



An investigation into the fire regimes of the upper Tsitsa River catchment

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Abstract

South African grasslands are rich in flora and fauna and are a dominant vegetation cover in many of the water catchment areas, while providing a multitude of services to local communities and the environment. Fires have been the primary tool used to manage grasslands for livestock production for many years. However, there is debate about how they impact and alter landscapes and there are two schools of thought throughout literature. One argument is that fires are detrimental to landscapes; altering hydrological processes by causing excess soil erosion and changing soil properties. The other opinion is that fires are beneficial to ecosystems; maintaining vegetation structure, preventing bush encroachment, cycling nutrients and allowing for new plant growth. This study focused on the effects of fire regimes on soil properties in order to better understand the role that fire plays in geomorphic processes in the upper Tsitsa River catchment. The catchment falls under two land management types-commercial and communal, with the latter being severely degraded. Fire regimes were determined using a combination of Landsat and MODIS remotely sensed data and testing was carried out on soils exposed to different Fire Return Intervals (FRI). It was found that soils that were exposed to a high fire frequency (1-2-year FRI) exhibited a significantly higher degree of water repellency ($p < 0.001$) and surface hardness ($p < 0.001$) than soils that were exposed to a low fire frequency (3-4 year FRI), which indicates increased erosion potential. However, whilst a higher fire frequency resulted in altered soil characteristics, it contributed to landscape degradation as suggested in the literature, and soils are able to return to their previous state in over a short period of time. Further investigation into factors affecting fire regimes found that geology influenced vegetation type, resulting in differences in biomass in the two land management areas. The commercially managed land supported a high biomass, which resulted in a more natural fire regime with frequent fires, whilst the communally managed land supported less biomass. Smaller fuel loads and increased landscape fragmentation through overgrazing and road networks resulted in an altered fire regime with less frequent fires. Although the communally managed land has an altered fire regime, this research suggests that fires are not influencing the severe degradation present in these areas, and this is rather a function of geology and mismanagement of land. Whilst fire and grazing management plans can be implemented to alter the modified fire regime in the communal areas back to its natural state, this would take a long time and an increase in fires would pose a threat to surrounding communities.

Overall the fire regimes in the upper Tsitsa River catchment are being altered by human influence and land management type, and whilst fire frequency is negatively altering soil properties, these soils can return to their natural state. The severe land degradation present in the communally managed areas are not a function of fire frequency and rather of geology and land management.

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List of Abbreviations

UTRC	Upper Tsitsa River Catchment
NVFFA	National Veld Fire and Forestry Act
MAP	Mean Annual Precipitation
SANBI	South African National Biodiversity Institute
WERT	Watershed Emergency Response Team
SWSA	Strategic Water Source Area
WRC	Water Research Council
NLEIP	Ntabelanga Lalini Ecological Infrastructure Project
FAO	Food and Agriculture Organization
GIS	Geographic Information System
WDPT	Water Drop Penetration Time
ANOVA	Analysis of Variance
DPM	Disk Pasture Meter
LSD	Least Significant Difference
PCA	Principal Component Analysis
SOM	Soil Organic Matter
NDVI	Normalized Difference Vegetation Index
EOS	Earth Observation System
NASA	National Aeronautics and Space Administration
VIIRS	Visible Infrared Imaging Radiometer Suite
MODIS	Moderate Resolution Imaging Spectroradiometer
FIRMS	Fire Information for Resource Management System
AFIS	Advanced Fire Information System
SST	Sea Surface Temperature
FRP	Fire Radiative Power

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1. Introduction

Fires are a largely misunderstood global phenomenon that have maintained and shaped the Earth's landscapes and atmosphere for millions of years (Archibald *et al.*, 2013; Doerr & Santin, 2016). They are an essential component in many ecosystems, maintaining landscape functions and biodiversity (Bond, 1997; O'Connor & Bredenkamp, 1997), preventing bush encroachment of grasslands and allowing for resprouting and reseedling, cycling nutrients and allowing new vegetative growth (Mucina & Rutherford, 2006; Kraaij *et al.*, 2011). The negative effects of fires are the removal of vegetative cover, exposing and altering soil properties which may lead to increased erosion rates as well as posing a threat to man-made infrastructure and crops (DeBano *et al.*, 1979; Vieira *et al.*, 2015). Our global view of fires is however restricted to the extent of burning detected by satellites. This limits our knowledge and views as fire is multidimensional and to focus on a single component denies its complexity (Archibald *et al.*, 2013).

Fires, although a natural feature in many landscapes have been identified as some of the biggest natural disasters in the second decade of the 21st Century, having significant effects on biodiversity, food, agriculture, ecosystem services and other human-related resources (San-Miguel-Ayanz *et al.*, 2018; FAO, 2019). Some of the most extreme disasters include the California Camp Fire, which burned from 8-25 November 2018, covering a total area of approximately 600 km² and destroying over 18 000 residences, commercial and other structures (WERT, 2018). Similarly, in South Africa, the Knysna fires in 2017 burned close to 200 km² of plantations and destroyed over 2000 buildings between Plettenberg Bay and Knysna, resulting in mass destruction and loss of life (Stehle, 2017). The apparent contradiction between the necessity of fires and the threats they pose assets is mainly studied on a global level, despite the known local importance. Whilst fires are key for ecosystem and biodiversity maintenance, when combined with improper land management practices they result in increased erosion and altered hydrological processes that lead to land degradation. To address this problem, key aspects of fires have been identified to further understand how fire regimes are associated with climate, vegetation and humans (Archibald, 2016).

Fires play an important role in grassland management throughout South Africa. They are regularly used to burn fire breaks and clear moribund grazing lands (Ratsele, 2013). All landscapes have natural fire return periods and naturally one should not be overly concerned with high fire frequencies in grasslands as they have adapted and evolved to be fire resistant (Vogl, 1979). However, human influence has altered the natural fire regimes and more needs to be understood about how this may be affecting various grassland ecosystems (Whisenant, 1989 ; Brotons *et al.*, 2013; Doerr & Santin, 2016). There is a lack of knowledge around fire regimes and the effects of fires on soil and vegetation in degraded and rehabilitated catchments within South Africa. The aim of this study is to provide insight into the fire

regimes and patterns in the upper Tsitsa River catchment (UTRC) and how that is affecting the surrounding landscape, which many commercial farmers and rural communities rely on.

1.1 Background and motivation

Fires are a fundamental tool used in land management in South Africa, however, the effects and implications of fires on soils and vegetation are controversial. Soil erosion is prevalent in South Africa, with social, environmental and economic implications (Parwada & Van Tol, 2016). Wildfires can alter hydrological, geomorphological and ecological functioning of hillslopes by removing ground cover and exposing soils to erosive rainfall post-fire. Fires can also reduce infiltration by enhancing soil water repellency, leading to high erosion rates and the loss of valuable topsoil (Martin & Moody, 2001; Moody *et al.*, 2009). Soil erosion potential is at its peak immediately after a fire disturbance as there is minimal vegetation to slow the movement of water and sediment and whilst there are multiple variables that influence runoff and soil erosion, this is arguably the contributing factor to erosion from fires. Many studies suggest that fire severity, rainfall intensity and reduced soil infiltration are the biggest factors affecting post-fire runoff and erosion rates (Inbar *et al.*, 1998; De Luis *et al.*, 2003; Wohlgemuth *et al.*, 2009; García-Comendador *et al.*, 2017). However, land management practices such as ploughing, terracing and logging tend to exacerbate the problem as they often involve heavy machinery and can displace topsoil, resulting in greater hydrological and erosive responses than those caused by the fire (Vieira *et al.*, 2016). Research has been able to pinpoint several key sources of fine-grained sediment that is reaching rivers and being actively transported. Topsoil from agricultural fields, overgrazed grasslands, areas disturbed by forestry and urban areas or villages are the main sources of sediment found in channels and reservoirs (Owens *et al.*, 2005). The Eastern Cape has a high erosion potential that can impact infrastructure and agriculture (Cottage & Tau, 2013; Le Roux, 2018). The Mzimbuvu River catchment, which is part of the Eastern Cape, has been defined as a Strategic Water Source Area (SWSA) by the Water Research Commission (WRC) and is currently undergoing a series of rehabilitation programs and water resource developments as part of social and economic upliftment in the catchment (Le Maitre *et al.*, 2018; Le Roux, 2018).

There are two proposed developments in the form of large dams, namely the Ntabelanga and Laleni Dams which will be multipurpose, supplying water for domestic use as well as agricultural irrigation and industrial use. The Laleni dam will have a storage capacity of 232 million m³ whilst the Ntabelanga dam will have a storage capacity of 490 million m³, as well as being fitted with a small hydroelectric power plant for the generation of electricity for the benefit of the surrounding communities (Muruvu & Calmeyer, 2015; Department of Environmental Affairs, 2016; Le Roux, 2018). For the dams to be filled with good quality water and avoid excess sedimentation, it is essential that the catchment area upstream is managed correctly. Unfortunately, the Tsitsa River catchment is known for rural poverty and land degradation and the Ntabelanga and Laleni Dams are predicted to silt up prematurely if land degradation in the catchments feeding them continues (Le Roux, 2018). Restoration work, in

conjunction with research in the Ntabelanga Lalini Ecological Infrastructure Project (NLEIP), recently renamed The Tsitsa Project, hopes to provide sustainable development of natural resources and land management for the surrounding communities (Department of Environmental Affairs, 2016). The Tsitsa catchment has been identified as an area that has high levels of erosion and gullying and the aim of the Tsitsa project is to research, rehabilitate and restore many of the affected areas according to guidelines, such as by the South African Biodiversity Institute (SANBI) (Cumming *et al.*, 2014).

Much of the Tsitsa River catchment is covered by grassland with various land uses including forestry, commercial and subsistence livestock farming. Fires are used by different entities to manage land and this research will assess the fire regimes and impacts of fires on soils within the catchment. It is often mentioned in literature that current land degradation and erosion is a function of fire frequency, however, this may not currently be the case in the upper Tsitsa catchment and needs to be further investigated (Doerr & Santin, 2016). Many perceive fires as an ever-growing threat to landscapes, ecosystems and livelihoods, with increasing frequency, size and intensity. However, there is evidence that suggests that these are misplaced perceptions as fires are essential to maintain specific ecosystem functions (Doerr & Santin, 2016). The frequency of fires and the effects on soils in both frequently and infrequently burned areas will be assessed in order to determine the potential effects fires are having on the Tsitsa catchment landscape. Although this project is focussed around determining the fire regimes and patterns of fires within the catchment, the results could potentially be used to inform land users on how to best utilise fires and the effects of fires if not managed correctly.

1.2 Thesis structure

The structure of this dissertation is as follows:

Research questions, aims and objectives are described in section 1.3, whilst an in-depth overview of the study area (upper Tsitsa River catchment) are described in section 1.4. Literature is reviewed in chapter two which describes the global view on fires and the key characteristics of fire regimes. Further literature was reviewed to investigate the role fires play in a landscape and how they affect soil properties and vegetation. Lastly, was reviewed on remote sensing as this links to the first objective (What is the best way to detect and define fire regimes in the study area using GIS and remote sensing). The methods are laid out in chapter 3 and describe the processes used to capture and measure fires via remote sensing and satellite imagery. Secondly, field methods describe what and how measurements were taken, with laboratory tests explained in depth. The 4th chapter presents the results found during this study and includes the definition of key fire characteristics in the study area. Statistical analyses are also presented on the impacts of fire frequency on soil properties. Due to the complex nature of fires, chapter 5 compares findings from this study to others, and synthesizes the results, bringing them into context. Chapter 6 concludes the work done during this research and links it to the original objectives and research questions. Limitations of the study and future research is also covered in the final chapter.

1.3 Aim, research questions and objectives

1.3.1 Aim

The aim of this research is to map and describe the fire regimes within the upper Tsitsa River catchment to understand the burning patterns of grasslands and assess the impacts of the recent fire regimes on soil properties and biomass.

1.3.2 Research questions

- What is the best way to detect and define fire regimes in the study area using GIS and remote sensing?
- Are there discernible patterns in the fire regimes of the study area in relation to grassland management for commercial forestry, commercial livestock and communal livestock farming?
- How does fire frequency affect soil properties in the UTRC?

1.3.3 Objectives

- Establish an effective methodology for detecting and analysing fire regimes in the Tsitsa River catchment.
- Determine fire regimes within the Tsitsa River catchment by observing fire frequency, extent, seasonality, size and intensity for commercial forestry, commercial livestock and communal livestock grassland management.
- Determine the effects of fire frequency on soil characteristics.

1.4 Study area

The study area (UTRC; quaternary catchment T35A-E) is located in the northern Eastern Cape, South Africa (Figure 1.1). It is part of the greater Tsitsa River catchment, which covers ~4000 km². This forms part of the Mzimbuvu catchment that comprises of commercial and communal land of the former Transkei and drains eastward towards the Indian Ocean (Bannatyne *et al.*, 2017). The UTRC lies approximately 100 km North of Mthatha on the south-eastern slopes of the Great Escarpment. The upper catchment covers ~2000 km² of land where the Mooi, the Pot, the Tsitsana, the Hlankomo and the Gqukunqa rivers feed into the Tsitsa River (Figure 1.1). The catchment is characterised by grassland plateaus and rolling hills, with steep escarpments near the headwaters (Bäse *et al.*, 2006; Bannatyne *et al.*, 2017).

