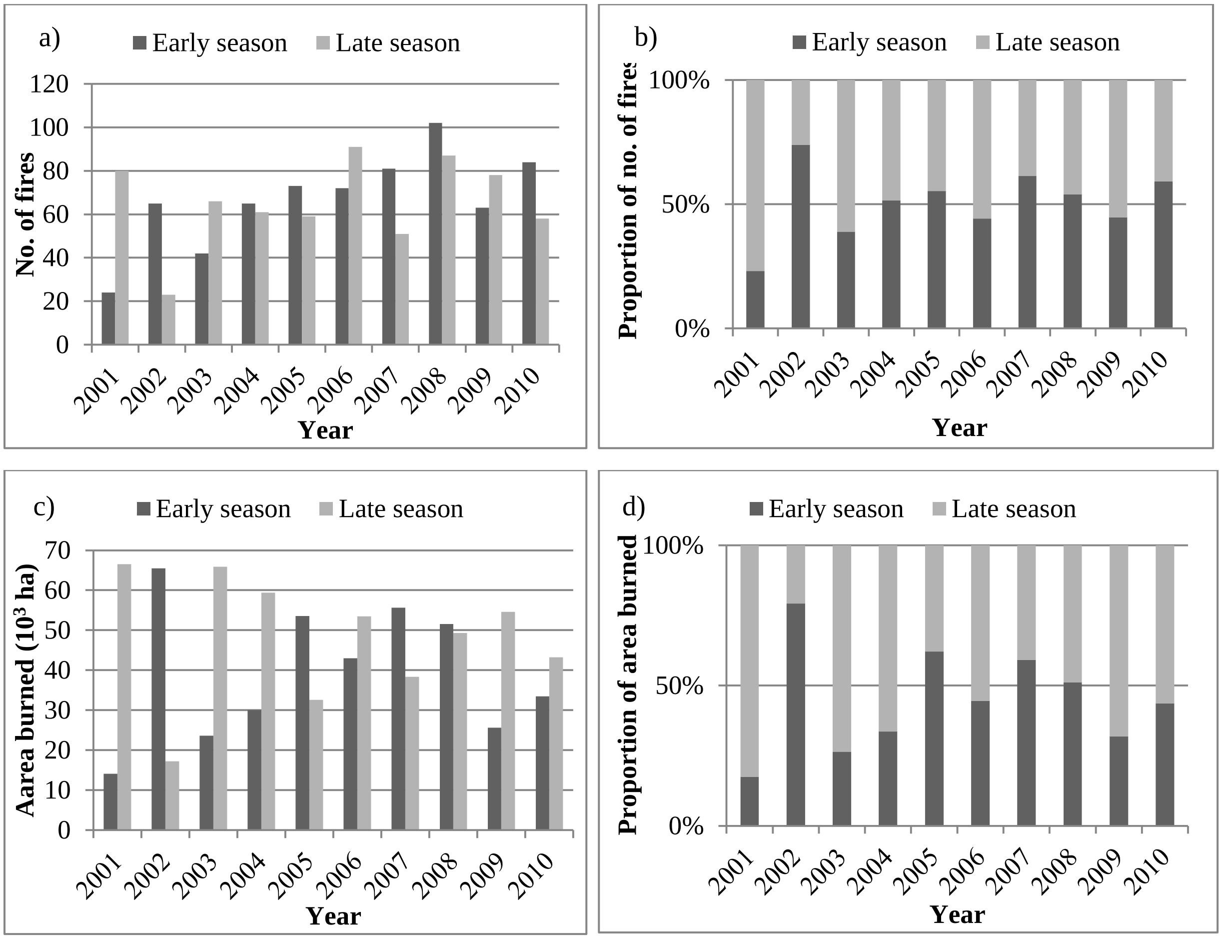

Figure 5.5 Number and proportion of fires and area burned in the early and late season from 2001 to 2010

5.2.2 Inter-annual variability of fires

Inter-annual variability is a measurement commonly used in fire regime analyses to determine the extent and cause of variations in fire patterns over a given period (Archibald et al. 2010b, N'Datchoh et al. 2012). The inter-annual variability of fires mapped throughout the fire season in this study, was determined by plotting the number of fires and area burned in each year between 2001 and 2010 against months within the fire season: May to November (Figure 5.6). The mean number of fires, standard deviation and coefficient of variation thereof was calculated for each fire season month across the10 year study period.

According to Figure 5.6, the inter-annual variability in the number of fires per month mapped across the fire season in the years 2001 to 2010 ranged between 1 and 11 fires, as reflected by the standard deviation.

June and August displayed the greatest level of variability, each deviating from the mean by 11 and 10 fires respectively. On average, 210 to 29 444 ha of land burned throughout each fire season in the 10 year period, with August showing greatest variability (SD = 16 400 ha) and November the least variability (SD = 290 ha). Standard deviation in the number of fires recorded from May to September in all 10 years under study was closely similar, ranging between 9 and 11 fires, while, deviation from the mean dropped to 4 fires in October and 1 fire in November (Figure 5.6). The occurrence of lower standards of deviation at the end of the fire season was also evident in the area burned. The sudden decline in the inter-annual variability of fire incidences and area burned in October and November, may be an indication of a consistency in the factors influencing the occurrence of fires at the end of the fire season in CSA, throughout the 10 year period of study. Zumbrunnen et al. (2012) cite three driving factors behind variability in a fire regime: fuel load, climate and ignition. Therefore, low standard deviation at the end of the fire season, suggest that there was a minor variation in the amount of fuel load available (possibly as most of the grass fuel would have been burned by October), climatic conditions (onset of the rainy season) and source of ignition in the months of October and November across the 10 years of study. Thus fires occurring at the end of the fire season tend to be more predictable because they show less variation from the mean than fires that occur between May and September.

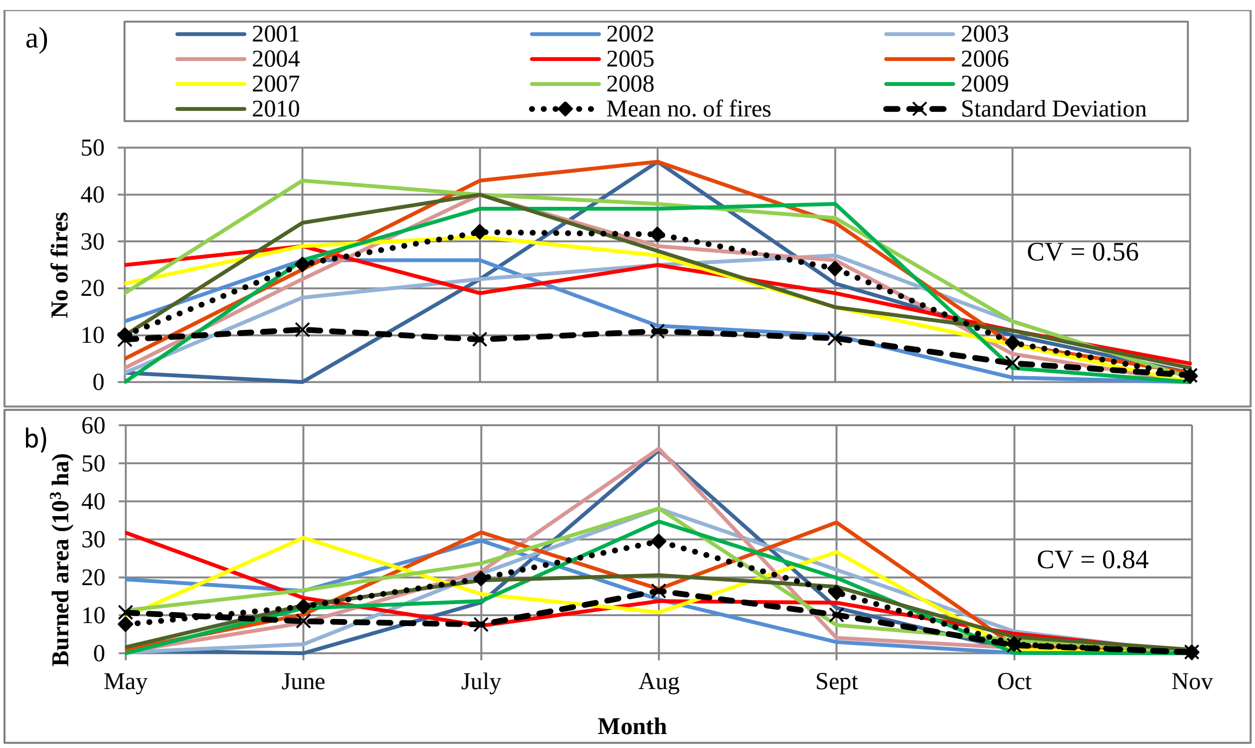

Figure 5.6 a) Number of fires and b) area burned per month for each year from 2001 – 2010

SD = standard variation and CV = coefficient of variation (SD/mean value)

A visual inspection of Figure 5.6 shows that fluctuations in the standard deviation for area burned are more distinct than those displayed by the number of fires; this suggests that there is greater inter-annual variability in the amount of area burned than the number of fires that occur throughout the fire season. However, because these two variables are measured in different units, the coefficient of variation (CV), described as the ratio between the standard deviation and mean of a variable, was calculated for each month in order to allow comparison in the levels of variability in both the number of fires and area burned. An average of the CV calculated indicated that CV was higher in the amount of area burned than in the number of fires, measuring 0.84 and 0.56 respectively (Figure 5.6). This confirms that there is greater deviance from the mean in the extent of area burned than in the number of fires that occur between the 10 years throughout the fire season. Furthermore, this also corresponds to findings by Parisien et al. (2004), where the inter-annual variability in burned area was also greater than the number of fires in the boreal biome.