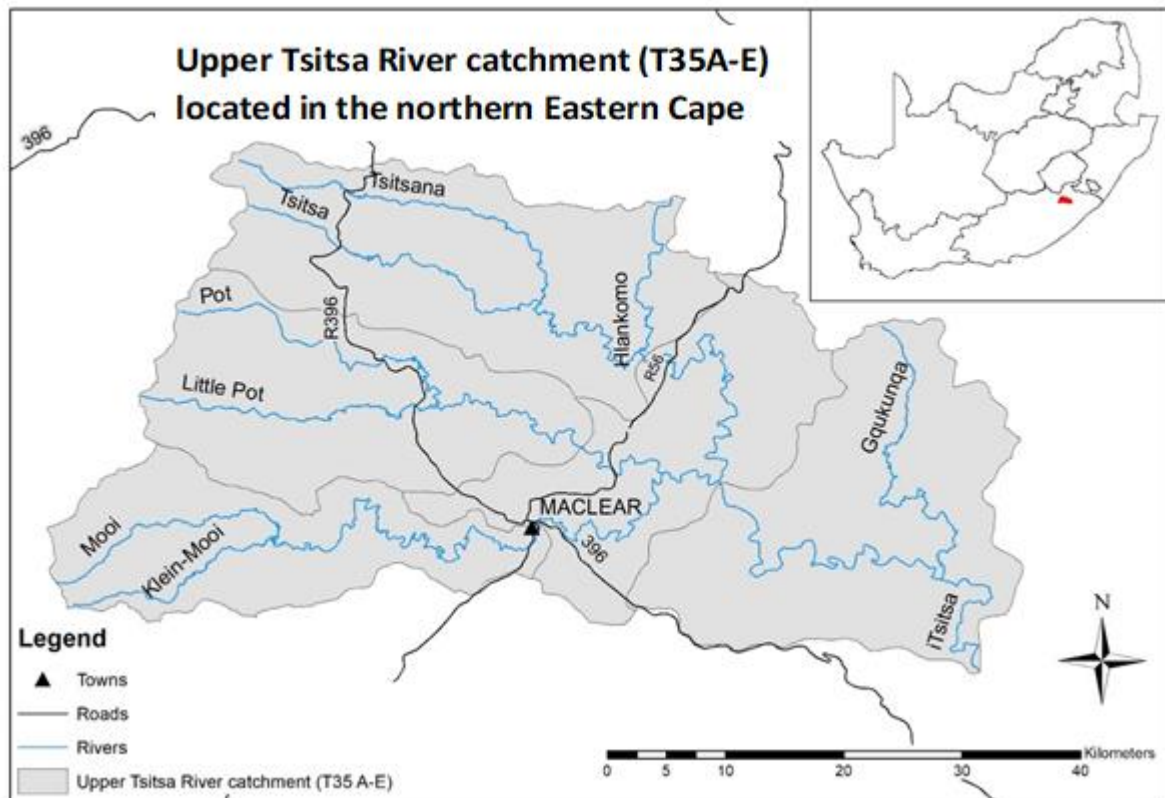


Figure 1.1: The upper Tsitsa River catchment located in the northern Eastern Cape, South Africa.

Land use in the UTRC is dominated by commercial plantations, rural subsistence farming and larger commercial farms, with smaller urban settlements scattered throughout. Many communities found within the former Transkei homelands on the western side of the catchment experience poor infrastructure and employment opportunities. Communities and livelihoods are to a large extent, reliant on natural resources and subsistence farming (livestock and crops) for a living (Porter & Phillips-Howard, 1997; Kakembo & Rowntree, 2003; van Tol *et al.*, 2014). Human impacts on the landscape have resulted in increased levels of erosion and sediment build-up in the surrounding rivers and streams. This is a result of mismanagement of cultivated land and unsuitable grazing and burning practices, severely affecting the local water sources (Porter and Phillips-Howard, 1997; Bannatyne *et al.*, 2017). Improper land-use practices such as overgrazing and frequent or recurrent burning have exacerbated soil loss and result in high land degradation and erosion rates (van der Waal, 2015).

1.4.1 Climate

The UTRC ranges in altitude from approximately 870 m above sea level just below the proposed Ntabelanga dam to 2700 m above sea level at the top of the Drakensberg Escarpment (Bäse *et al.*, 2006; Moore, 2016). The UTRC has a temperate subtropical climate with high-intensity summer rainfall events (thunderstorms). During these thunderstorms, lightning is a common occurrence, being the primary natural ignition source for grassland fires (Mucina & Rutherford, 2006). Globally, grasslands experience a high frequency of lightning strikes. However, in the study area there are very few extreme

weather events and a low chance of lightning strikes during the winter/ dry period when fires are most common (Mucina & Rutherford, 2006; Archibald, 2010; van Tol *et al.*, 2016). The relatively high elevation of the catchment tends to influence the vegetation and climate patterns. An increase in altitude on the Escarpment results in a decrease in temperature and an increase in rainfall during the wet, summer season. This is because mountains have a high prevalence of orographic rainfall, forcing moist air to rise forming rain clouds. This results in areas on the higher-lying escarpment experiencing increased rainfall and mist from November through to January (Mucina and Rutherford, 2006). Most of the annual rainfall (approximately 75%) occurs between November and March, with little rain falling during the winter period (Bäse *et al.*, 2006). Maclear, the largest settlement in the catchment receives a Mean Annual Precipitation (MAP) of approximately 750 mm, with a low of 15 mm in July and a high of 135 in January (Figure 1.2) (Parwada & Van Tol, 2017). Annual rainfall varies from approximately 600 mm per annum to over 1100 per annum. Temperatures range from just below 0°C in July to up to 30°C in January, and in combination with rainfall, results in warm moist conditions during the summer months and cooler dry conditions during winter. During the coolest period of the year, frost is a common phenomenon throughout the catchment generally occurring between June and August, which coincides with the fire season (Mucina and Rutherford, 2006). Higher lying areas receive frost earlier than lower lying areas, drying and curing grasses, increasing their susceptibility to burning (SANBI, 2013). On average, the grasslands in the UTRC experience 50-60 Mean Frost Days (MFD) per year (calculated by the temperature reaching below 0°C) (Mucina & Rutherford, 2006).

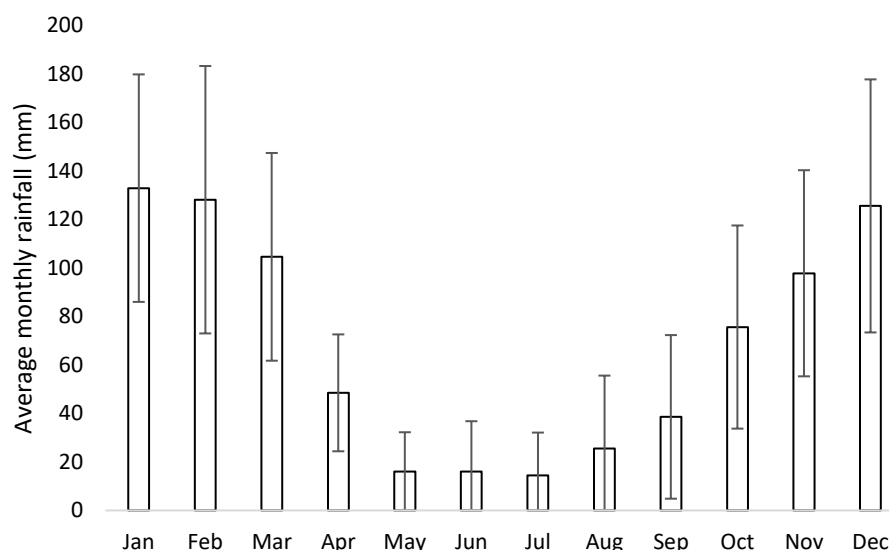


Figure 1.2: Maclear average monthly rainfall (mm) from 1978-2012 (Adapted from Moore, 2016).

1.4.2 Geology and soils

The geology of the study area is dominated by Stormberg sandstones, forming the Molteno, Elliot and Clarens Formations which belong to the Karoo Supergroup, characterised by layers of sandstone and

mudstone as well as lenses of siltstone and carbonaceous shales. Within the Karoo sequence, intrusions of dolerite dykes are present throughout the catchment (Table 1.1; Figure 1.3) (Bäse *et al.*, 2006).

Soil properties and characteristics in the UTRC vary considerably due to the diverse geology. Most soils are relatively shallow loams, ranging between 30 cm and 50 cm in depth. Deeper, sandy loams are less common but still of significance and range between 50 cm and 120 cm (Le Roux *et al.*, 2015). Soils derived from the Tarkastad, Molteno and Elliott formations are found to be highly erodible as they are dispersive and commonly form duplex soils. A duplex soil has a sharp distinction between the topsoil and subsoil layers due to the high clay content found within the subsoil layers due to leaching (Le Roux *et al.*, 2015). Duplex soils are prone to erosion and gullyng due to the abrupt change in texture between the horizons in the soil profile. Coarse, sandy loam topsoil is eroded whilst the heavier, finer-textured clay subsoils remain. It was also found that subsoils erode through saturation, dispersing and causing subsequent tunnelling (van Tol *et al.*, 2014).

There is a high degree of erosion and gullyng within the Tsitsa River catchment. High rainfall intensities and steep slopes, the erodible nature of the soils, poor land management practices and use of fires to maintain grasslands are thought to be the main causes of erosion in the catchment (van Tol *et al.*, 2014; Le Roux *et al.*, 2014). Loss of vegetation cover and root stability due to overgrazing, fires and over-cultivation also leads to increased sheet and rill erosion. A soils ability to resist erosion is determined by its resistance to dispersion (Laker, 2004). The two most important factors in soil stability are organic matter content and the presence of iron and aluminium oxides, and this is largely controlled by the physical and chemical properties of the soils which fires are known to affect (Laker, 2004).

Table 1.1: Geological formations within the UTRC from the escarpment to the Tsitsa River in the lower catchment. (Adapted from: Norman & Whitfield, 2006).

Geological features and formations of the Stormberg, Drakensberg and Beaufort Groups belonging to the Karoo Supergroup

Age	Group		Thickness	Dominant rock types	Susceptibility to erosion
183 Ma	Drakensberg group		>2000 m	Basaltic lava flows from Karoo dolerite dykes	Low
198 Ma	Clarens Formation	Stormberg Group	<300 m	Fine grained sandstone and siltstone	Intermediate
215 Ma	Elliot Formation		< 500 m	Mudstone with interbedded sandstones	High due to dispersive soils
240 Ma	Molteno Formation		< 600 m	Alternating sandstone, mudstone and shale	High due to dispersive soils
250 Ma	Beaufort Group		< 7000 m	Mudstone and sandstones	Very high due to dispersive soils

*Please note that Table 1.1 and Figure 1.3 are colour coordinated.

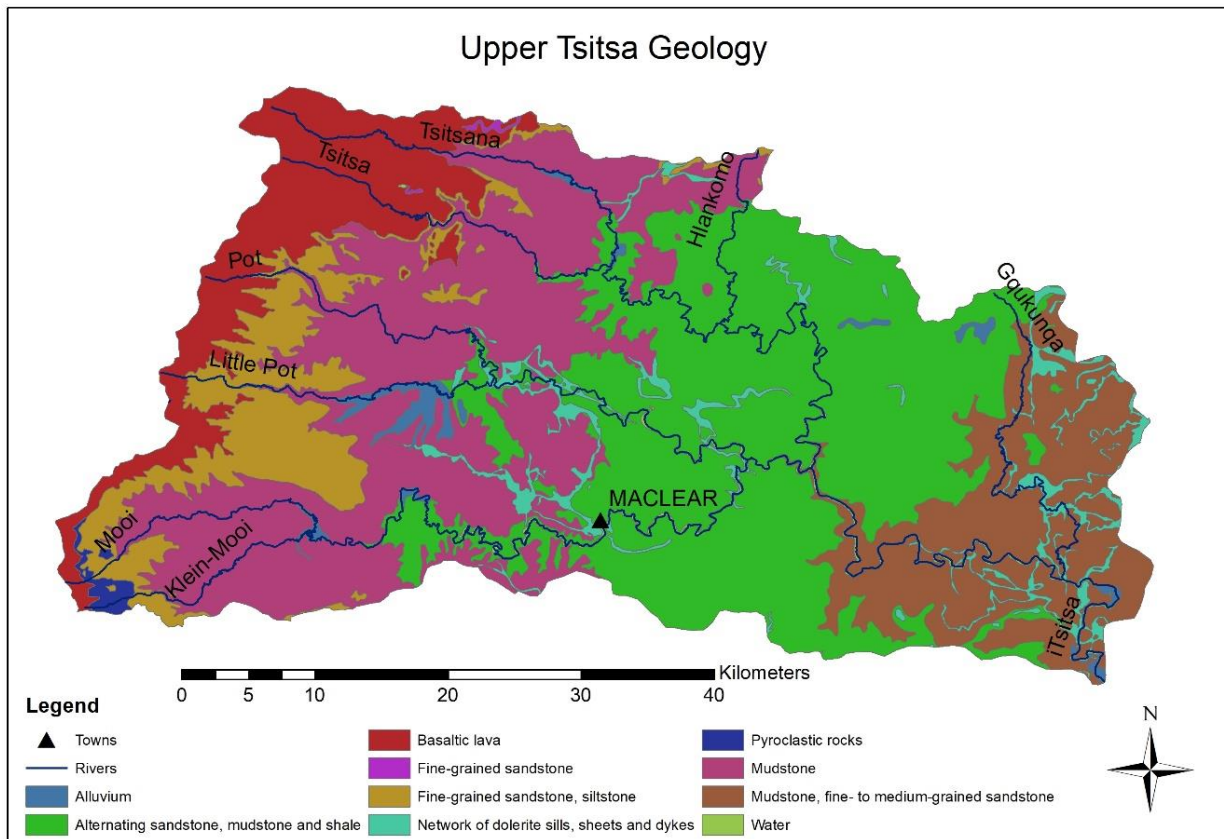


Figure 1.3: Geological map of the showing alternating mudstones and sandstones dominating the throughout the UTRC.

1.4.3 Vegetation

The upper Tsitsa catchment is dominated by grasslands comprising of an intricate system of patches and corridors of highland and sub-escarpment grasslands. Several natural grassland types dominate the UTRC, namely the Drakensberg Foothill Moist Grassland, the East Griqualand Grassland, the Southern Drakensberg Highland Grassland and the Lesotho Highland Basalt Grassland on the western borders of the catchment (Figure 1.4). Smaller patches of Southern Mist belt forest, Eastern Valley Bushveld, Mthatha Moist Grassland and Eastern Temperate Freshwater Wetlands can be found scattered throughout (Mucina & Rutherford, 2006). A high degree of the East Griqualand Grassland and Drakensberg Foothill Moist Grassland in the communal areas are secondary grasslands, as large plots of abandoned agricultural land have been allowed to shift back to a grassland state. Although secondary grasslands may look like primary grasslands, they differ in species composition and structure, ecological functioning and ecosystem services delivered (Mucina & Rutherford, 2006; SANBI, 2013). Moisture availability is a key climatic factor which can be used to separate grasslands into two different groups; Moist grasslands (Southern Drakensberg Highland Grassland, Lesotho Highland Basalt

Grassland and parts of the East Griqualand Grassland) receive more than 650 mm rainfall annually and are characterised by sour, less palatable grasses (to livestock), dystrophic soils, high plant production, and a high fire frequency. Dry grasslands (Parts of the East Griqualand Grassland and parts of the Drakensberg foothill Moist Grassland) receive less annual rainfall than the moist grasslands and are dominated by sweet, palatable grasses with eutrophic soils and a much lower plant production and fire frequency (Mucina & Rutherford, 2006). This makes moist grasslands fire dependant and dry grasslands climate dependant (Kruger *et al.*, 2006). High tannin contents of the grasses in the upper Tsitsa result in a slow rate of decomposition, therefore allowing a build-up of highly flammable fuel. Some woody species are found in grasslands and these include small trees such as *Acacias* and certain species of *Protea*, that have thick, fire-resistant bark which allows them to tolerate low intensity, high-frequency fires (Mucina & Rutherford, 2006). Fire is a vital aspect of grassland ecosystems and is essential to the maintenance of structural and textural patterns. Man-made and natural fires maintain grassland dominance over woody species, preventing bush encroachment and serving as an important ecological component within the grassland ecosystems (Kruger *et al.*, 2006; Mucina & Rutherford, 2006).