5.3 FIRE SIZE DISTRIBUTION

More insight into the trends in burned area was derived from separating the burned area into 10 fire size classes. Studies such as Parisien et al. (2004) and Cui & Perera (2008) have used fire size classification in fire regime analyses to determine the contribution of fire sizes to total burned area and identifying periods where potentially damaging fires occur. Following Archibald et al. (2009) fire sizes selected for this study were classified as follows: < 25ha, 25 – 100 ha, 100 – 250 ha, 250 – 500 ha, 500 – 1000 ha, 1000 – 2500 ha, 2500 – 5000 ha, 5000 – 10000 ha, 10000 – 25000 ha, and > 25000 ha. Each fire size class was named by letters A to J respectively (Table 5.2). The number of fire incidences and proportion of area burned by each fire size class over the 10 year period is illustrated in Figure 5.7.

Table 5.2 Fire size classification

Class	A = < 25 ha	B = 25 – 100 ha	C = 100 – 250 ha	D = 250 – 500 ha	E = 500 – 1000 ha
Class	F =1000 – 2500 ha	G = 2500 – 5000 ha	H = 5000 – 10 000 ha	I = 10 000 – 25 000 ha	J = > 25 000 ha

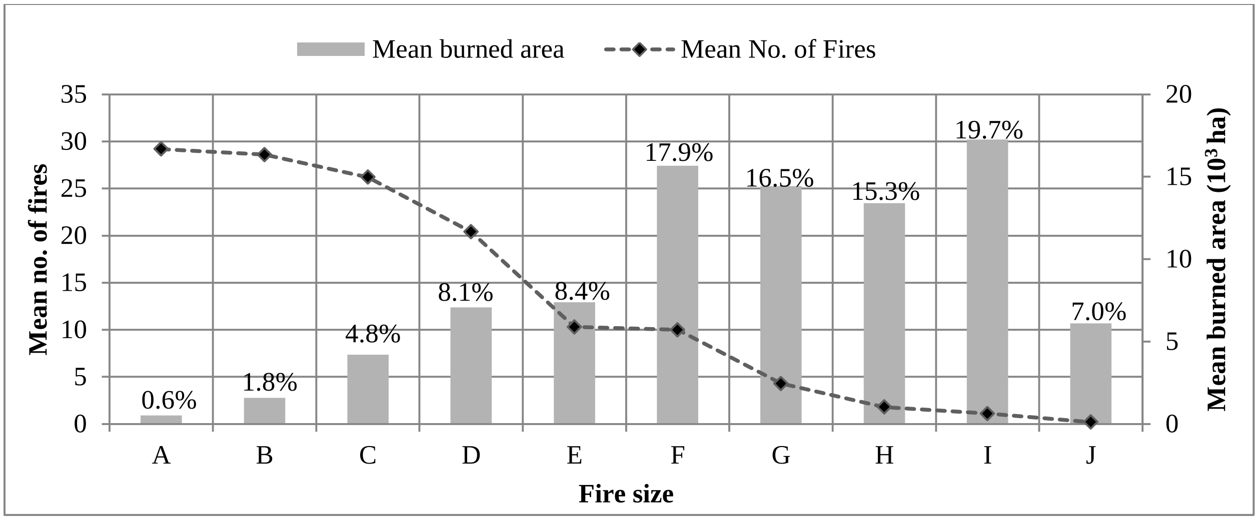

Figure 5.7 Mean number of fires and area burned per fire size class recorded for the period 2001 to 2010 Values on top of bars represent the proportion of area burned by each corresponding fire size class in the 10 year period.

Figure 5.7 shows that there is an inverse relationship between the number of fires identified in each fire size class and the corresponding amount of burned area. As fire size increases, the number of fire incidences decline. This trend between number of fires and fire sizes was also identified in the seasonal fire patterns mapped by Diaz-Delgado & Pons (2001), northeast of Spain. It also evident from Figure 5.7 that a majority of the fires occurring in CSA between 2001 and 2010 were characterised by small to medium sized fires (>25 - 1000 ha), while fires >1000 ha were represented by fewer numbers of fires. On the other hand, small to medium sized fires accounted for a lesser proportion of the area burned in CSA while fires between 1000 and 25 000 ha burned 76.4% of the total area burned in the safari (Figure 5.7). It is difficult to compare these findings with other studies on fire size, because what constitutes a small or large fire varies across ecological regions and size of the study area. For example, van Wilgen et al. (2010) define large fires in the fynbos region as fires exceeding 10 000 ha, Parisien et al. (2004) described large fires as 1000 ha or more in boreal forests and Archibald et al. (2010a) refers to fire sizes of 2500 ha as large and 10 000 ha as very large in Southern Africa. Nonetheless, studies such as Diaz-Delgado & Pons (2001) and Forsyth & van Wilgen (2008), although with different classifications of fire size, draw similar conclusions on fire size as those made in this study, i.e. while the bulk of the number of fires recorded were small fires, these small fires burned a relatively small amount of the area, whereas larger fires, which were fewer in number, burned the greater part of the study area. It is also worthy to note that although it has been established that fires between 1000 ha and 25 000 ha in size burned over 50% of the area in CSA

(Figure 5.7), because the relationship between fire size and ecological damage to the miombo is unknown for the study area, it is not possible to comment on the potential risk of these fires.

The annual trend in all 10 fire size classes based on the number of fires and extent of area burned between 2001 and 2010 was also plotted (Figure 5.8).

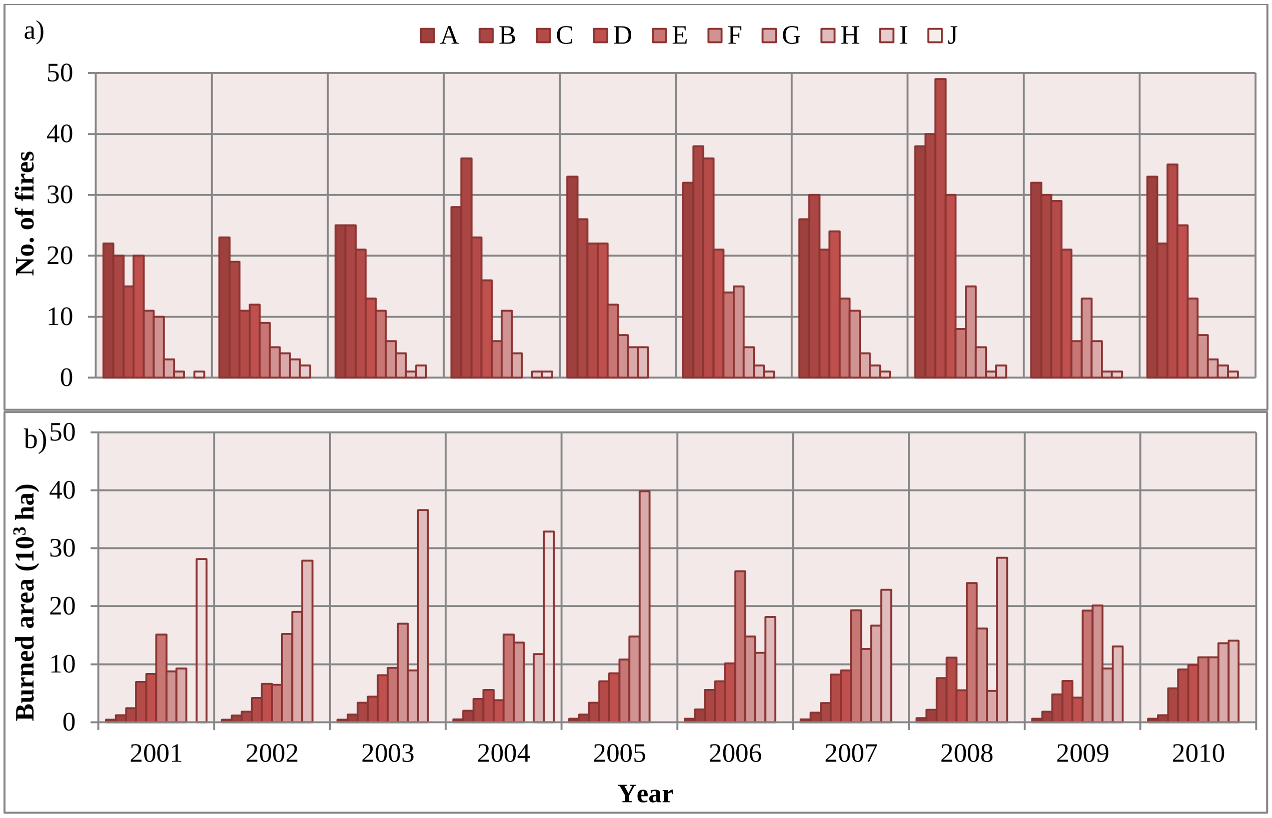

Figure 5.8 (a) Number of fire incidences and (b) area burned per fire size class between 2001 and 2010