The immediate effects of fires on vegetation depend on fire intensity, however, long-term effects depend on other variables such as fire frequency, size and season (Gill, 1975). Typically, grassland fire return intervals (FRIs) range between 1-4 years and fires most often occur during late winter (July to

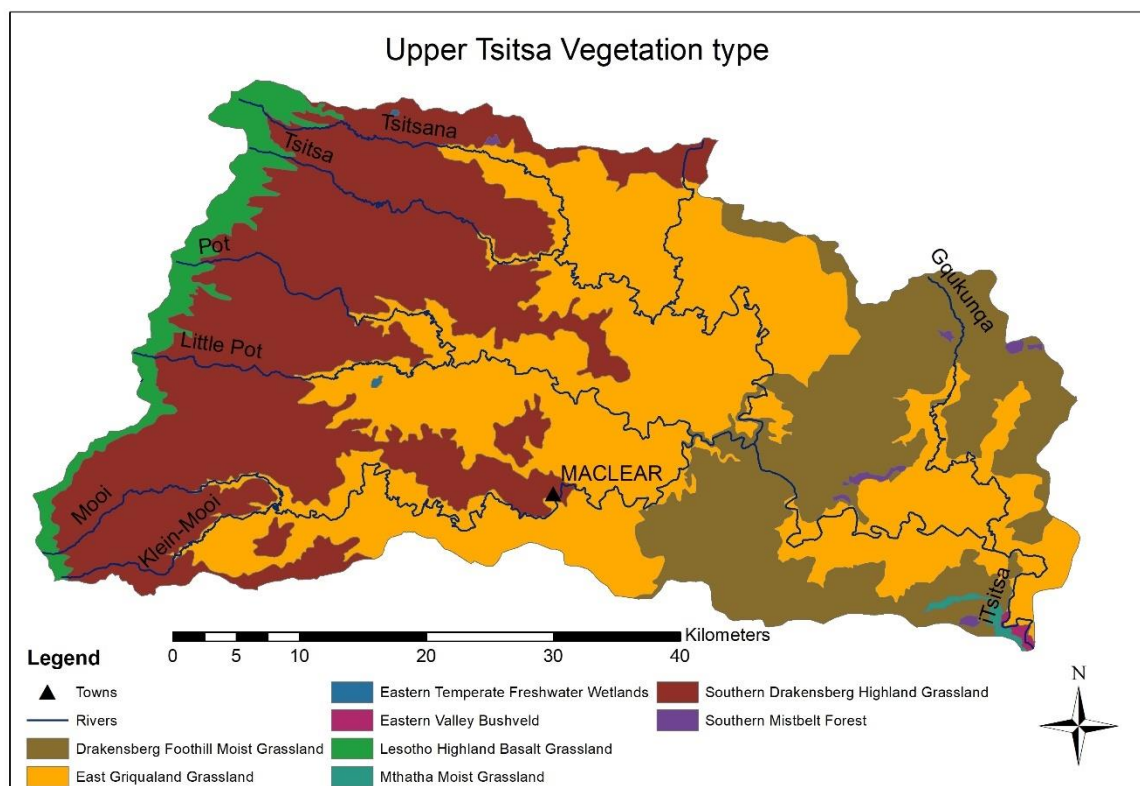


Figure 1.4: Vegetation types found in the UTRC (SANBI).

September) (Mucina & Rutherford, 2006). It is common to come across landscapes composed of grassland patches in various stages of recovery after fires as natural breaks and man-made structures act as barriers, preventing large scale fires. As mentioned above, lightning strikes are the primary natural cause. However, using fires as a land management tool is common practice in the UTRC.

The UTRC is split into two land management groups (commercial and communal), and the vegetation cover differs between the two. As an example, Normalized Difference Vegetation Index (NDVI) images (Figure 1.5) from different times of the year (January, April and October) are used to compare the two different land management types. The images focus on grasslands from two river catchments within the UTRC, namely the Little Pot River and the Gqukunka River which feed into the Tsitsa River and ultimately the greater Mzimbuvu River. The NDVI indicates whether the grasslands contain live, actively growing, vegetation or not. There is an observable difference between the two land management types, with the commercially managed land containing considerably greener, healthier vegetation during and shortly after the wet season than the communally managed land. Although most of the grassland types in the UTRC are classified as moist, rainfall is variable throughout the catchment and in general there are moist, sour grasslands in the commercially managed areas and dry, sweet grasslands in the communal areas. Higher moisture availability in the commercial areas results in high fuel loads during the dry winter periods (June-October), and less so in the communal areas. The low NDVI values within the communal area indicate a high degree of fragmentation and bare ground within the grasslands and lack of vegetation or dead (brown) vegetation (Wessels *et al.*, 2004; Meneses-Tovar, 2011). Herd-Hoare, 2019 found a positive relationship between NDVI and biomass, which meant that higher NDVI values represented higher above-ground biomass. In this case the commercially managed areas have a higher biomass cover and less fragmentation than the communally managed areas.

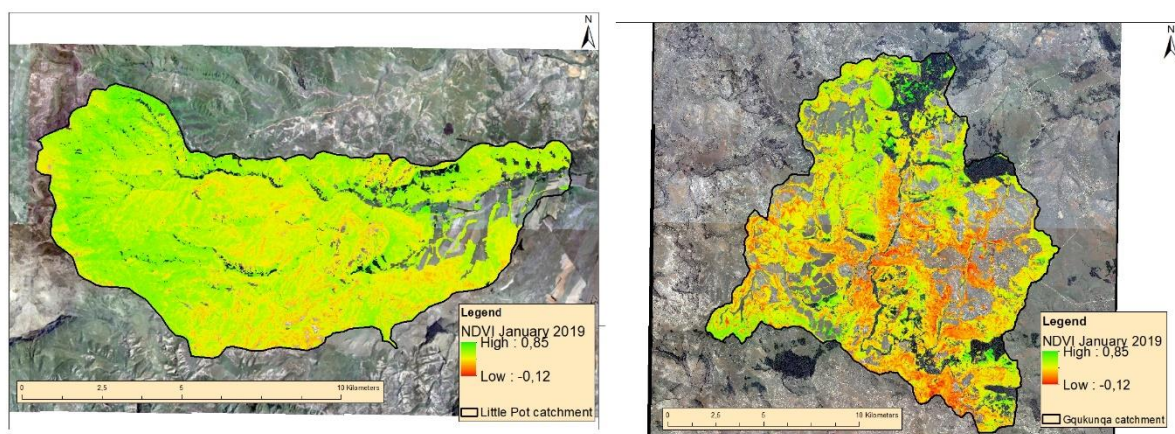


Figure 1.5: Images comparing the NDVI values for grasslands commercially managed grasslands in the Little Pot catchment (left) and communally managed grasslands in the Gqukunka catchment (right)(Herd-Hoare, 2019). See the location of the two rivers in Figure 3.

1.4.4 Land cover

The UTRC has various types of land cover, with natural vegetation making up ~72 % of the total land cover, comprising mostly grasslands. Smaller land cover classes such as thicket, natural forests and shrublands make up the remainder of the natural vegetation. A total of 65 % of all grasslands within the catchment are grazed by either commercial or subsistence livestock farmers. Cultivated commercial and subsistence agriculture make up a total of 15 % of the land cover, whilst the remaining 13% comprises plantations, villages/towns and waterbodies (Pretorius, 2016; Figure 1.6). Commercial farms are focused on the production of livestock and crops, while the plantations also play a large role in the functioning of the catchment (Pretorius, 2016). Maclear is the largest settlement in the catchment, whilst the eastern side of the catchment has scattered settlements and communities. This used to be part of the former Transkei and much of the subsistence farming takes place here (indicated in yellow). Much of the catchment is covered by grasslands and this is the focus for this study.

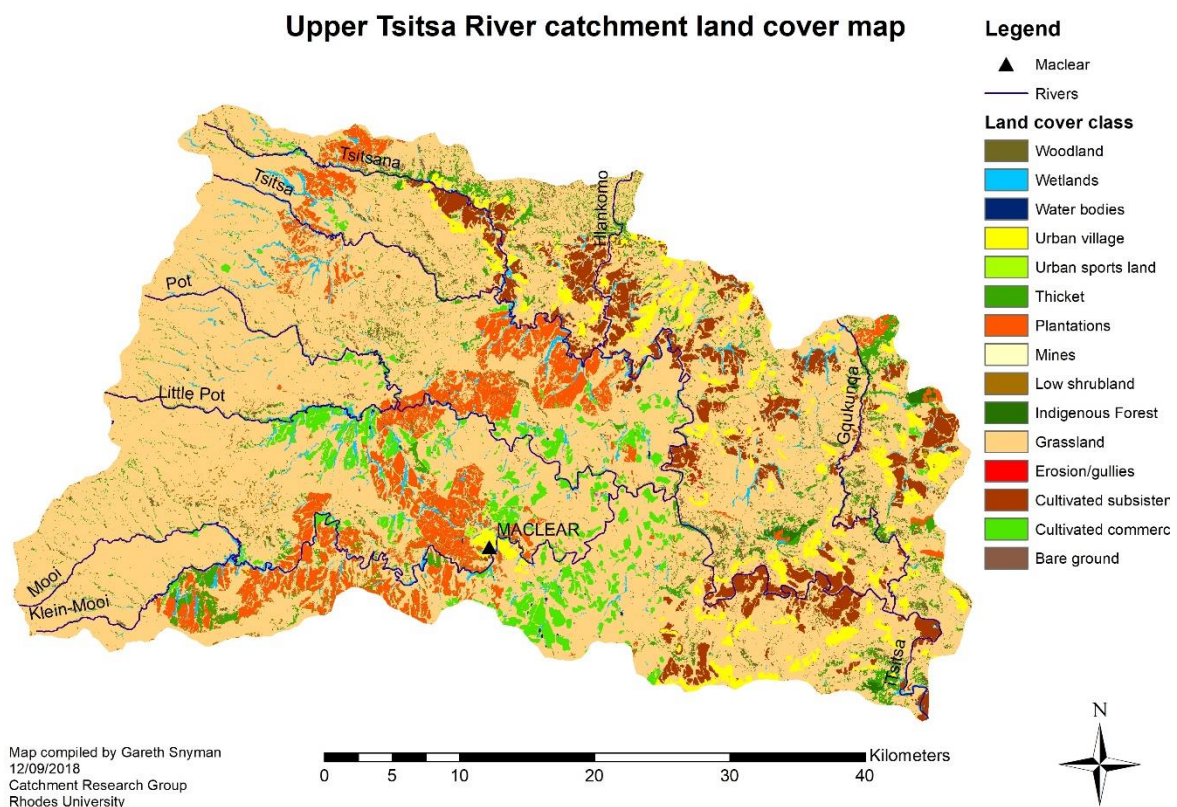


Figure 1.6: Land cover types across the UTRC (Department of Environmental Affairs, 2019)

1.4.5 Commercial and communal land management

The UTRC has a variety of land cover types, and this stems from the stakeholders involved, and determines the land management type. The catchment is roughly split into two land management types (commercial and communal; Figure 1.7). This is evident when analysing the type of landcover; commercial agriculture and plantations in the upper, western side of the catchment, whilst subsistence agriculture and communal grazing occur in the lower, eastern side of the catchment (Figure 1.7). This is because the lower, eastern part of the catchment is part of the former Transkei and falls under the management of chiefs and senior traditional leaders. Land is managed very differently in the commercial areas, compared to the communal areas and these two management areas were considered when assessing the different fire regimes in the UTRC.

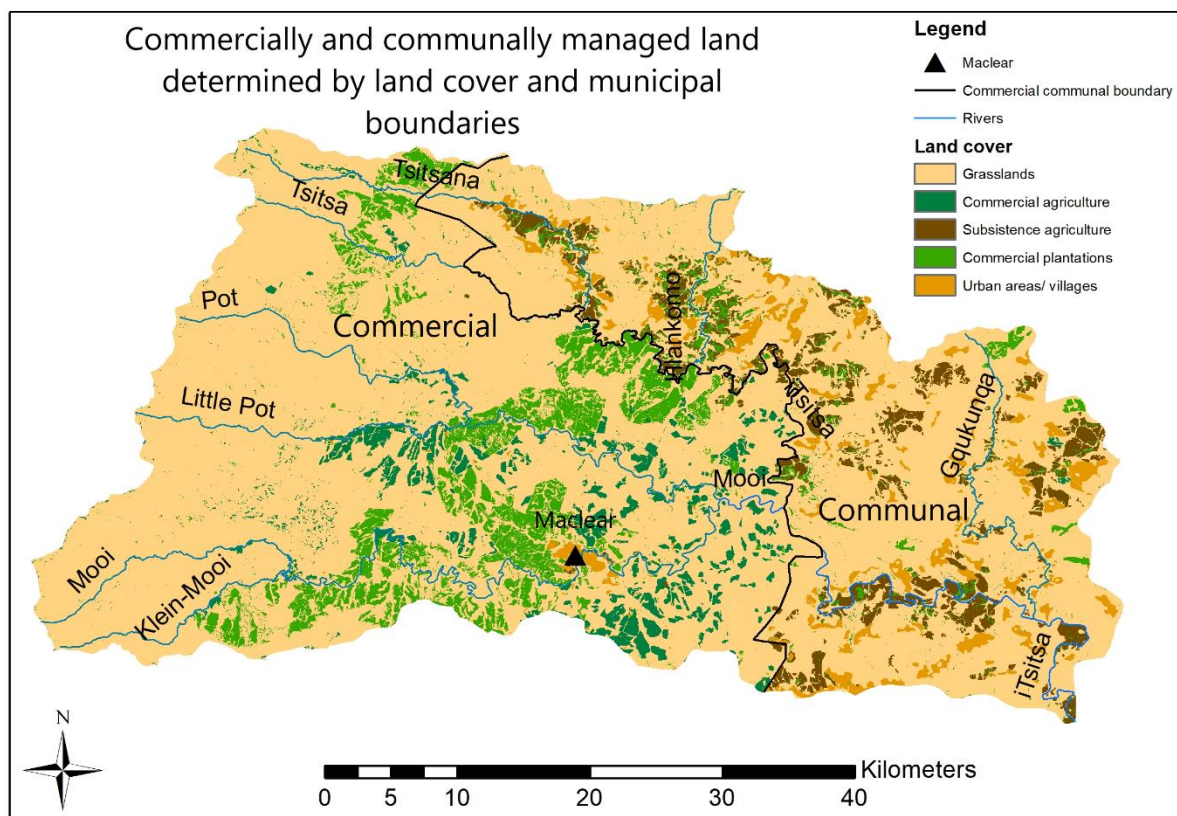


Figure 1.7: Demarcation of commercial and communally managed land in the UTRC determined by land cover and municipal boundaries (Data sourced from Department of Environmental Affairs, 2019).