It was observed from Figure 5.8 that fire sizes ranging between 5000 and 25 000 ha did not occur in every year i.e. no fires were recorded in size class H in 2004, size class I in 2001 and 2005, and more significantly, fires in size class J which were only recorded in 2001 and 2004. Therefore, fires greater than 25 000 ha can be considered to be rare and extreme fire events in CSA, which generally occur in dry, hot and windy weather conditions, following a good rainfall season (Archibald et al. 2009). This is confirmed by Figure 5.2, which shows that the largest extent of area burned in 2001 and 2004 was in August. August in Zimbabwe is characterised by high temperatures (26 to 36^0 C), windy conditions and large amounts of dried grass (WWF 2001), conducive to the ignition of large fire events. Figure 5.8 also displays a steady increase in the number fires and extent of area burned in the small to medium size classes (A to E) throughout the 10 year study period, while the rest of the fire size classes remain relatively constant.

Fire sizes were also assessed on a seasonal basis by plotting the total number of fires and area burned in each month of the fire season between 2001 and 2010 for each fire size class. From Figure 5.9, it can be established that, the number of fire incidences in each fire size class generally increase from May, maintain a steady rise until September, and thereafter begin to decrease towards the end of the fires season in November. This corresponds to fire size patterns noted by Archibald and co-workers (Archibald et al. 2010a) for Southern African savannas. Figure 5.9 (a) illustrates the dominance of small to medium sized fires in the number of fire incidences throughout the fire season, with fires between 25 and 1000 ha (A to E) accounting for the majority of the fires mapped. In contrast, fire sizes greater than 1000 ha (F to J) contributed to the majority of area burned. This suggests that while the bulk of the number of fires that occur in CSA are small to medium sized fires (A to E), it is the larger fires (F to J) that contribute to majority of the total area burned. Lastly, according to Figure 5.9 (b), not all fire size classes burned throughout the whole fire season, for instance fires in class I were confined to mid fire season: July to September while, fires in class J burned only in August. This reaffirms the suggestion that large fire events tend to occur in the hottest and driest part of the fire season.

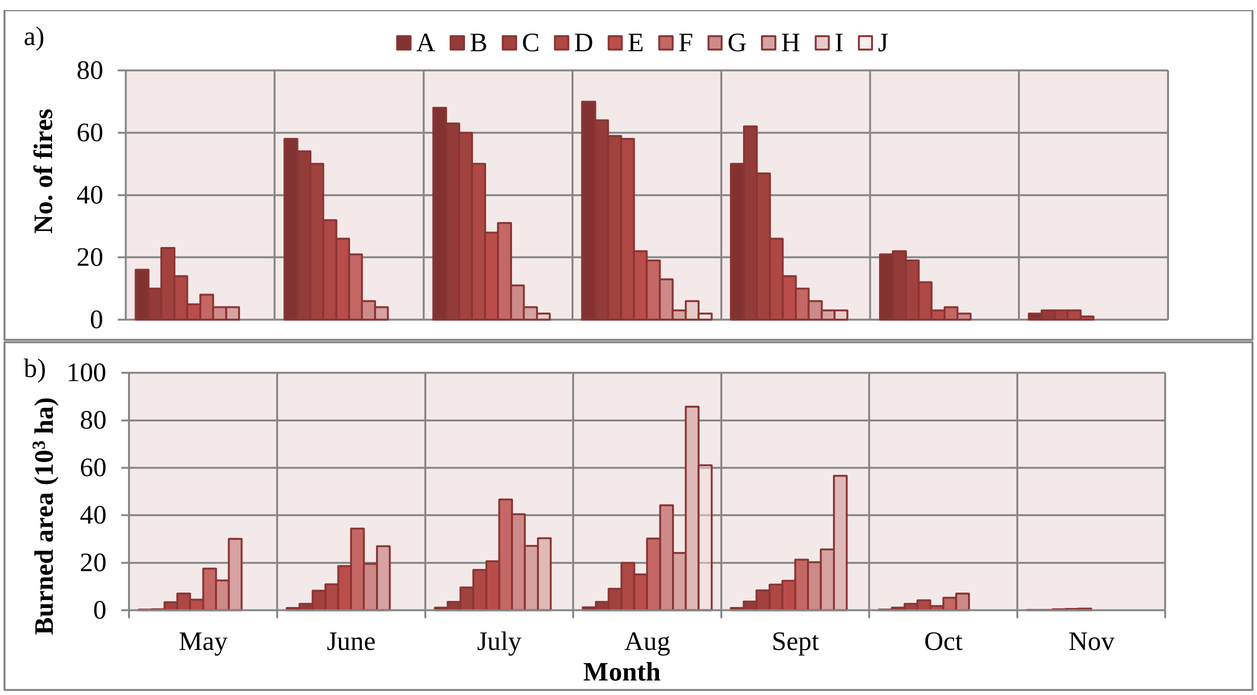

Figure 5.9 (a) Total number of fires and (b) area burned between 2001 and 2010 in each fire size class per month throughout the fire season

After evaluating trends in fire sizes, both on an annual and seasonal basis, the effective fire sizes for each year were calculated to assess if there was a change in the contribution of different fire sizes to the total burned area, over the 10 year study period. Effective fire size is defined by Johansson (2011), as the fire size that contributes the most to total area burned in a given period, and is useful in fire management, particularly in determining ecologically appropriate fire sizes for a given study area and also identifying anomalies in the occurrence of certain fire sizes.

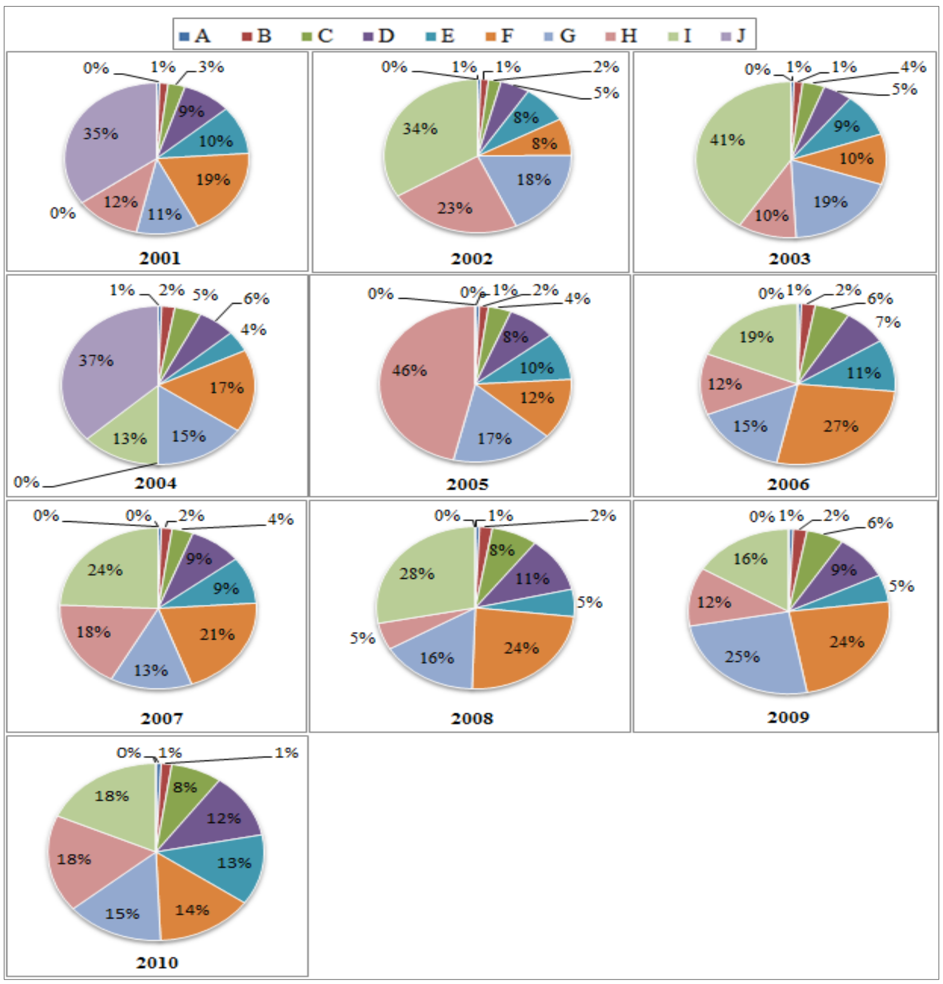

Figure 5.10 Contribution of fire sizes towards total burned area per year

Figure 5.10 shows that the effective fire sizes for the whole 10 year study period were predominantly class sizes H, I and J (5 000 – >25 000 ha), with class size I being the dominant effective fire size. However, the contribution of these fire sizes towards total area burned in each year show a decreasing trend from 2005 onwards. In the first half of the study period effective fire sizes: H to J, accounted for 34 to 46% of the total area burned in each year, but from 2006 to 2010 they contributed between 18 and 27% of the total area burned (Figure 5.10). This shows that there was a shift towards a fairly even distribution in the contribution of fire sizes to area burned in CSA in the 10 year study period. More so, while larger fire sizes (H, I and J) accounted for over 46% of the area burned between 2001 and 2005, medium sized fires, i.e. E, F and G (500 - 5 000 ha), gradually accounted for greater portions of total area burned from 2006 onwards. This illustrates a shift towards a fairly even contribution of medium to large fires (500 – 25 000 ha) towards total area burned, in the latter half of the 10 year study period (Figure 5.10).

5.4 FIRE RETURN INTERVAL

FRI is defined as the time taken for a fire event to reoccur at a given point (Hely & Alleaume 2006), and is a significant indicator of possible modifications in vegetation structure and floristic composition, as demonstrated by Govender et al. (2006), Masocha et al. (2011) and Ryan &Williams (2011). Such changes are particularly important in protected areas such as CSA, as they could translate to disruptions in ecosystem function, thus leading to potential socio-economic consequences with regards to its role as a safari area e.g. no game to hunt and also surrounding communities whose livelihoods are dependent on the natural resources derived from CSA. The average FRI for miombo woodland ranges between two and four years (Frost 1996, Scholes et al. 1996 and Barbosa et al. 1999b), therefore the 10 year data set derived from the MCD45AI product was considered sufficient in determining the FRI for CSA. Burned data from 2001 to 2010 was merged together to calculate fire frequency (Figure 5.11 and Figure 5.12), expressed as the number of times a pixel burned over the 10 year period (Archibald et al. 2010a), and the corresponding affected area. The maximum number of times a pixel burned over the 10 year period was found to be 10 times; therefore fire frequency in CSA was classified into 10 classes (i.e. Class 1, 2, 3…10) each referring to the number of times a pixel burned between 2001 and 2010. The FRI and affected area was also calculated from the merged burned data (Figure 5.13 and Figure 5.14). This variable was expressed in years and classified into four classes: 0 – 2 years, 2 – 4 years, 4 – 8 years and 8 – 10 years.

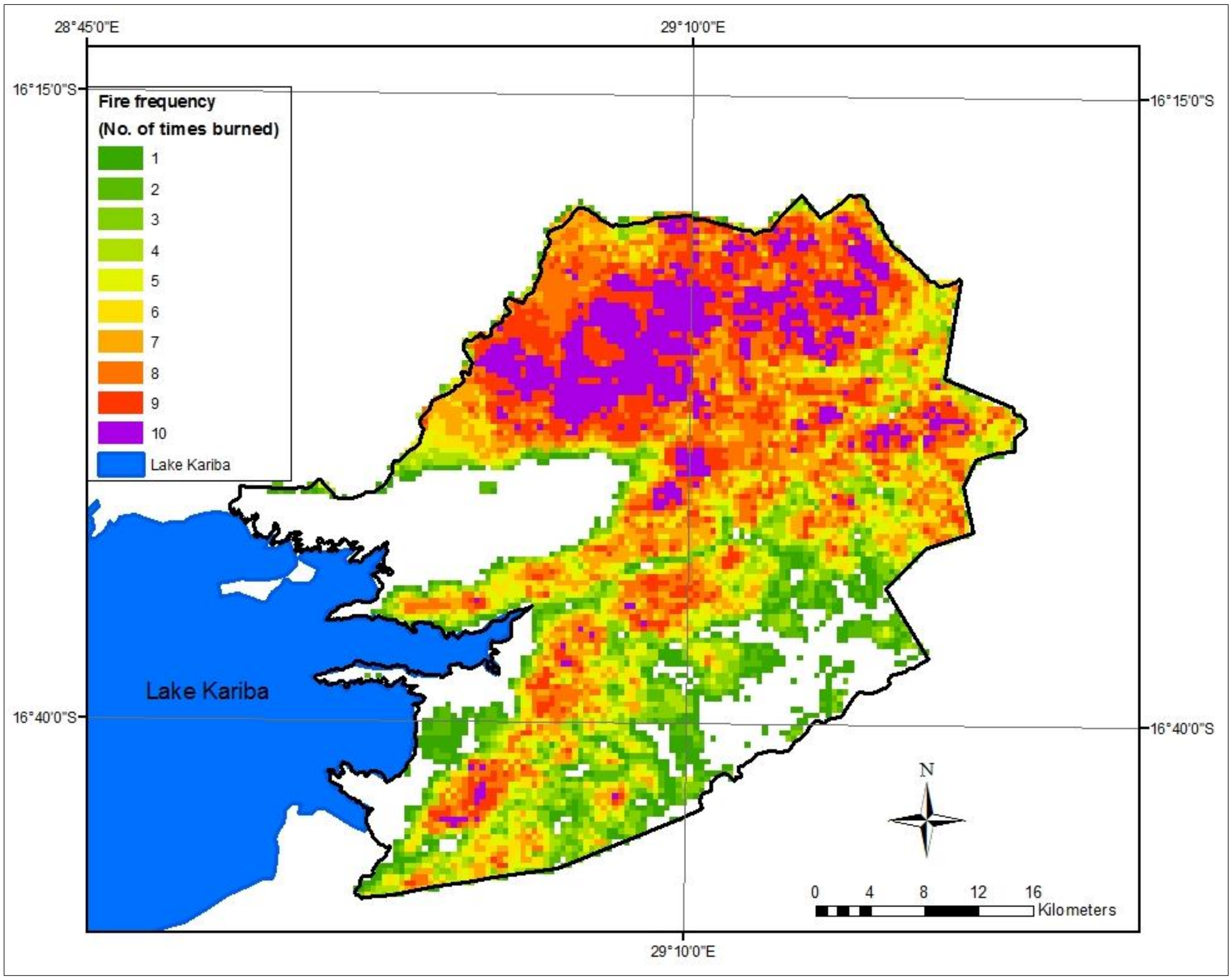

Figure 5.11Fire frequency on CSA (expressed as the number of times a pixel burned over the 10 year study period)

Figure 5.12 Fire frequency and proportion of total area burned 1-10 times between 2001 and 2010