1.4.6 Summary

To summarize, the UTRC is typical of a sub-escarpment catchment varying geology and grasslands which experience a high interannual rainfall variability. This study focuses on grassland/veld fires as they are a valued natural resource utilised by livestock farmers. As mentioned above, there are two different types of land management, commercial and communal, and these are taken into account throughout this research. Generally, the commercially managed land in the catchment is covered by

sour veld, broken by pine plantations and steep slopes with little evidence of severe erosion. Alternatively, the communally managed land is severely degraded and is covered by sweet veld which is heavily grazed and fragmented.

For the purpose of this study, different aspects of fires and their effects will be assessed in depth, (fire regimes, how fires affect soil properties and vegetation) this serves to explore the nature of the fires occurring in the UTRC and determine how they are affecting the landscape with regards to erosion and vegetation cover.

2. Literature Review

This literature review will cover relevant information on fires, with a brief introduction and explanation of fire regimes, followed by factors affecting and influencing them. The literature on fire regimes will aid in understanding how they behave on a global, regional and catchment scale in order to accomplish the 2nd Objective (Determine fire regimes within the Tsitsa River catchment by observing fire frequency, extent, seasonality, size and intensity for commercial forestry, commercial livestock and communal livestock grassland management). I will then cover literature that pertains to the impacts that fires can have on the landscape, as well as their importance in ecosystems which relates to Objective 3. Lastly, it will cover how fires can be effectively measured and monitored via remote sensing which will cover Objective 1.

2.1 An introduction to fires

Fires are a chemical process whereby molecules of oxygen combine with molecules of cellulose and lignin. The solids are changed into gases comprising of molecules of different substances and is seen as a chemical process rather than a physical process. Changes during this process produce heat and flames, and this is called combustion or fire (Gisborne, 2004; Bond & Keane, 2017). The process of combustion requires three essential components which are referred to as the fire triangle: fuel will combine with oxygen to generate heat, heat to raise fuel to the ignition point (e.g. lightning strikes or match) and oxygen to combine with fuel molecules (Figure 2.1). At a larger scale, fires and their distribution are determined by similar controlling factors: vegetation to burn, environmental conditions (temperature, relative humidity etc.), topography and ignition sources (Figure 2.2) (Archibald *et al.*, 2009; Krawchuk *et al.*, 2009).

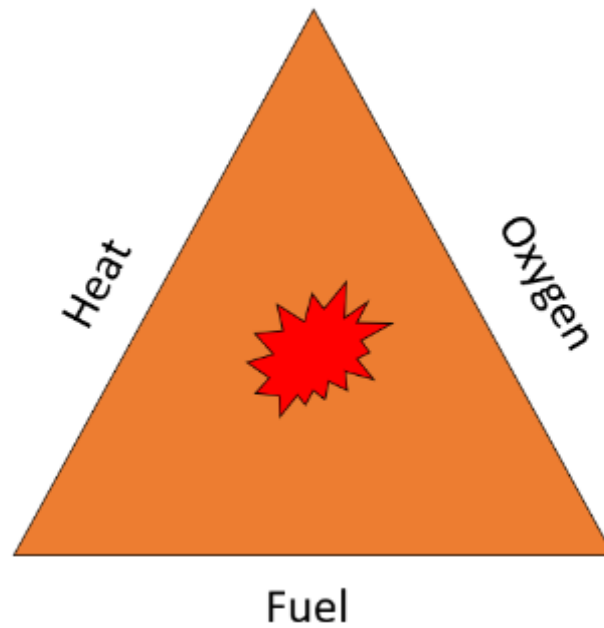


Figure 2.1: The fire triangle represents components needed for combustion (Adapted from Bond & Keane, 2017)

The spatial and temporal occurrence and patterns of fires in a landscape are referred to as a fire regime (Bond & Keane, 2017). Fire regimes are determined by the combined effects of vegetation, climate and ignition frequency and are defined by key characteristics (Frequency, size, intensity, season) (Archibald *et al.*, 2013; Bond & Keane, 2017; Figure 2.2). Fires have been present on earth since the appearance of plants to provide fuel over 350 million years ago and form a vital part of our landscapes and ecosystems, transforming and altering them in a multitude of ways. Natural fire regimes are essential to certain landscapes; however, human influence has altered the timescale of natural fire regimes with the possibility of knock-on effects on vegetation and soil properties (Bond & Keane, 2017). Fires pose a potential hazard to human-made infrastructure (buildings, agriculture etc.) and negatively impact many landscapes.

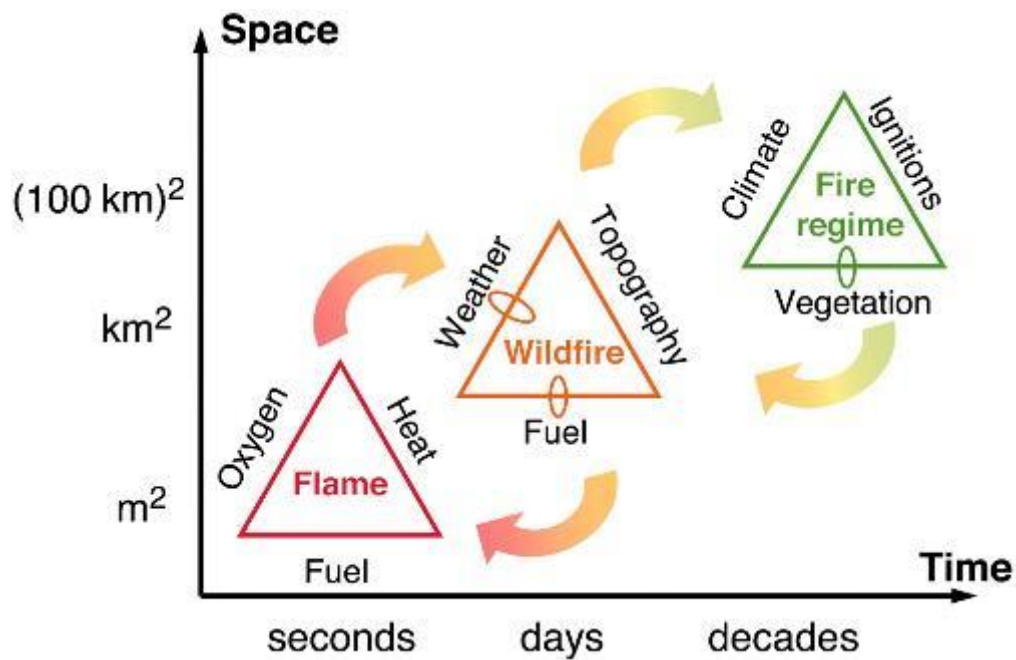


Figure 2.2: Using the Fire triangle concept to explain controlling factors of fires at different spatial and temporal scales (Mortiz *et al.*, 2005).

Although often viewed negatively, fires are beneficial to many environments by cycling nutrients, maintaining vegetation cover and controlling vegetation composition that are all vital in natural and human-made settings (Bond & Keane, 2017). Much of the literature supports the fact that degradation is strongly related to frequent fires, whilst this, in many cases is not true. It is important to investigate literature regarding the most efficient way to monitor and observe fires as this has evolved with the rapid advance in remote sensing technology and satellite monitoring. The topics mentioned here address some of the key points related to this research project and have been explained in depth below.

2.2 Fire Regimes

Fires are often described in terms of fire regimes that describe the repeated pattern of fires in a specific location or area over an extended period of time (Brown, 1995; Archibald *et al.*, 2013). The term ‘fire regime’ was originally used to explain vegetative responses to fire, but they can also be used to quantify emissions from fires and when planning fire suppression or control (Collins, 1992; Archibald *et al.*, 2013). Being able to characterize fires is particularly important if we hope to successfully predict how fire patterns might change in response to environmental and human drivers. Archibald *et al.*, (2013) identified five key characteristics that should be considered when studying fire regimes: size, frequency, intensity, season and extent. Despite the above, widely accepted, definition, fires are still difficult to define or quantify at a spatial scale. A new term coined by Archibald *et al.*, (2013) is the term ‘Pyrome’. This, to an extent, describes the fire regime of a particular region, defined by frequency, intensity and fire size (Archibald *et al.*, 2013).

When identifying pyromes, Archibald *et al.*, 2013 identified and mapped the five key fire regime characteristics. The measurements derived from the key characteristics are insightful and describe

ecologically meaningful metrics of fire (Archibald *et al.*, 2013). However, there is some confusion between two of the key characteristics that Archibald described; namely size and extent. The difference can be seen in the way that these two variables were measured. The measurements of Archibald *et al.*, (2013) for fire frequency, intensity, seasonality, size and extent were: Fire Return Interval FRI, maximum fire intensity using Fire Radiative Power (FRP), length of the fire season, maximum fire size and mean annual burned area respectively. Fire intensity is a measure of the rate of energy released by a fire (Keeley, 2009; Archibald *et al.*, 2013), FRI is indicative of the available growth period between fires for plants and vegetation to recover. Small changes in fire frequency or intensities can result in altered vegetation community composition by promoting or excluding different types of plants that may play important roles between vegetation and fire feedbacks. The size of fires and the time of year in which they occur reflects the availability and flammability of fuels (Archibald *et al.*, 2013).

Five distinct pyromes were identified by Archibald *et al.*, (2013), each associated with specific types of fire (Figure 2.3). Pyromes are defined by their size, fire intensity and spatial distribution (Archibald *et al.*, 2013). The different pyromes identified were [frequent, intense, large (FIL)], found mostly in Australia, whereas in Africa, [frequent, cool, small (FCS)] fires dominated. Two pyromes had irregular fire patterns [rare, intense, large (RIL)] and [rare, cool, small (RCS)]. The final pyrome has an intermediate fire return period with relatively small fires [intermediate, cool, small (ICS)], and is common across the globe in disturbed areas, such as regions of deforestation or agriculture (Archibald *et al.*, 2013). Pyromes tend to correspond to specific biomes, with RIL and RCS pyromes most commonly occurring within boreal forests that are dominated by conifers, pines and spruces. It was also found that FIL and FCS pyromes were most common in tropical grasslands, whilst ICS fires were common in moist broadleaf forests. The correspondence of the pyromes to specific biomes is due to the different types of fuel that are available to the fire, namely grass vs. litter fires. Grasses can regrow lost biomass quickly enough to sustain frequent fire regimes (1-3 years), which would be characteristic of FIL and FCS pyromes (Archibald *et al.*, 2013). This is supported by the research which shows that FIL and FCS pyromes dominate high seasonality rainfall tropical and temperate grasslands.

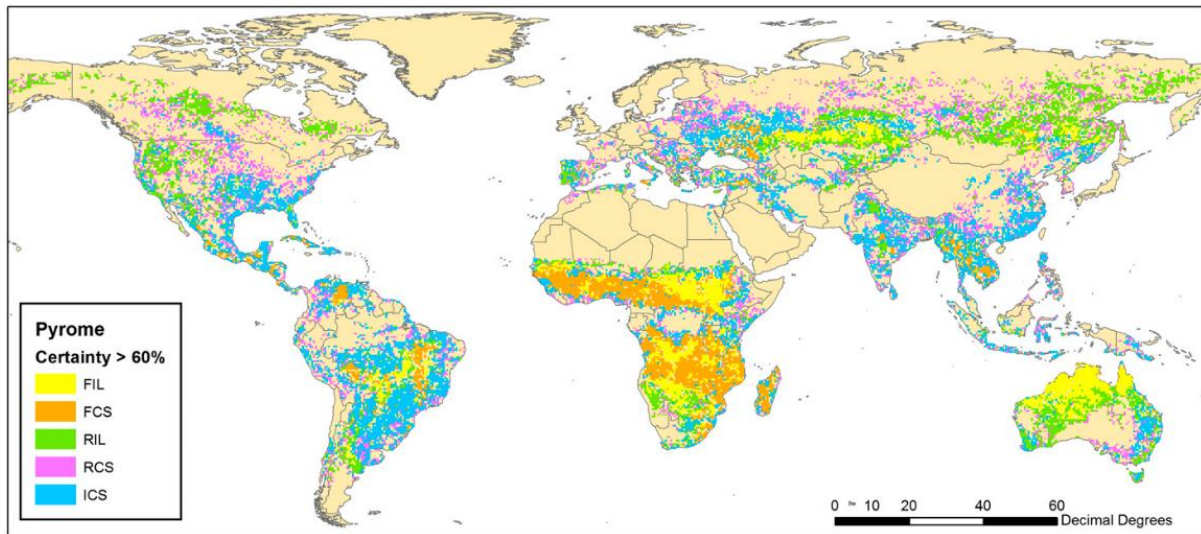


Figure 2.3: Global pyromes (From Archibald *et al.*, 2013)

2.2.1 Seasonality

Fires can burn all year round, provided they meet the criteria for combustion; available fuel loads, correct environmental conditions and an ignition source. Fire seasonality refers to the time of year and season when fires most often occur in a specific region or area and is a critical variable that affects the impact of fires on grasslands. However, it is evident that different regions have specific seasons or times of year when fires are most common. Fire seasonality differs depending on climate and when seasons fall during the year. The key trends that determine fire seasonality in grasslands are high fuel loads with a low moisture content. Environmental factors such as a low relative humidity and warm temperatures also allow for high ignition potential (Archibald *et al.*, 2009; Krawchuk *et al.*, 2009). These variables are most common during the winter season in the upper Tsitsa catchment, when rainfall is low and dry conditions are experienced.