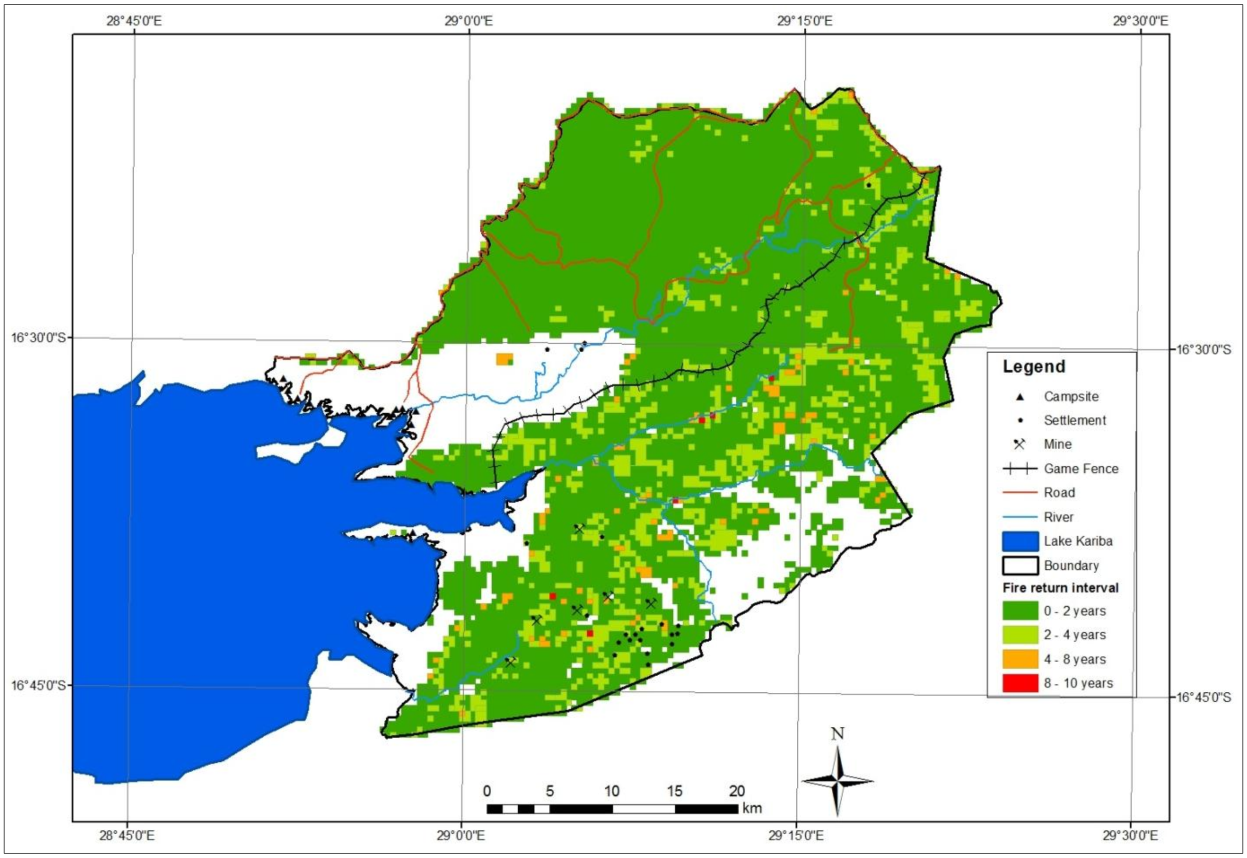

Figure 5.13 FRI calculated for CSA between 2001 and 2010

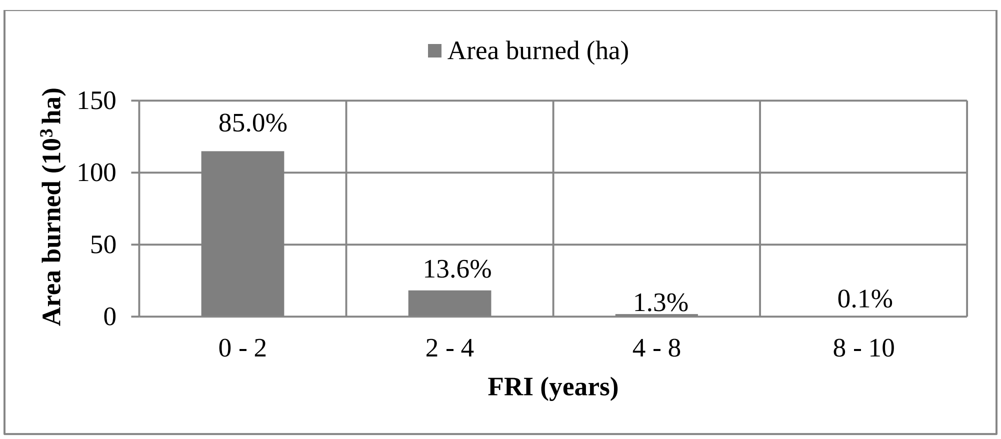

Figure 5.14 FRI and proportion of total area burned by each FRI class: 0 – 2 years, 2 – 4 years, 4 – 8 years and 8 – 10 years

As shown in Figure 5.12, 64.2% of the total area burned in CSA burned between 6 and 10 times over the period 2001 to 2010. Frequencies between 1 and 5 times were largely dominant in the southern parts of

CSA, and higher frequencies (6 – 10 times) were most evident in the northern parts of the safari, while fires did not occur closer to the shores of Lake Kariba on the western side of CSA. The distribution of FRI illustrated in Figure 5.13 is almost similar to that of fire frequency shown in Figure 5.11, with a short FRI (0 – 2 years) dominating the northern regions of CSA, while a moderate FRI was observed in the southern and north eastern section of CSA, and a moderate to long FRI (>2 – 10 years) mostly confined to the southern parts of the CSA.

The distribution in FRI across CSA is partially explained by Nielsen & Rasmussen (2001), who observed, in the savannas of Burkina Faso, that the higher the land use intensity, the lower the fire frequency, and hence the longer the FRI. As shown in Figure 5.13, this is the case in the southern part of CSA, which consists of mines and small communal settlements, where FRI longer than 2 years is evident. According to Hudak & Brockett (2004) and Devineau et al. (2010) fires may be actively suppressed by farmers in communal areas to preserve forage for their livestock and also through intensive agricultural activities which result in the reduction of fuel loads and therefore the prevention of fires in some years. Furthermore, areas of FRI ranging between 2 and 10 years in CSA may be an indication of patches of closely knit, mature *Brachystegia*, or other dominant tree species in the miombo woodland. As noted by Frost (1996), mature woody plants (>2 m) in the miombo are relatively resistant to fire, and tend to limit grass growth in the understory as they develop, thus resulting in the reduction of fire occurrence and hence, longer fire return intervals may be expected. On the other hand, the area north of the game fence, which is mostly defined by a short FRI, is largely uninhabited. This observation corresponds to findings by Devineau et al. (2010), where FRI in the absence of human activity was found to be shorter than FRI in inhabited areas. Prescribed annual and biennial burning may be considered as an explanation for the short FRI particularly evident in the north of CSA. Such frequent burning might have been employed to provide fodder for large herbivores resident in CSA such as the Cape buffalo (Victor Hunting Safaris), and also ensure game visibility for visitors (Hudak & Brockett 2004). However, according to Trollope (1993) the accumulation of grass biomass, following periods of above average rainfall, combined with grazing, significantly influences FRI, while fire management practices, such as prescribed burning, have minimal impact on FRI (van Wilgen et al. 2004). Therefore, the short FRI (<2 years) observed in CSA may be better explained by the annual occurrence of rainfall over CSA, which in turn facilitates the growth and senescence of grass biomass over a year, and consequently leads to shorter FRI (Archibald et al. 2010a).

The FRI observed in CSA is a significant cause for concern. A FRI of approximately 3 years is generally considered to be acceptable in maintaining a functioning savanna ecosystem (Furley et al. 2008), however the fact that only 13.6% of the total area burned in CSA has a FRI ranging from 2 to 4 years, while 85% of the total area burned is defined by a FRI of less than 2 years may have negative implications on the resilience of the miombo in CSA (Figure 5.14).

The consequences of short FRI in miombo woodlands are discussed by various studies (Masocha et al. 2011, Ryan & Williams 2011) based on a long fire experiment carried out in Marondera Grasslands Research Station, Zimbabwe. From this 50 year experiment it was found that plots subjected to annual and biennial burning experienced alterations in vegetation structure, density and tree composition vegetation (Furley et al. 2008). According to Ryan & Williams (2011), a one year FRI over the 50 year period, regardless of the intensity of the fire, resulted in the complete eradication of woody biomass, and conversion to a grass and shrub dominated ecosystem. Therefore, such a change may also be expected for CSA in areas that burned 10 times over the 10 year study period (Figure 5.11), although the length of time required for annual burning to eventually convert woodland to grasslands is unknown. Masocha et al. (2011), also suggests that a FRI of <2 years may promote the invasion of alien tree species that are more tolerant to frequent fires than the dominant miombo tree species e.g. *Lantana camara* and *Jacaranda mimosifolia*, and therefore pose a threat to the natural biodiversity of CSA. Lastly, another cause for concern is that a short FRI may reduce the resilience of the miombo against fires in CSA. For example, studies carried out in the south eastern savannas of Zimbabwe (Nefabas & Gambiza 2007, Gandiwa & Kativu 2009) found that short FR1 retards the growth of dominant tree species, and consequently exposes the tree canopies to the flame zone, thus increasing their susceptibility to fire events.