Climatic changes between seasons, and biomass availability, heavily influences fire intensity (Bragg, 1981). A study of fires in South African savannas (largely grass cover; summer rainfall) was conducted in Kruger National Park (KNP). It was found that fire intensities were drastically affected by season (Govender *et al.*, 2006), and were lowest during summer, increased during autumn, and were at their highest during winter (June to October). The fire season was associated with the difference between the moisture content of the fuel loads in summer and winter, being 88 % and 28 % respectively (Govender *et al.*, 2006). Grassland fuel loads were found to increase with post-fire age over the first two years but did not differ significantly afterwards. Vegetative growth post-fire was dependent on the climate, as increased rainfall resulted in higher fuel loads (Govender *et al.*, 2006). Fuel loads tend to accumulate in proportion to rainfall for the first 5 years after a fire, after which they, or biomass, were found to decline because of grazing, decomposition and loss of plant vigour, leading to an equilibrium that is

reached. Fire return periods in the KNP are between 3 and 6 years, meaning that fires often remove fuel loads before equilibrium is met (Govender *et al.*, 2006). An analysis of approaches to fire management in African grasslands by Govender *et al.*, (2006) suggests fire management had little impact on fire return periods, which are dependent on grass biomass and in turn determined by the amount of and variability of rainfall. However, seasonality of fires and fire intensities are elements of fire regimes that could be influenced using the correct management strategies.

2.2.2 Frequency

Fire frequency refers to the number of times a plot of land is burned over a period of time. This can be expressed using the FRI, probability of occurrence or rotation period (Morgan *et al.*, 2001). FRI is the time in between fires where the biomass has time to accumulate and recover (Brooks *et al.*, 2004; Archibald *et al.*, 2010; Archibald *et al.*, 2013). Fire frequency varies drastically and has many influencing factors, one of the biggest being humans (Whisenant, 1989; Morgan *et al.*, 2001; Archibald, 2016). Anthropogenic impacts on fire regimes, and more specifically frequency, began with the evolution of modern humans being able to ignite fires and manipulate land cover. Globally, natural fire regimes are affected by a colonial past focussed on fire suppression and preservation of landscapes, whilst nowadays fires are used as a land management tool (Bond & Keeley, 2005; Archibald, 2016). Grassland fires have natural return periods ranging from two to six years. However, when influenced by humans (as in the case of forestry fire breaks, fire suppression, introduction of alien invasive plants), return periods are as high as every two years (Archibald *et al.*, 2010). The frequency of fires is closely linked to fire intensity as it is closely linked to biomass. Infrequent return periods result in intense fires due to biomass build-up, however frequent return periods result in smaller, less intense fires as a result of smaller fuel loads (Murphy *et al.*, 2013). Southern Africa experiences large annual burned area due to the accumulation of many small to medium-sized fires rather than a few large fire events (Archibald *et al.*, 2010).

2.2.3 Size and extent

The size of fires is determined by a variety of climatic and physical conditions. Fuel load, fuel moisture, fuel type, wind direction, topography and slope all play a part in determining the size and behaviour of fires (Trollope & Trollope, 2002). Many of the properties mentioned above also affect fire intensity, so it can be expected that there is a relationship between fire size and intensity (Archibald *et al.*, 2013). The higher the intensity, the more likely the fire is to grow to a larger size by igniting surrounding fuel through high heat transfer. Other factors such as physical barriers, fragmented landscapes and even previous fires influence and limit how fires can grow and take shape (Archibald *et al.*, 2013).

Spatial information on fire extent and distribution is rare due to the difficulty of recording and analysing the data. Detailed records of fire scars are only usually kept for locations where studies have been carried out. Archibald *et al.*, (2013) has developed a method for identifying individual fires from the MODIS burned area dataset using a flood-fill algorithm. This algorithm assumes that pixels adjacent to each other have burned in a similar period and are from the same fire. This allows MODIS data to be used to identify fire scars, however, these are low-resolution data (1 km²) and are only applicable to large scale studies. Human densities also have different effects on the size and frequency of fires, with the number of fire ignitions increasing with population density. However, there is a decrease in the size of fires, which means that total burned area decreases despite an increase in fire ignitions with increasing human populations (Archibald *et al.*, 2010). Studies by Archibald *et al.*, (2010) show that areas that burn most frequently have many fires, but not necessarily the largest fires. The majority of the total burned area is caused by the largest fires - the top 1% of fires burn 99% of the area (Archibald *et al.*, 2010).

2.2.4 Intensity

Fire intensity describes the physical combustion process of energy being released from organic matter, mostly vegetation (DeBano, 1998; Keeley, 2009; Scott, 2012). The terms intensity and severity are often used incorrectly to describe fire and the effect it has on a landscape. Fire intensity can be defined as the measure of the energy output of a fire, whilst severity is defined as the loss or change in organic matter above ground and below ground after a fire (Keeley, 2009). The term severity relates to the ecosystem's response to fire, which includes soil erosion, vegetation regeneration, restoration of community structure and a variety of related responses (Morgan *et al.*, 2001).

In the field of physics, intensity is defined as the energy per unit volume multiplied by the velocity at which the energy is moving. The result is measured in W.m² which is the rate of heat transfer per unit length of the fireline and represents the radiant energy of the front of a fire. There are many uses for knowing the intensity of a fire, and this equation has been adapted to suit other areas of fire science (Keeley, 2009). Fireline intensity can be used to characterise a fire and such information can be sensed remotely (Keeley, 2009). FRP can be detected by a variety of satellite sensors such as ASTER and MODIS (Giglio *et al.*, 2008). Satellite sensors identify and record energy emitted from a fire using different thermal infrared channels in Megawatts per pixel. FRP over time allows one to work out the biomass consumption rate, as well as the fireline intensity (Giglio *et al.*, 2008).

An example of how fire intensity can be used in the field of fire science is presented in Trollope and Trollope, (2002). Different types of fires tend to affect vegetation distinctively with grasslands being the exception. Head and crown fires affect the tops of trees as their growing points are exposed to the release of heat and energy in the canopy, making them more sensitive to fire intensity. Head and crown

fires have little effect on grasses and shrubs due to the vertical structuring of fuels in a landscape (Trollope & Trollope, 2002). Surface fires affect grasses and shrubs directly, however, grasses are not generally sensitive to increased fire intensities due to the low-lying location of their growth points, away from the release of heat energy (Trollope & Trollope, 2002). Intense fires, thus, favour the development of grasslands and open savannas and vice versa (Trollope & Trollope, 2002). Some guidelines presented by Trollope and Trollope, (2002) recommend the application of different intensity fires for controlled burns. When burning is conducted to remove moribund grass material, a cool fire of <1000 kJ/s/m is recommended, because it is safer and relatively easy to manage and will not have any different effect on the grass sward when compared to a high-intensity fire. Optimal climate conditions for a cool fire would be to burn when the air temperature is less than 20°C and the relative humidity greater than 50 %. When utilising fire to control undesirable vegetation such as trees or shrubs, an intense fire of >2000 kJ/s/m is necessary. Fuel loads need to be greater than 4000 kg/ha, the air temperature greater than 25°C and the relative humidity lower than 30%. This will cause a top kill of stems and branches in trees and shrubs up to a height of approximately 3 m. In both cases, the wind speed should not exceed 20 km/h to prevent runaway fires (Trollope, 1999). These are the ‘optimal’ conditions suggested when managing fires, but in reality, it is difficult to manage the variables mentioned above. It is very difficult to control a fire in this way as fires will burn according to the climatic conditions, topography and fuel load availability, as well as the fuel load density (DeBano *et al.*, 1998; Brooks *et al.*, 2004).

2.2.4.1 Fire severity

As mentioned above, severity is used to describe the level of alteration caused by fire and is inherently different from fire intensity (Keeley, 2009). Fire severity can be difficult to classify as wildfires vary spatially, resulting in complex mosaics of burned and untouched patches of vegetation that may differ in burn severity (Vega *et al.*, 2013). Severity is recognised as a key controlling factor when determining post-fire erosion rates and although there are broad definitions of the term, the relationship between soil burn severity and post-fire hydrological response has not been fully established (Vieira *et al.*, 2015). Being able to determine the level and extent of soil burn severity will aid in post-fire rehabilitation treatments, essentially preventing the alteration of hydrological and erosion responses in the environment (Vega *et al.*, 2013). Several visual indicators of soil burn severity have already been established using immediately observed changes in affected areas. The most common indicator of burn severity is the consumption of soil organic matter, changes in colour and structure in soils and the deposition of ash post fires. These indicators are difficult to compare across landscapes due to environmental factors such as differences in vegetation and soil types (Shakesby & Doerr, 2006; Vega *et al.*, 2013). A study by Vega *et al.*, 2013 investigated visual signs of soil burn severity levels to test the relationship with changes in soil properties. It was found that a categorical approach worked well when looking at burn severity and a visual assessment was able to determine the degree of degradation of important chemical and microbiological properties in soils. This only applied to sites similar to that

of the study and cannot be applied at a larger scale. However, the research did show that it is possible to rapidly visually assess fire severity and possibly make a link between soil burn severity and the effects it has on the landscape and is beneficial to ecosystem managers and the rehabilitation processes that take place post-fires (Wohlgemuth, 2009).

2.3 Factors affecting fire regimes

Fire regimes are complex systems that are influenced by many variables, however, the three main determining factors of fire regimes are climate, vegetation and human influence (Ryan, 1991; Morgan *et al.*, 2001; Archibald *et al.*, 2009; Archibald *et al.*, 2013). Each of the above factors can be further sub-divided into multiple variables that contribute to fire regimes and their specific characteristics. These multiple variables and characteristics can alter and determine the where, how, when and why fires occur. Ryan, (1991) explained the effects of climate and vegetation on fires before human influence and found that there are multiple feedbacks between variables affecting fires, creating a complicated system which changes over time (Figure 2.4).

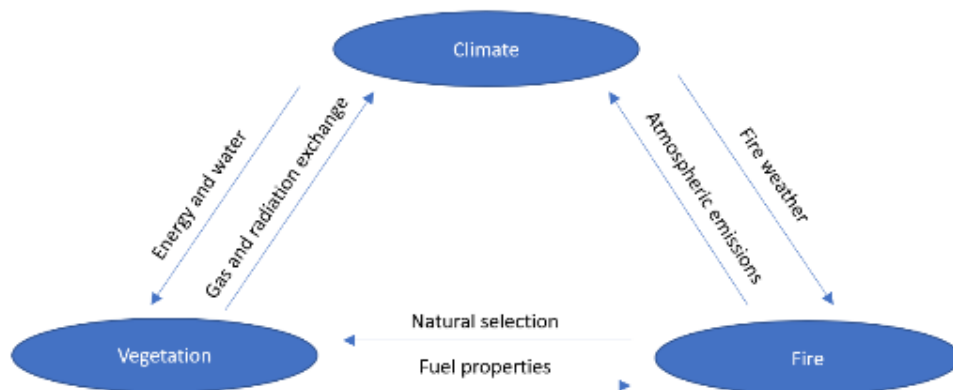


Figure 2.4: Climate, Vegetation and Fire interactions (Adapted from Ryan, 1991).

Humans have influenced how, when and where fires occur through specific legislation, fire suppression, creating man-made barriers and using fires as a tool for land management (Brontons *et al.*, 2013; Archibald, 2016).

2.3.1 Anthropogenic activity and its effects on fire regimes

Earth is prone to fires due to the presence of carbon-rich vegetation, seasonal dry periods, atmospheric oxygen and variety of widespread ignition sources such as lightning and volcanoes (Bowman *et al.*, 2009). Before humans began to influence our planets natural systems, the interactions between climate, vegetation and ignition sources defined fire regimes (Ryan, 1991). However, humans added another major factor to characterising fire regimes once they were able to harness fire, altering natural fire regimes and making them more difficult to predict in densely populated areas (Brotons *et al.*, 2013).

Anthropogenic activity comprises anything related to humans, but in the context of fires, it refers to the management of fire, as well as altering the natural environment (land cover) which in turn effects fire regimes. This section will cover some of the direct impacts human activity has on fire regimes.

2.3.1.1 Fire suppression

Natural fire regimes (independent of humans) have played an integral role in the evolution of most ecosystems, affecting biological evolution of fauna and flora, as well as global biogeochemical cycles (Syphard *et al.*, 2007; Bowman *et al.*, 2011). However, human influence and activities alter natural fire regimes that can result in a change within ecosystems (Syphard *et al.*, 2007). Severe impacts of fires seen now are influenced by our colonial past, which focused on preventing ignition and suppressing fire (Archibald, 2016). Periodic wildfire (according to natural fire regimes) maintains fuel loads as well as species composition of ecosystems (Kraaij *et al.*, 2001). Fire suppression is a direct anthropogenic impact, altering fire regimes, with mixed results (Brotons *et al.*, 2013). Some research suggests that fire suppression may reduce the frequency of large fires, however, studies have been limited to specific landscapes. Alternatively, there is also evidence to suggest that intense fire suppression can result in large and intense fires (Brotons *et al.*, 2013). This has been well documented and is thought to be the reason for some of the recent catastrophic fires in California (WERT, 2018) and Knysna in 2017. Part of the problem, in both cases, was the build-up of excess vegetation or fuel loads over many years, due to the absence of smaller, regular fires which are often found to be beneficial. In prairies and grasslands, fire suppression has caused the loss of up to 50% of plant species (Bond & Keeley, 2005). The loss of species compositions shows that altering fire regimes can have negative effects on the landscape (Bond & Keeley, 2005). Extreme fire suppression can result in extremely intense and severe fires which have the potential to do serious damage to the landscape and man-made infrastructure, as well as endangering lives. Human activities have altered the natural range and variability of fires and this has had large an impact on ecosystems and fire return intervals (Syphard *et al.*, 2007).

2.3.1.2 Landscape fragmentation

In South Africa, the natural seasonal patterns of fires in grasslands has been altered by a variety of human activities, causing fragmented landscapes. Fire is the main grassland management tool as it is the simplest and least expensive to improve poor quality grasslands for livestock (Ratsele, 2013). Using fires as a tool to manage grasslands is not seen as fire suppression, but rather the ability of humans to decide where and where not to burn and this can cause fragmentation (Gill & Williams, 1996). Human-made fragmentation refers to any structure or activity that alters the natural landscape. Examples of what may cause fragmented landscapes are towns, roads, agricultural, grazing lands, plantations, exotic/alien invasive vegetation, topography, conservancies, planned burning (fire breaks) and fire suppression strategies (Gill & Williams, 1996; Bond & Keane, 2017; Archibald, 2016). Fragmentation of grasslands results in a shifting mosaic of plant and animal communities within a landscape, with the rate of vegetation growth determining the fuel availability for future fires (Bond & Keane, 2017). The

degree of fragmentation can vary depending on location, with more than one type of fragmentation being present in certain areas. The Tsitsa River catchment has multiple land cover classes, possibly resulting in higher fragmentation and altered fire regimes. Commercial plantations fragment the landscape as they create areas where fires are suppressed. However, fire breaks are a common tool used to protect these areas, creating a break in vegetation and fuel loads to prevent wildfires from destroying valuable crops (Archibald, 2016; Bond & Keane, 2017). Another factor that causes fragmentation of vegetation is mismanagement of livestock, which leads to overgrazing. High livestock numbers in the communally managed areas are resulting in excessive degradation and heavy use of natural resources. Heavily grazed areas can act as barriers for fires due to the low availability of fuel and are altering fire regime characteristics (Fica, 2011).

2.3.1.3 South African legislation

Veld (vegetation) fires were identified as a risk to life, property, assets and the environment in many parts of South Africa, therefore, the National Veld and Forest Fire Act (NVFFA) (Act No. 101 of 1998) was adopted as a means of improving veld management across the country (Kruger *et al.*, 2006). The Act had two purposes; to “prevent and combat veld, forest and mountain fires throughout the country and provide a variety of institutions, methods and practices for achieving the purpose” (Act 101, 1998). The risk as mentioned above is also broken into two main parts, the risk fires pose to assets as well as the risk that inappropriate fire regimes pose to ecologically fire-dependent areas (Kruger *et al.*, 2006).

The burning of fire breaks in South Africa is determined by the NVFFA, which outlines when, why and how fire-breaks and strategic burning can be carried out (O’Connor *et al.*, 2004). This was done by implementing a fire danger rating system, which ultimately determines when it is safe to ignite fires for the purposes of land management and fire breaks. This is controlled by fire protection associations and fire protection officers, who oversee the development of a fire management strategy for the local area and ensure that rules and regulations are adhered to in order to minimise uncontrolled fires. Burning rules and regulations are often in accordance with the SANBI grazing and burning guidelines which consider natural fire regimes, the ecological importance of fires, and the impacts of fires on a landscape (SANBI, 2014). What differentiates the SANBI guidelines from the NVFFA is that the SANBI guidelines were designed to encourage land managers to consider the ecological importance of grasslands and apply biodiversity-friendly principles to their management scenarios rather than act as a set of rules. The SANBI guidelines are centred around three main concepts; (1) grasslands response to burning and grazing, (2) the best practice principles for burning and grazing, and (3) a farm-scale approach to burning and grazing (SANBI, 2014). These main concepts set up the guidelines for the best use of fires for biodiversity, whilst taking commercial livestock farming practices into account. SANBI recommend that in a rotational grazing and burning plan, fires should burn every 2 years. However, this is not the case for fire breaks as their purpose is to prevent runaway fires from destroying assets, and

the NVFFA states that fire breaks in grasslands need to be burned before mid-winter on an annual basis (O'Connor *et al.*, 2004; van Wilgen *et al.*, 2012).

2.3.2 Climate, season, weather and how they influence fire regimes

Climate is one of the overarching governing factors of fire regimes and changes in climate are expected to result in an altered distribution of wildfires (Krawchuk *et al.*, 2009). The effects of climate on local scale fire regimes are complex and difficult to measure, however, regional fire activity fluctuates year to year with climatic variability (Swetnam, 1993). Climate determines a resource's availability to burn and includes weather conditions (temperature and precipitation variables) and ignition potential. Archibald *et al.*, (2013) used the term 'pyrome' to define the average size, how often and at what intensity fires burn in different regions around the world. The study identified three climate variables (effective rainfall, interannual variability of rainfall and seasonality of rainfall) that are applicable on a global scale and explain the spatial variability of fire regimes. Using these variables, it was found that frequently burning pyromes are more prominent in arid environments where there is strong rainfall seasonality with distinct wet and dry periods (Archibald *et al.*, 2013). Similar studies found that fire-prone areas (Africa, America, South East Asia and Siberia), are generally sub-equatorial and are associated with cyclical dry-wet conditions, seasonally and interannually (Chuvieco *et al.*, 2008). There is a strong correlation between water and energy budgets and vegetation composition and structure, which is determined by regional climatic conditions (Ryan, 1991). Historical data shows that climate drives fire regimes when it promotes vegetative growth and accumulation of fuel loads followed by drought and warmer periods that are fire conducive conditions (Bowman *et al.*, 2009). Further dendrochronological (dating of tree rings) and fire scar observations suggest that there is a relationship between levels of fire activity and climate oscillations (Simard *et al.*, 1985; Swetnam, 1993; Kitzberger *et al.*, 2007). Changes in sea surface temperatures (SSTs) have knock-on effects on climatic conditions on a continental scale. Simard *et al.*, (1985) contend that the area burned annually across the southern U.S is affected by the El Niño-Southern Oscillation (ENSO) and was found to be higher during La Nina years with decreases in the annual burned area during El Nino years. Further research by Kitzberger *et al.*, (2007) over western North America included ENSO, Pacific Decadal Oscillation (PDO) and Atlantic Multidecadal Oscillation (AMO) as impacts. The changes in climate from these interannual and multidecadal cycles influence sea surface temperatures and climatic conditions, driving variability in fire regimes. Kitzberger *et al.*, (2007) found that fires occurred more often during certain phases of ENSO and PDO, with these two climatic cycles being the main driver of high fire frequency variability. However, fires were found to be synchronous in western America during warm phases of the AMO. The AMO was also shown to alter the strength and spatial influence of ENSO and PDO on fire occurrence at a multidecadal scale. With climate change being a prominent issue, current warming trends in the AMO might suggest an increase of fires across the western U.S. (Kitzberger *et al.*, 2007). Warmer weather and changes in climate cycles have caused certain regions to experience extreme fire-

conductive conditions such as severe drought, accumulation of fuel loads, low fuel moisture and fire-conductive weather which has been attributed to the outbreak of many wildfires across the planet (Whitlock, 2003).

Southern Africa is not exempt from climatic cycles and is heavily influenced by the Antarctic Annular Oscillation (AAO) and the ENSO, which is thought to be the biggest climate driver of interannual rainfall variability in southern Africa (Philippon *et al.*, 2012; Dieppois *et al.*, 2015). Southern Africa receives most of its rainfall during the summer seasons and is influenced by the ENSO, experiencing dryer than normal conditions during El Nino events and wetter conditions during La Nina events. However, winter rainfall variability is related to the AAO, Antarctic sea ice movement and South Atlantic sea surface temperatures (Philippon *et al.*, 2012). Wetter summers during La Nina events may result in large fuel loads and more intense fires during the dry (winter) season. However, El Nino events favour drought in southern Africa and this can cause extended dry periods, influencing fire season time and length, ultimately effecting and altering fire regimes (Dieppois *et al.*, 2015).

A question posed by researchers and land managers is to what extent are fire regimes caused and altered by human influence (land and fire management) or is it a product of climate change- human and/or natural. It is evident that the climate is changing drastically due to human influence and this is evident in the increase of atmospheric carbon dioxide (CO₂) levels (Bowman *et al.*, 2009; Williams & Abatzoglou, 2016; Figure 2.5). The industrial revolution marked the beginning of burning large amounts of fossil fuels, releasing excess CO₂ into the atmosphere.

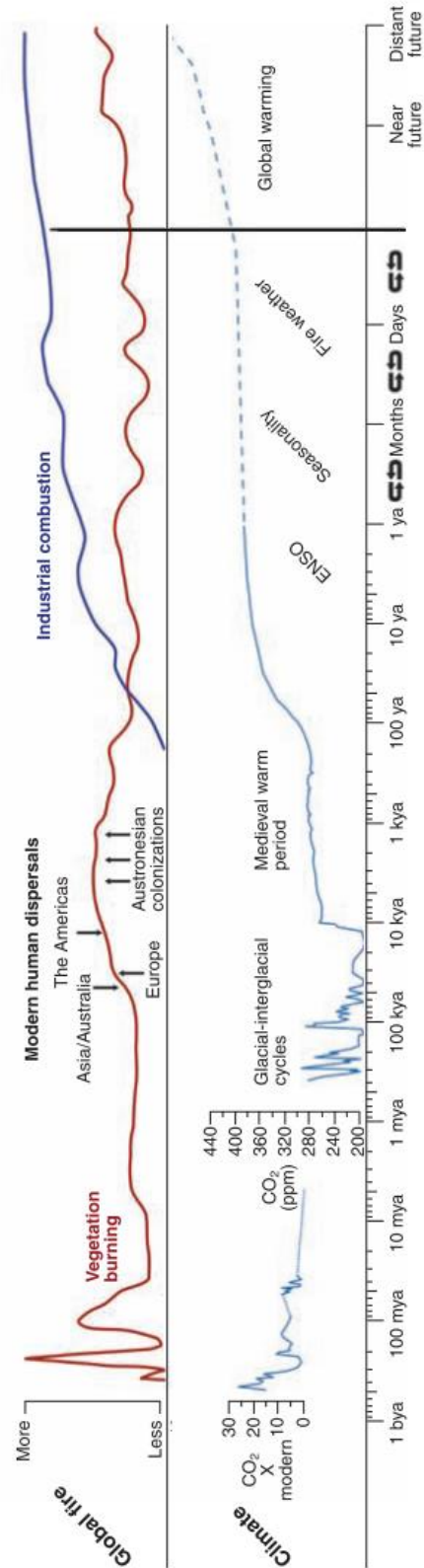


Figure 2.5: Global fire trends compared to climate change, more specifically atmospheric carbon dioxide (Adapted from Bowman *et al.*, 2009)

Modernization and mechanization of many processes required fossil fuels to be burned, and whilst the earth's atmosphere has seen fluctuations and high levels of CO₂ before, it has never seen the drastic increase in atmospheric CO₂ that it is currently experiencing (Bach, 1979; Friedlingstein *et al.*, 2001). Atmospheric CO₂ is expected to keep on increasing as climate change reduces the ability of land and

oceans to absorb CO₂ (Friedlingstein *et al.*, 2001). Fires emit carbon dioxide in the process of combustion, however, it is difficult to estimate the contribution of grassland/veld fire emissions among the many other uses of fire, whether it be the burning of fossil fuels, industrial or domestic use. It is thought that all sources of landscape and biomass fires (land management, deforestation/land clearing, natural) contribute up to half of the CO₂ emissions that all fossil fuel combustion creates (Bowman *et al.*, 2009). This is a significant contribution to the total atmospheric CO₂ levels as fossil fuel combustion results in over 7 Pg. C year⁻¹ (1 Petagram = 1000 000 000 tonnes) (Bowman *et al.*, 2009). Increased levels of CO₂ and other greenhouse gasses have resulted in increased global temperatures over the past century (McCarthy *et al.*, 2010; Dai *et al.*, 2015). Increases in global temperatures directly affect the climate and indirectly affects vegetation and fuel loads, two key drivers of fire regimes (Bowman *et al.*, 2009). Fire prediction models offer an insight into what fire regimes may be like in the near future. However, predicting fire regimes requires accurately simulating multiple complex ecological, physical and human interactions which is near impossible. Prediction models are used instead to examine the direction and scale of simulated changes in the environment which infer the potential impacts on fire regimes (Whitlock, 2003). It is unclear how the climate will change, however, there is a concern that a significant increase in summer temperatures without the accompanying increase in rainfall will result in water-stressed areas. Drought-stressed vegetation and warmer temperatures are conducive to fire activity, which could result in longer fire seasons and an increase in the annual burned area (Whitlock, 2003).

2.3.2.1 Fire conducive weather conditions in South Africa

South Africa is home to several different biomes where fire is an integral part of the landscape such as the fynbos region and the open savannas. The UTRC is in the northern part of the Eastern Cape and experiences seasonal rainfall, with wet summers and cool dry winters, which are when most fires occur. A Fire Danger Rating System has been implemented by the Department of Agriculture, Forestry and Fisheries (DAFF) and provides a measure of the seriousness of burning conditions, with the prohibition of igniting fires when the danger rating is too high (DAFF, 2013). The danger-rating system takes Fire Danger Indices (FDI) into consideration which are maximum temperature (°C), relative humidity, wind speed and days since last rainfall. The relative indices are modelled and produce a value (0-100) indicating of the likelihood of fires occurring in a certain area on a given day, with different alert stages (DAFF, 2013; Table 2.1).

Table 2.1: Fire Danger Indices and danger ratings (DAFF, 2013).

Alert stages	FDI	Fire danger	Ratings
Blue	0-20	Low	Insignificant
Green	21-45	Moderate	Low
Yellow	46-60	Dangerous	Medium
Orange	61-75	Very dangerous	High
Red	76-100	Extremely dangerous	Extremely high

The FDI system considers fire behaviour, suppression measures and whether fires can be ignited by farmers for land management purposes. The higher the rating, the more dangerous it is to ignite fires for fire breaks and land clearing. Ideally, commercial farmers would burn on blue or green days as minimal suppression measures need to be put into place and a low chance of fires getting out of hand and burning uncontrollably (D. Sephton, personal communication, 10 October 2018). With regards to natural sources of ignition, i.e. lightning, which occurs during the wet summer season, the FDI is generally low due to the high relative humidity and recent rainfall. This results in lightning ignitions being relatively low and low intensity, slow-burning fires. However, during drought and low rainfall years, lightning ignition could be problematic. Archibald *et al.*, 2009 found that lightning during the fire season was one of the least important ignition sources, and that anthropogenic influence played a larger role in fire occurrence.