CHAPTER 6: CONCLUSION

6.1 DISCUSSION

Reports of an increase in uncontrolled veld fires in Zimbabwe has given rise to concerns over the loss of lives and property, and consequently the launch of the National Fire Protection Strategy (NFPS) in 2006 (Phiri et al. 2011). However, according to Chigurah & Jerie (2010) the effectiveness of the NFPS has been hindered by factors such as: discrepancies in fire reports, lack of both human and financial resources to collect fire data and the inaccessibility of some fire affected areas. This study therefore proposes the use of the open source remotely sensed data (i.e. MODIS burned area product) as an alternative, cost effective and timeous method to provide information on the occurrence of fires in Zimbabwe. Furthermore, while much attention has been focused on the short term impacts of these veld fires, less attention has been given to the long term ecological ramifications of these fires on Zimbabwe`s ecosystems, particularly the miombo woodlands which account for approximately 45.2% of the country`s total land area (Olsen et al. 2001). The research therefore highlights the specific potential ecological risks of these fires towards the sustainability of the miombo. This was achieved by drawing parallels between aspects noted in the fire regime in Charara Safari Area (CSA) and how they may affect the miombo based on existing literature pertaining to the fires in the miombo.

In order to demonstrate the potential of the MODIS burned area product in providing fire data to meet the demands of Zimbabwe`s current veld fire crisis, a 10 year fire regime was reconstructed and analysed for CSA to give a historical account of the fires that occurred within the safari. Prior to the fire regime analysis, an accuracy assessment of two MODIS burned area products (i.e. MCD45A1 and WAMIS) was carried out. Having established the appropriate burned area product to use, six elements of the fire regime were considered: fire incidence, extent, seasonality, fire size, frequency and fire return interval. Potential ecological risks to the miombo ecosystem in CSA posed by the fire regime were also highlighted.

Using burned data derived from Landsat TM imagery as reference data, the accuracy of two burned area products offered by MODIS, i.e. MCD45A1 and WAMIS, was tested. It should be noted that ground truthed fire data was not used to determine the true accuracy of the MODIS burned area products; therefore, results given in this study should be interpreted with a level of caution (Siljander 2009). It was observed that although the MCD45A1 and WAMIS are based on two different algorithms, both products bore similar levels of accuracy, detecting approximately 60 to 80% of the fires derived from Landsat (Figure 4.1). However, despite this similarity in accuracy, anomalies revealed in the data derived from

WAMIS in 2008 gives reason to suggest that at certain thresholds, either tree cover or fire intensity, WAMIS fails to detect as many fires as MCD45A1. Further investigation into the impact of tree cover and fire intensity upon the detectability of fires by the two algorithms is therefore recommended. In general, the MCD45A1 was found to be a more reliable source of fire data for reconstructing the fire regime in CSA. MCD45A1 showed to be more consistent in mapping fires across the four years of data evaluated, dependable in terms of access to data, and provided a complete set of data for the whole 10 year period.

Factors that impair the accuracy of the MODIS burned area products vary from fires that are too small (<25 ha) to be detected at a 500m MODIS resolution, to spectral confusion between burned areas and other areas with similar reflectance (e.g. deforested areas), leading to an over-estimation of fires (Roy et al. 2005a, Giglio et al. 2009). Therefore, it is imperative to consider the land use and land cover, as well as weather conditions of an area, when deciding on the use of the MODIS burned area product, as it may not always be applicable. For example, it would not be suitable to use either the MCD45A1 or WAMIS in regions or time periods of persistent cloud cover as the detectability of fires would be greatly reduced. Also, although the MODIS burned area product is unable to map the entire spectrum of fires that occur in a given area, it has been proven to be sufficient in providing a reliable account of general fire patterns in this study as well as other studies across African savannas (Archibald et al. 2010a, Devineau et al. 2010, Rucker & Tiemann 2012).

Based on the fire regime analysis (chapter 5), fires in CSA generally conform to fire patterns described for southern African savannas in various literature (Chidumayo 1993, Frost 1996, WWF 2001). The fire season in CSA spans across the dry season, May to November, with peaks in fire activity between July and September (Figure 5.2). It was also revealed that close to 50% (equivalent to ±87 000 ha) of CSA burned every year from 2001 to 2010, with an average of 132 fires occurring annually (Figure 5.1). Such a high prevalence of fires in protected areas such as CSA signifies their importance in shaping the natural ecosystem and also highlights the need to establish a cost effective system that can adequately monitor the occurrence of these fires.

Although there have been reports of a general increase in veld fires across Zimbabwe since 2002 (Phiri et al. 2011), no trend in the area burned over the 10 year period was found to suggest either a decrease or increase in the amount of area affected by fires in CSA. It is possible that 10 years was too short a time span to determine a trend in the amount of area burned. However, the calculation of effective fire size revealed trends in burned area that were not apparent when burned area was simply measured over time.

Effective fire size, however, revealed that there was shift from larger fires (5000 to 25 000 ha) contributing to the bulk of the total burned area between 2001 and 2005, to a more even contribution of both large and small to medium sized fires (500 to 5000 ha) towards total burned area from 2006 onwards. These results suggest that statistics on burned area should not be taken at face value. Fire is a complex phenomenon that is driven by a myriad of factors, ranging from land use to climate, which may not necessarily alter the overall extent of area burned over time, but will affect other underlying aspects such as fire size.

On the other hand, an increasing trend in fire incidence was noted in CSA. However, the lack of a corresponding increase in the area burned suggests that fires in CSA may not have increased per se, but rather that fires became smaller as the years progressed, and therefore increased in number but not in extent. The observed shift in effective fire size also supports the assumption. This seems to suggest that an increase in fire incidences does not necessarily point towards an increase in fire affected area or vice versa. It also highlights a possible shortcoming in the few studies that have assessed fire incidences in Zimbabwe without measurements on the extent of area burned (e.g. Chigurah & Jerie 2010), as it may be erroneously assumed that an increase or decrease in fire incidences is indicative of a corresponding change in the actual amount of area affected by the fires. The MCD45A1 therefore proves to be an ideal source of information in such cases, as both data on fire incidences and extent of area burned can be extracted from it.

Generally, findings from the fire regime analysis seem to suggest that there are two aspects of the fire regime in CSA that pose a threat to maintaining the natural state of the miombo woodland. Firstly, the concentration of late season fires in the northern parts of CSA (Figure 5.3) is indicative of a potential reduction in the resilience of the dominant tree species against fires, and possibly the consequent eradication of woody biomass in the northern section of CSA. Secondly, 85% of the total area burned in CSA over the period 2001 and 2010 had a FRI of less than 2 years (Figure 5.14). The prevalence of such a short FRI in CSA is a significant cause for concern. Annual and biennial FRI have been noted to cause major alterations in the miombo ecosystem by retarding the growth and regeneration of the dominant tree species, encouraging the invasion of alien tree species, eradication of the dominant tree species and the eventual conversion of woodland to grassland (Trapnell 1959, Gandiwa & Kativu 2009, Masocha et al. 2011, Ryan & Williams 2011). Alterations in the natural diversity and state of the miombo in CSA may have negative implications on the viability of the safari, due to a reduction in game as the habitat is gradually transformed. The livelihoods of surrounding communities may also be affected, particularly for those who earn a living from working in the safari, or other industries (e.g. arts and craft) that depend on tourists that visit the area, as well as those that hunt and gather wild fruits and other edibles from CSA.

The assessment of fire occurrence in Zimbabwe has been largely focused on fire frequency (expressed as the number of fires that occur in a given area), and fewer studies utilize remotely sensed data in their assessments (Nkomo & Sassi 2009, Chigurah & Jerie 2010). However, a full understanding of the spatial and temporal distribution of fires in Zimbabwe requires an assessment of the fire regime in its entirety. Reconstructing the fire regime for CSA in this study shows that the capabilities of the MCD45A1 exceed the mere mapping of fire occurrences but also adds the element of extent to fire assessments. As it has been observed in this study, an isolated assessment of factors such as fire incidence or extent may still overlook underlying trends in what may seem like a stable fire regime. Therefore, through the spatial and temporal nature of the MCD45A1 data set, other elements within a fire regime such as seasonality, fire size, frequency and FRI may also be analysed.

This study has shown that the MCD45A1 has the capacity to provide much needed fire data for the improved management of veld fires in Zimbabwe. The MCD45A1 offers the opportunity to not only fill in the gaps in Zimbabwe`s fire records, but also to carry out an in-depth study of the fire regimes prevailing over Zimbabwe. Compared to current methods of fire collection used in Zimbabwe, the cost and time involved in using the MCD45A1 data set as an alternative source of fire information is relatively low. The data is freely available online and fires that occur in inaccessible areas may also be assessed through the use of the MCD45A1 data set. However, limitations in using the MODIS burned area product in fire assessments do exist. Firstly, the detection of fires on the ground surface is limited by combination of land uses, environmental factors and weather (Roy et al. 2005a, Giglio et al. 2009). Secondly the MCD45A1 data set is only available from 2000 to current day. Therefore, studies of fires prior to 2000 cannot be carried out using this data set (Roy et al. 2008).