2.3.3 Vegetation-fire interactions

Vegetation is one of the major determinants of fire regimes as it affects the type, intensity and size of fires. Vegetation is controlled by climate at a regional scale; a strong relationship exists between water and energy budgets and the composition and structure of vegetation (Ryan, 1991). Important characteristics include vegetation type, density, distribution and the amount of above-ground biomass available to burn (Bond & Keeley, 2005). Vegetation determines the type of fire due to its chemical and spatial properties in a landscape. Fire is crucial in the maintenance of grassland plant diversity, but its effects are poorly understood in a South African context. Vegetation models show that without fires, most of the eastern half of South Africa would be covered in trees (Bond, 2003), and that fire frequency plays a large role in determining the diversity of grasslands (Morgan *et al.*, 2001; Uys *et al.*, 2004; Arnan *et al.*, 2007). Areas with short fire return periods are generally dominated by resprouting species and grasses, whereas areas characterised by long fire return periods tend to be dominated by obligate seeding woody species (Arnan *et al.*, 2007; Archibald *et al.*, 2013). Resprouters and grasses can recover and produce flammable biomass faster than reseeding species and it is not yet clear whether the vegetation is driving fire return times, or the fire return period is driving the type of vegetation present

(Arnan *et al.*, 2007; Archibald *et al.*, 2013). This may be true for natural, pristine areas; however, it is not the case in areas where fire is used as a tool to clear land and manage crops.

It is thought that species richness would be at its peak at intermediate levels of disturbance, however, this is not always the case. It was found that in different parts of the world, with different climates and plant compositions, fire frequency could cause an increase or decrease in plant diversity (Morgan *et al.*, 2001; Uys *et al.*, 2004). Whilst in temperate grasslands in Australia, there were no clear trends in diversity in relation to fire frequency. The variation in results shows that local studies are required for an in-depth understanding of how fires affect plant diversity in a system and that there is no overarching theory that can be applied to grasslands (Uys *et al.*, 2004).

2.3.3.1 Grasslands

Grasslands are irreplaceable resources that act as watershed covers, effectively controlling erosion and serve as grazing land for wildlife and livestock. Fires and grasslands have coexisted through time and are an integral part of those systems (Vogl, 1979). Key components of fire regimes in grasslands are frequency, seasonality and intensity of fires (Mucina and Rutherford, 2006; Archibald *et al.*, 2013). Lightening is the primary natural source of ignition in grasslands; however, the Tsitsa River catchment sees many fires being started in commercially farmed areas which are used for clearing vegetation for livestock and burning fire breaks (Mucina and Rutherford, 2006). Sour grassland covers most of the upper Tsitsa catchment and makes up the highest percentage of South Africa's vegetation. It is the most fire-prone vegetation in the study area and has been defined as having an extremely high fire frequency compared to other vegetation types as it is not limited by climate, burning every 1-2 years (Kruger *et al.*, 2006). Fire exclusion studies in these grasslands indicate that without fires, there is a succession trend towards shrublands and then to forests, reinforcing reduced fire frequency (Bond *et al.*, 2003). This is only the case in grasslands with an annual rainfall higher than 650mm such as the Tsitsa catchment. In areas that receive less than 650mm of annual rainfall, the exclusion of fires results in an increase in tree density without a change in species composition. Grasslands that receive less than 650mm annually are, therefore, climate limited, whereas areas that receive more than 650mm of annual rainfall are fire limited and can be described as being in a meta-stable state with alternate climate-dependent states (Bond *et al.*, 2003).

2.3.2.2 Invasive species

Invasive plants pose a significant threat to the integrity of native ecosystems around the world. Invasive species can modify vegetation structure and fuel properties of ecosystems, which can affect fire behaviour, altering fire regime characteristics (Brooks *et al.*, 2004). A change in fire regimes can promote the dominance of invasive species and upset the natural fire return intervals, which native species rely on (Morgan *et al.*, 2001; Brooks *et al.*, 2004). Alien invasive plants have the potential to directly affect native vegetation by monopolizing or limiting essential resources. Indirect effects include

altering soil stability, promoting erosion, colonizing previously unoccupied areas, changing fuel loads and ground litter and promoting or suppressing fire. Invasive plants that alter fire regimes are some of the most important ecosystem altering species on earth (Brooks *et al.*, 2004). An example of this is the case of the Snake River Plains in Idaho, North America. Natural fire return intervals varied between 60 and 110 years, with the dominant vegetation being a native plant known as sagebrush (Whisenant, 1989; Brooks *et al.*, 2004). The return interval has since increased to only five years, due to the introduction of Cheatgrass; an alien invasive species introduced by early settlers, that is highly competitive and combustible. Cheatgrass thrives after disturbances such as fires and quickly replaced the native sagebrush altering the dominant plant species (Whisenant, 1989). Once an invasive species establishes itself, fire return intervals are altered by the new dominant species. Fire regime characteristics that have the potential to be altered are: frequency, intensity (based on fuel loads), seasonality and extent (Brooks *et al.*, 2004). Changes in fire regimes don't only affect vegetation and the landscape, but have a knock-on effect on the ecosystem, affecting all fauna and flora. The multiple effects caused by a change in fire regimes can complicate the ecosystem restoration or rehabilitation to pre-invasion conditions (Brooks *et al.*, 2004).

2.3.3.2 Commercial plantations

Forestry plays a large economic role in developing countries such as South Africa. Industrial/commercial plantations can be defined as evenly aged stands of exotic or invasive tree species outside of their natural habitat (de Ronde *et al.*, 2011). Since the introduction of plantation-based timber production to South Africa during the late 1800s, it has been recognised as a major land use and has been considered a critical component of catchment management schemes (Kraaij *et al.*, 2011). Fires are a threat to large scale plantations as the resulting loss of crop can amount to millions of Rands, hence fire management practices are aimed at fire hazard reduction (Fernandes & Rigolot, 2007; Kraaij *et al.*, 2011). This can be seen through much of the early legislation regarding fires and burning plans. However, fire management strategies aimed at hazard reduction in commercial plantations often conflict with ecological objectives that are set for the natural vegetation surrounding forested areas. Compromises intended to accommodate conflicting objectives may achieve neither unless clear, realistic management goals are set by either agency (Kraaij *et al.*, 2011). The main causes of fire in plantation forests are arson, escaped fires (planned and natural) and ignition by vehicles (Cameron *et al.*, 2007). As mentioned above, in South Africa and many other countries, it is common practice to burn annual fire breaks around plantations to prevent trees being affected by wildfires or escaped fires (O'Connor *et al.*, 2004).

2.3.3.3 Fuel loads

Fuel load is a term used to define the total dry weight of fuel per unit of surface area available for fires to consume after ignition. Fuel loads determine fire intensity as dry vegetation contains the potential energy that fires release during combustion (Stott, 2000). Fuel loads are classified by the structure of

the vegetation such as grassland, shrubland or forest fuels. There are further classifications, defined by the vertical arrangement of the fuels in the landscape (e.g., ground fuels, surface fuels and crown fuels), and these are known as 'layers'. Fuel type and layers determine the types of fires that can occur in a landscape (Bond & Keeley, 2005). Other factors to consider when evaluating fuel loads are the intrinsic and extrinsic properties of the fuels. Intrinsic fuel properties are characteristic of the plants themselves, and these properties primarily affect the fire frequency, intensity and seasonality components of fire regimes (Brooks *et al.*, 2004). Some of the intrinsic fuel properties of plants are chemical volatility and heat content of the plant tissue. These properties are related to the ease of ignition of the fuel and the amount of heat produced during combustion (DeBano *et al.*, 1998; Brooks *et al.*, 2004). However, a critical intrinsic fuel property to consider when assessing fuel loads and fires is moisture content. (DeBano *et al.*, 1998). The physiological condition of the plant, as well as the developmental stage, affect the moisture content of live vegetation. Comparatively, the moisture content of dead plant tissue is affected by the ratio between the total surface area of a fuel particle and the volume of the particle (classified by stem diameter). The moisture content of a plant is determined by external environmental factors such as relative humidity. The Tsitsa River catchment experiences dry winters with low precipitation as well as low temperatures, causing frost which causes a decline in vegetation moisture content, improving combustion potential and making fuels more susceptible to ignition (Brooks *et al.*, 2004; Mucina & Rutherford, 2006).

Extrinsic fuel properties relate to the way in which plants are arranged throughout the landscape, with densely vegetated areas offering higher fuel loads and sparsely populated areas. Bare patches can form natural barriers for fire where there is no vegetation to burn, either slowing or killing the fire (Bond & Keeley, 2005). Vegetation fragmentation can influence every aspect of a fire regime and includes the amount of fuel per unit area (e.g. hectares, km² etc.), and fuel continuity within specific sites and across the landscape. Fuel loads are of the primary determinants of fire intensity (Brooks *et al.*, 2004). This all depends on the type of fuel load and the properties of the fuel load as explained above.

2.4 The impacts of fires on the landscape and soil properties after exposure to fires

Fires alter the chemical, hydraulic and geophysical properties of soil depending on the intensity and severity of a fire event (DeBano *et al.*, 1979; Cawson *et al.*, 2016; Moody *et al.*, 2016; Magomani & van Tol, 2018). Soil physical properties change when organic matter is removed during a fire event (DeBano *et al.*, 1979; Laker, 2004). Soil organic matter (SOM) plays an important role by improving soil aggregation and structure by binding soil particles together and creating pores, allowing for improved aeration and water penetration. A study done by Hosking (1938) shows that humic acids in organic matter can be lost at temperatures below 100°C. At temperatures between 100°C and 200°C, non-destructive distillation of volatile organic substances occurred, whilst between temperatures of

200°C and 300°C, ignition of organic matter was evident, and above 450°C, organic matter was completely consumed. Intense ground fires, such as those in grasslands, will consume all organic matter at the soil surface, however, temperature will decrease with increasing soil depth. It was found that maximum temperatures during intense ground fires at 2.5 cm depth will destroy a large proportion of SOM (DeBano *et al.*, 1979; Doerr *et al.*, 2000). Moderate intensity fires will destroy most of the plant litter on the soil surface, and low-intensity fires can remove up to 85 % of the litter covering the soil (DeBano *et al.*, 1979; Doerr *et al.*, 2000). This has negative implications for soil protection through plant litter cover for erosive rainfall events soon after fire events.

Fires affect both soil properties and nutrient availability. The chemical properties most affected by fires are organic matter, pH, nitrogen, sulphur, potassium, divalent cations and cation exchange capacity (DeBano *et al.*, 1979; Mao *et al.*, 2016). When organic matter is destroyed by a fire, plant nutrients are released and become available in the soil medium for plant growth, or loss by erosion. The increased solubility of cations after fires is due to the slight increases in pH, particularly in the upper layers of soil and ash. Nitrogen has been found to be easily volatilized by heating at varying degrees depending on fire intensity (DeBano *et al.*, 1979; Doerr *et al.*, 2000). It has been shown that in intense fires with temperatures over 500°C, 100 % of the nitrogen found in plant and litter material can be lost and below 200°C, no nitrogen will be lost (DeBano *et al.*, 1979). This is only true for the top 2.5 cm of soil due to poor heat conductive properties. Although nitrogen may be volatilized during a fire, the remaining ammonia and nitrate are made highly available to other plants and micro-organisms (DeBano *et al.*, 1979). Both burning and natural biological decomposition release nutrients from organic matter. However, natural decomposition releases nutrients slowly over time compared to fires. This slow decomposition allows the nutrients to be taken up by other plants and microbes before being lost by erosion or seepage (DeBano *et al.*, 1979). Fire events quickly release nutrients which are highly soluble and the quantity of nutrients available after a fire depends directly on how much organic matter was burned (DeBano *et al.*, 1979).

2.4.1 Fire induced water repellency

Fires can decrease water infiltration in soils by creating a water-repellent surface layer in burned areas. Water repellency in soils affects the ability of moisture to penetrate certain layers of the soil profile; often the water-repellent layer is just below the wettable surface (Macdonald & Huffman, 2004). After a fire, a water-repellent layer is typically found within the top few centimetres of soil lying parallel to the surface. This layer can vary spatially, in thickness and in depth (DeBano, 2000; Certini, 2005; Cawson *et al.*, 2016). Fires also can strengthen an already existing water-repellent layer or destroy it. This has been viewed as one of the main causes of increased hillslope runoff and erosion following fires (Bodí *et al.*, 2013; Doerr *et al.*, 2006; Cawson *et al.*, 2016). Organic matter, which contributes to water repellency, accumulates in the surface litter during fire intervals when vegetation has the opportunity to grow. The organic matter mixed into the upper layers of soil decomposes and mixes with mineral

soils, sometimes forming a water-repellent layer (DeBano, 1981). Fire can rapidly change and intensify the water repellency in soils by intensely heating the litter and upper surface layers. Intense fires may heat the soil surface to up to 900°C, however the poor conductive properties of dry soil prevent the heat from being transferred to greater depths (DeBano, 1981; Cawson *et al.*, 2016). Heating of organic particles within the soil matrix causes them to chemically bond with and coat mineral particles, essentially forming a hydrophobic layer within the soil profile (DeBano, 1981).

The degree of water repellency can be measured by dropping water onto the surface of soil and timing how long it takes to be completely absorbed. This is known as the Water Drop Penetration Time test (WDPT). The drop of water may only sit on the soil surface for a short period of time before being absorbed, giving it a low water repellency (Doerr *et al.*, 1996; Dekker *et al.*, 2009; Rodríguez-Allereset *et al.*, 2012; Badía-Villas *et al.*, 2014). In a completely water repellent, non-wettable soil it may never be absorbed and instead evaporate or form runoff (DeBano *et al.*, 1967).