6.2 RECOMMENDATIONS

Fires are driven by three factors: climate, fuel load and ignition (Zumbrunnen et al. 2012). A fire regime analysis based on fire data alone describes fire patterns, but data related to the three main drivers of fires explains the patterns noted in the analysis. Therefore a thorough fire regime analysis should be accompanied by additional information such as rainfall data, vegetation cover and land use. This will enable one to determine the influence of human activities and on-site conditions e.g. climate on fire occurrence in a given area (Morgan et al. 2001).

The potential impact of fires on the miombo was primarily based on hypothetical analyses in this study. It is recommended that further studies be carried out to measure the extent to which fires in Zimbabwe may be affecting the miombo and also the consequent ecological implications of changes in the natural state of the miombo.

GIS and remote sensing are powerful tools in not only acquiring information on fires, but also communicating it to those who are responsible for managing fires, yet it is grossly underutilised in Zimbabwe. It is highly recommended that methods used in this study be considered by the Ministry of Natural Resource Management and Environmental Management Agency as a basis for the development of a national fire data base that can be used in the formulation of current and future fire management strategies.

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APPENDICES

APPENDIX A: Conversion of burned area data from raster to vector

Figure A.1 Burned area in raster format

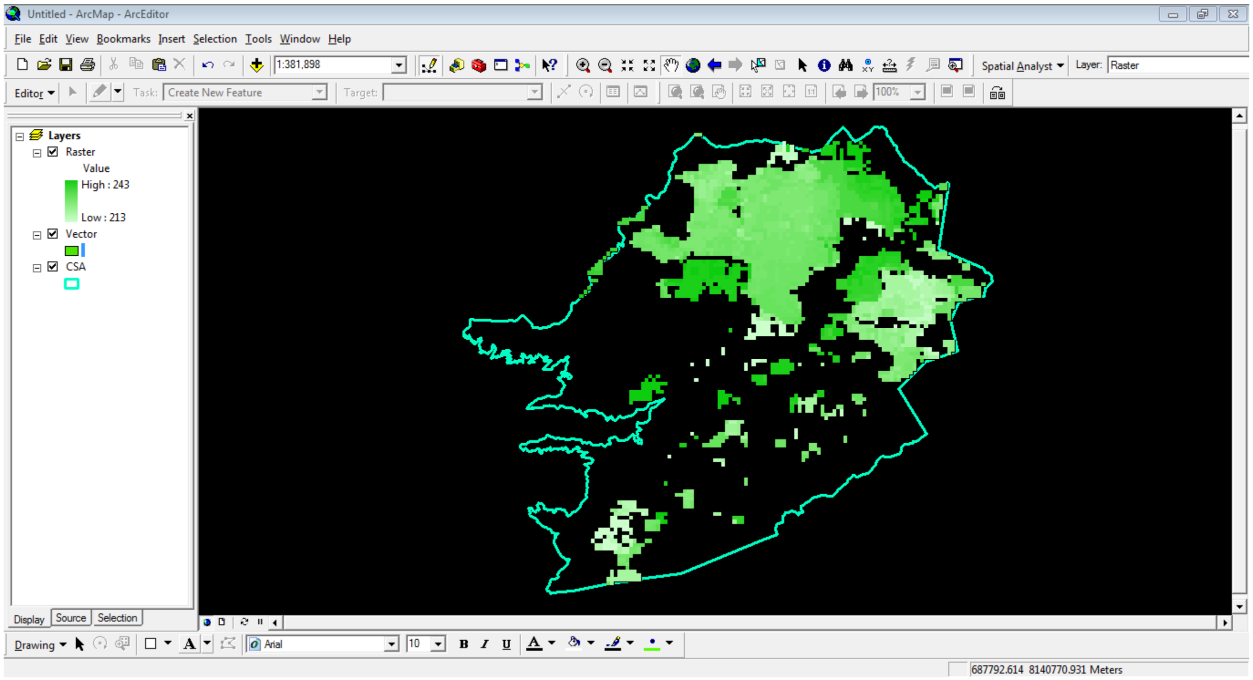
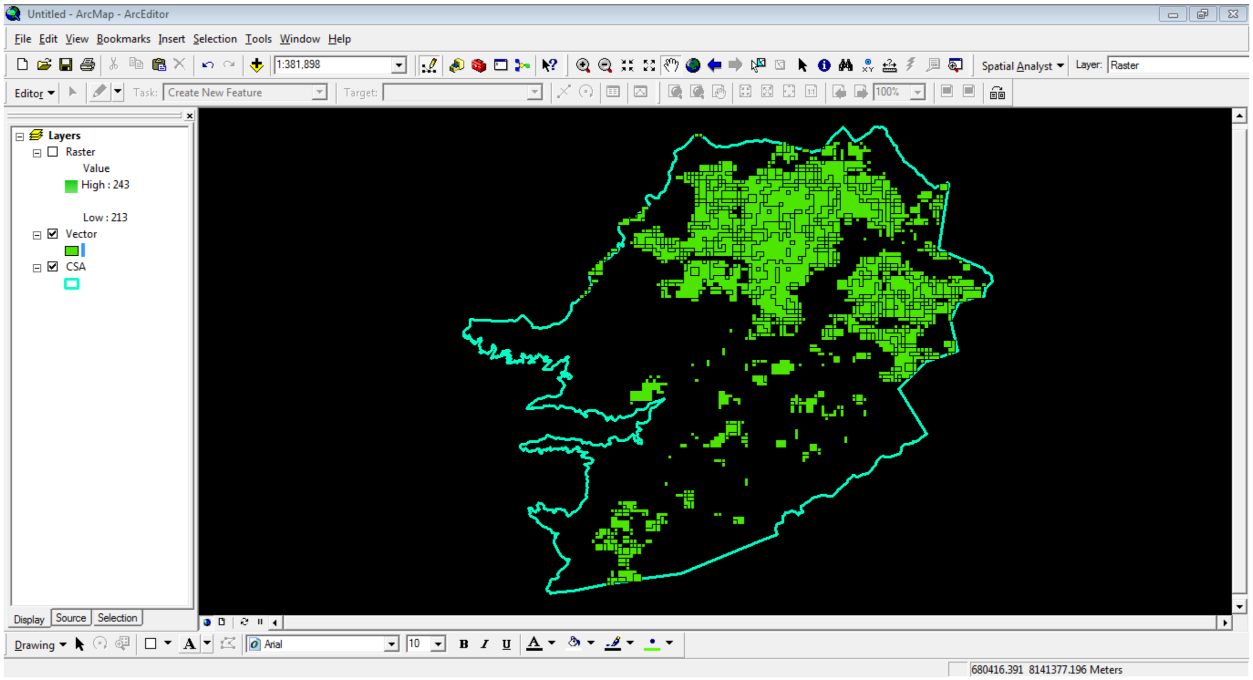
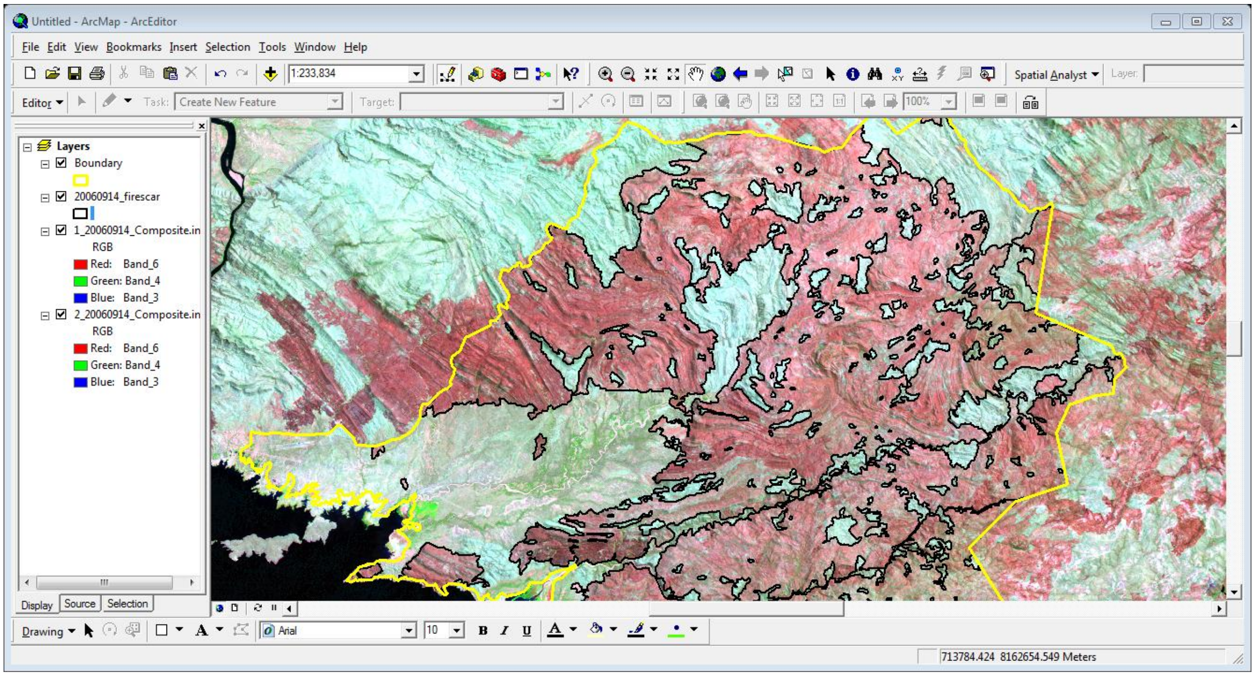


Figure A.2 Burned area in vector format



APPENDIX B: Digitized fire scar from Landsat



APPENDIX C: Data extracted from WAMIS and MCD45A1 product.

		WAMIS		MCD45A1	
		No of fires	**Burned Area (ha)**	**No of fires**	**Burned Area (ha)**
2001	May	2	684.40	2	684.40
	June	-	-	-	-
	July	27	13958.21	22	13363.86
	August	46	53203.30	47	53491.47
	September	22	11670.87	21	11941.03
	October	8	882.52	10	810.48
	November	2	360.21	2	288.17
TOTALS		**107**	**80759.52**	**104**	**80579.41**
		No of fires	**Burned Area (ha)**	**No of fires**	**Burned Area (ha)**
2002	May	12	20550.09	13	19451.44
	June	27	15903.36	26	16407.65
	July	28	28708.89	26	29609.42
	August	12	13850.15	12	14156.33
	September	9	3259.92	10	2935.73
	October	-	-	1	72.04
	November	-		-	-
TOTALS		**88**	**82272.41**	**88**	**82632.62**
		No of fires	**Burned Area (ha)**	**No of fires**	**Burned Area (ha)**
2003	May	2	324.19	2	252.15
	June	18	2197.29	18	2341.38
	July	27	21162.45	22	21000.36
	August	33	38740.79	25	38092.41
	September	26	21828.84	27	21954.92
	October	12	5241.08	13	5691.35
	November	1	72.04	1	108.06
TOTALS		**119**	**89566.70**	**108**	**89440.62**
		WAMIS		MCD45A1	
		No of fires	**Burned Area (ha)**	**No of fires**	**Burned Area (ha)**
2004	May	3	306.18	3	432.25
	June	24	8248.85	22	7996.70
	July	32	21252.50	40	21558.68
	August	27	54229.91	29	53905.72
	September	24	4538.67	26	4016.36
	October	5	1548.91	6	1458.86
	November	-	-	-	-
TOTALS		**115**	**90125.03**	**126**	**89368.58**

		WAMIS		MCD45A1	
		No of fires	**Burned Area (ha)**	**No of fires**	**Burned Area (ha)**
2005	May	28	32130.90	25	31716.66
	June	30	15291.00	29	14588.58
	July	19	7438.38	19	7222.25
	August	27	14138.32	25	13742.09
	September	18	13039.67	19	13345.85
	October	9	5205.06	11	5133.02
	November	3	252.15	4	378.22
TOTALS		**134**	**87495.48**	**132**	**86126.67**
		No of fires	**Burned Area (ha)**	**No of fires**	**Burned Area (ha)**
2006	May	5	738.43	5	936.55
	June	20	9833.79	24	10157.98
	July	43	31698.65	43	31860.75
	August	43	16245.56	47	16911.95
	September	32	34544.32	34	34436.26
	October	7	1927.13	8	1963.16
	November	1	180.11	2	144.08
TOTALS		**151**	**95167.99**	**163**	**96410.72**
		No of fires	**Burned Area (ha)**	**No of fires**	**Burned Area (ha)**
2007	May	16	9833.79	21	9635.67
	June	31	30636.03	29	30383.88
	July	32	15399.06	31	15615.19
	August	28	10103.94	27	10770.34
	September	16	26547.62	16	26637.67
	October	7	1044.61	8	936.55
	November	-	-	-	-
TOTALS		**130**	**93565.05**	**132**	**93979.29**
		No of fires	**Burned Area (ha)**	**No of fires**	**Burned Area (ha)**
2008	May	14	10230.02	19	11328.67
	June	13	8428.96	43	16569.75
	July	26	15777.28	40	23665.92
	August	18	26565.63	38	38164.45
	September	18	5006.95	35	7402.36
	October	4	2035.20	13	3476.05
	November	1	54.03	1	270.16
TOTALS		**94**	**68098.07**	**189**	**100877.35**

		WAMIS		MCD45A1	
		No of fires	**Burned Area (ha)**	**No of fires**	**Burned Area (ha)**
2009	May	-	-	-	-
	June	16	6880.05	26	11886.99
	July	18	7870.63	37	13706.06
	August	24	33427.67	37	34670.40
	September	12	14588.58	38	19865.69
	October	-	-	3	54.03
	November	-	-		-
TOTALS		**70**	**62766.93**	**141**	**80183.18**
		WAMIS		MCD45A1	
		No of fires	**Burned Area (ha)**	**No of fires**	**Burned Area (ha)**
2010	May	-	-	10	1620.95
	June	-	-	34	12607.42
	July	-	-	40	19199.30
	August	-	-	28	20532.08
	September	12	13706.06	16	17488.29
	October	9	2953.74	11	4304.53
	November	-	-	3	918.54
TOTALS		**21**	**16659.80**	**142**	**76671.11**

APPENDIX D: A Flow diagram of the confusion matrix.

Modis Burned Area.shp

Study Area Boundary

Digitised Landsat Burned Area.shp

Append

Append

Append LBA.shp& MBA.shp to study area boundary (separately)

- Data Mngmt/General/Append
- Schema type: No Test
- Open Editor/Edit Attr table for the new.shp: 0 – Unburned and 1 - Burned

New MBA.shp
Where, Value 0 = Unburned and 1 = Burned

New LBA.shp
Where, Value 0 = Unburned and 1 = Burned

Rasterize

Rasterize

Convert New MBAshp & New LBA.shp to Raster

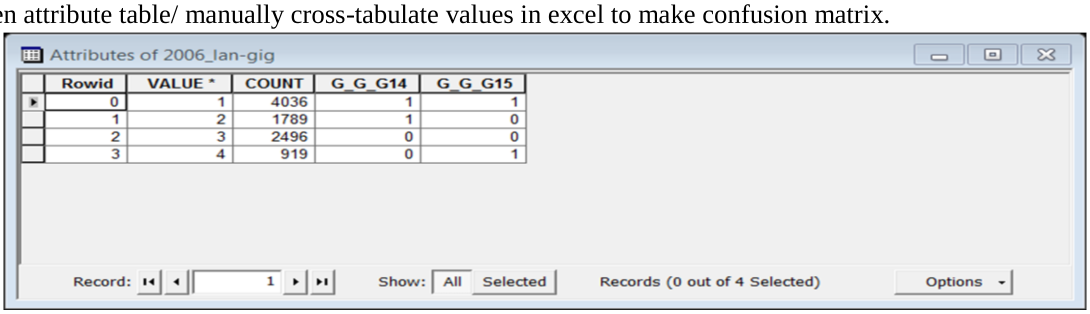
- Conversion tools/To raster/Polygon to raster
- Value field: FID
- Priority field: ID
- Pixel size: 424,388
- For MBA.shp go to Environments/General settings/Snap raster – LBA.grid

MBA.grid
- Where, Value 0 = Unburned and 1 = Burned

LBA.grid
- Where, Value 0 = Unburned and 1 = Burned

To get count of matches and mismatched of Burned and Unburned pixels as input to the xls confusion matrix:

-Spatial Analyst Tool box/Math algebra
-Enter equation: CombinatorialXOr ("LBA.grid", "MBA.grid") NB: enter reference grid first ie: LBA.grid
-New calculation: Right click/Data/Make permanent
-Open attribute table/ manually cross-tabulate values in excel to make confusion matrix.


Attributes of 2006_lan-gig

Rowid	VALUE *	COUNT	G_G_G14	G_G_G15
0	1	4036	1	1
1	2	1789	1	0
2	3	2496	0	0
3	4	919	0	1

Record: 1 Show: All Selected Records (0 out of 4 Selected) Options