The inability for water to penetrate deeper into the soils can result in significant hydrological, geomorphological and ecological changes. Two of the effects of soil water repellency are modified evaporation from the profile and a reduction in infiltration into the soil. Reduction in infiltration often causes an increase in overland flow and in severe cases has been known to hinder vegetation growth as plant-available soil water decreases. Increased movement of water over the soil surface can cause an increase in erosion due to reduced surface roughness and binding roots (DeBano *et al.*, 1967; Doerr *et al.*, 1996). The degree of water repellency in soils may differ depending on the vegetation type and amount of organic matter in the soils, the frequency of fires that the soil has been exposed to, the temperature the fires reached and the amount of time the soil was exposed to fire (Scott, 2000). The duration of heating and the intensity of the fire play an important role in creating water repellent soils. It was found that soils experienced very little change to water repellency at temperatures less than 175°C, whilst repellency was increased at temperatures ranging between 175°C and 200°C. Further research reported that water repellency was lost or destroyed from between 280°C to 400°C (Cawson *et al.*, 2016). Soils heated to between 204°C and 260°C for 15 minutes intensified water repellency. However, the same soil heated to more than 371°C for more than 15 minutes destroyed the water repellency (DeBano, *et al.*, 1979). Similarly, Doerr *et al.*, (2004) found that sandy soils in eucalypt forests experienced maximum water repellency when heated for five minutes 250-280°C, whilst it was destroyed after heating it over 310°C for the same period of time (Doerr *et al.*, 2004). Although there is a strong relationship in the laboratory between soil heating and water repellency, this has not been extensively tested in the field. The few studies that have been conducted in the field have not observed the same results found in the laboratory. This research suggests that water repellency can be developed and lost over time depending on factors such as soil moisture, oxygen supply, vegetation type, soil texture and pre-existing water repellency. These variables can influence the relationship between soil heating and repellency (Doerr *et al.*, 2004; Cawson *et al.*, 2016).

2.4.2 Post-fire soil erosion and runoff

Soil erosion by overland flow is one of the most common types of erosion worldwide. It degrades soil productivity, water quality and can cause sedimentation of streams and water bodies (Blake *et al.*, 2005; Durán Zauzo & Rodríguez Pleguezuelo, 2008; Zhou *et al.*, 2008; Shakesby, 2011). . Soil erosion is a two-stage process that involves the detachment of particles by raindrop splash or overland flow of water, and the subsequent movement of the soil particles downslope (Florsheim *et al.*, 1991; Shakesby, 2011). The removal of vegetation by fires has been shown to be a cause of hydrological and geomorphological change in a landscape (van der Waal, 2009; Vieira *et al.*, 2015; Estrany *et al.*, 2016; García-comendador *et al.*, 2017). Plant cover controlling soil erosion has been studied in-depth and is widely accepted by the academic community (Durán Zauzo & Rodríguez Pleguezuelo, 2008; Zhou, 2008). During a rainfall event, vegetation controls erosion by protecting the soil surface against the impact of raindrops and intercepting surface runoff. The long term presence of vegetation improves soil stability and infiltration of water, thereby influencing water and sediment fluxes, slowing erosion rates (Durán Zauzo & Rodríguez Pleguezuelo, 2008). Alternatively, the change or removal of vegetative cover reduces the interception and infiltration of rainfall and sediments that would be trapped by litter, roots or plant structures. Other factors affecting post-fire soil erosion are altered physiochemical soil properties such as water repellency, soil structure, texture and particle size. Environmental factors also affect the erosion potential of a landscape, these are slope and climate (rainfall patterns and intensity) (De Luis *et al.*, 2003). As mentioned earlier, soil erosion potential is at its highest immediately after a fire disturbance as there is very little vegetation present to slow the movement of rain drops, water and sediment. Land management practices such as ploughing, terracing and logging exacerbate the problem as they involve heavy machinery and often displace topsoils, resulting in greater hydrological and erosive responses than those caused by fires (Vieira *et al.*, 2016; Malvar *et al.*, 2017; Figure 2.6).

Catchment flow responses to fires are variable due to the complex nature of landscapes and other controlling factors. Research suggests that the key driver of post-fire sediment production is the intensity and duration of rainfall events (Leopardi & Scorzini, 2015). It was found that high magnitude, low-frequency rainfall events have the biggest impact on erosion and sediment movement shortly after fires (Shakesby & Doerr, 2006). During high-intensity rainfall (100 mm h^{-1}), erosion is highly concentrated and large amounts of sediment are transported off hillslopes and down rivers and streams (Inbar *et al.*, 1998; De Luis *et al.*, 2003). De Luis *et al.*, (2003) found that sediment yields were significantly higher in burned areas than in unburned areas. However, soil loss in burned sites showed high variability and can be related to fire severity. Although short-duration high-intensity rainfall events cause excess runoff and erosion, it was also found that long-duration low-intensity rainfall events can also result in high sediment loads being produced and transported along rivers from recently burned hillslopes (Lane *et al.*, 2006). A study by Lane *et al.*, (2006) in south-eastern Australia found that annual flow increased by up to 70 % in the first two years in severely burned catchments. After two years the flow and sediment concentration in rivers decreased dramatically and were expected to return to pre-fire baseline levels. The decrease in sediment concentrations and flow over time can be accounted for

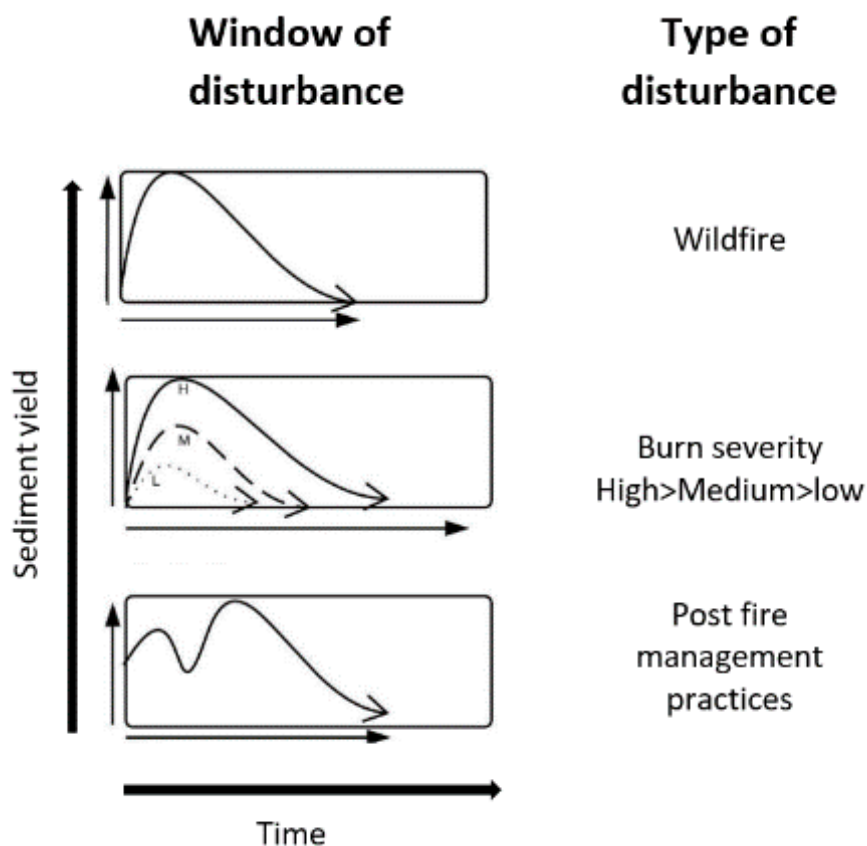


Figure 2.6: Post-fire erosion response to a single fire event, varying fire severity and post-fire management practices. Each peak represents a fire event (Adapted from Vieira *et al.*, 2016).

by the recovery of vegetation on hillslopes (Figure 2.7). These results may vary depending on the soil type, slope, climate, vegetation type and fire severity.

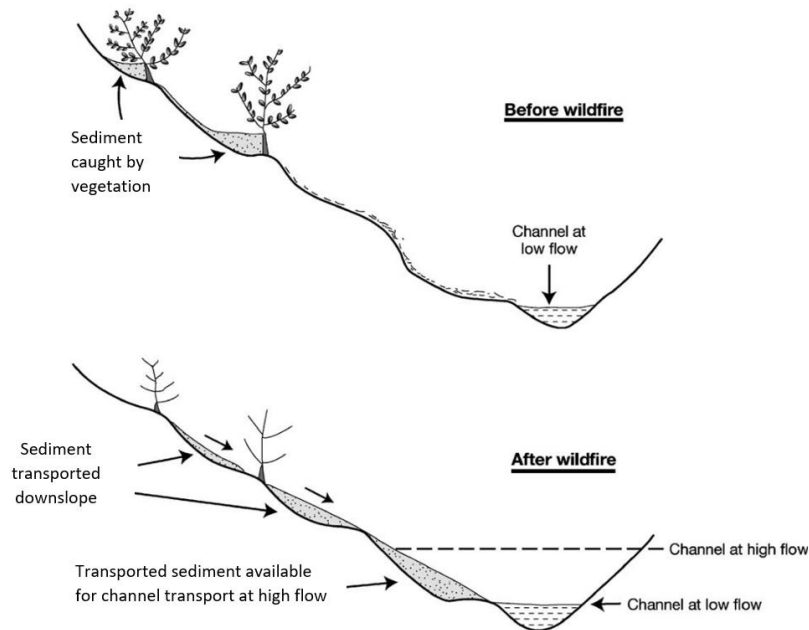


Figure 2.7: Sediment transport and availability down a hillslope (Adapted from Florsheim *et al.*, 1991).

Many studies have been done on hillslope processes after fires, however, assessing sediment yields post-fire at a catchment scale is extremely difficult. This is because only small parts of the catchment may be affected by fire and that sediment storage capacity and delivery at a catchment-scale is greater than at a hillslope-scale, causing sediment yield measurements to be inaccurate (Shakesby & Doerr, 2006). Although it is difficult to accurately measure sediment yield on a catchment-scale. Some research has found that fires can cause elevated sediment yields for a longer time period than expected. This was due to connectivity and sediment storage in and near stream channels which sustained high post-fire sediment yields for an extended amount of time (Shakesby & Doerr, 2006).

2.5 Using satellites to detect and monitor fires

Due to the disturbance fires can cause in many ecosystems and their effects on soils, vegetation and ecosystem functioning, accurate maps are essential in land management and risk assessments (Morgan *et al.*, 2001). In order to achieve this one needs to collect data on fire size, frequency, patterns and severity. There are multiple ways to undertake such a task, however, the most reliable and efficient method is to use remotely sensed data from satellites. Satellites provide an easy way to detect and monitor fires around the world and have been growing in numbers as the technology improves (Giglio *et al.*, 2008). Sensors fitted to different satellite platforms provide varying types of data and images, ranging from point fire data to infrared imagery. There are two different techniques that can be used to identify fires using satellites, and those are through the detection of burn scars and thermal signatures

(Hudack & Brockett, 2004; Giglio *et al.*, 2008; Hantson *et al.*, 2012). Burn scars leave distinct, short-lasting boundaries between fire-affected areas and untouched landscape (Johnson & Gutsell, 1994), whilst thermal sensors onboard satellites can identify active fire heat signatures which contribute to real-time fire detection (Joyce *et al.*, 2009; Hantson *et al.*, 2012). There are a plethora of satellites and techniques that can be used when detecting fires, and this section serves to identify and discuss some of the options available. Whilst burn scars and thermal sensors both provide reliable and accurate data; they serve different purposes and have different limitations (Bowman *et al.*, 2003; San-Miguel-Ayanz *et al.*, 2005) which will be discussed below and why they may be of best use for this study (Table 2.2).

Table 2.2: A Summary of some of the satellites and sensors used in hazard/fire mapping and monitoring for research and commercial purposes (Adapted from Joyce *et al.*, 2009). *Landsat 7 has problems with its scan line corrector resulting in data gaps in images.

Satellite	Sensors	Nadir spatial resolution (m)	Revisit capability	Data span	Fire detection method
Landsat 4-5	TM Multispectral	30			
	TM Thermal	120	16 days	1982-2013	Burn Scar
	ETM+ Panchromatic	15			
Landsat 7*	ETM+ Multispectral	30			
	ETM+ Thermal	60	16 days	1999- present	Burn Scar
	OLI Panchromatic	15			
Landsat 8	OLI Multispectral	30			
	TIRS	100	16 days	2013- present	Burn Scar
Sentinel 2a/2b	MSI	10-60	10 days	2015- present	Burn Scar
Suomi NPP	VIIRS	375	15 days	2011- present	Heat signature
Aqua/Terra	MODIS	250, 500, 1000	16 days	2001- present	Heat signature
Terra	ASTER	15, 30, 90	4-16 days		Heat signature
NOAA	AVHRR	1100	Daily	1978- present	Heat signature
Worldview 1-2	Panchromatic	0.46			
	Multispectral	1.85	1.1 Days	2009- present	Burn Scar
Worldview 3-4	Panchromatic	0.31			
	Multispectral	1.24	4.5 days	2014- present	Burn Scar
Quickbird	Panchromatic	0.6			
	Multispectral	2.4	1.5-3 days	2001-2014	Burn Scar
SPOT 5	Panchromatic	5			
	Multispectral	10	2-3 days	2002- present	Burn Scar
SPOT 6-7	Panchromatic	1.5			
	Multispectral	6	1 day	2012- present	Burn Scar

2.5.1 Efficiently detecting fires through thermal signatures (MODIS and VIIRS)

The Moderate Resolution Imaging Spectroradiometer (MODIS) was launched on the Terra platform late in 1999 as part of the National Aeronautics and Space Administration's (NASA) Earth Observation System (EOS), and at the time was the only global daily active fire product available (Justice *et al.*, 2002). The MODIS instrument collects data and produces several products which provide information on a global scale and can be used for practical applications (Justice *et al.*, 2002). Two products are

currently available, and these are the active fire detection which gives the location of live fires, and the burned area product which provides the extent of burn scars over a set period. The active fire algorithm uses multiple channels to detect thermal anomalies on a pixel per pixel basis (500 m x 500 m). As this algorithm detects thermal anomalies, high-temperature points such as gas flares and power plants will be detected too. However, this can be avoided using land cover mapping and removing unwanted points from the dataset (Justice *et al.*, 2002).

Visible Infrared Imaging Radiometer Suite (VIIRS) is another sensor which is often used to detect fires. VIIRS has a slightly better resolution than MODIS at 375 x 375 m. The VIIRS is aboard the Suomi-NPP satellite whilst the MODIS sensors are on the Aqua and Terra satellites (Viirsfire, 2018). They both have very similar orbital characteristics, return cycles and are sun-synchronous. Sensor differences which include resolution, swath width and pixel sizes result in slight differences in the expected fire observations. The standard VIIRS product is comprised of a list of fire-pixel point locations and is based on a version of the MODIS 4 collection (Viirsfire, 2018).

MODIS active fire detection is based on a contextual algorithm developed by Giglio *et al.*, (2008). The MODIS collection of archived data are reprocessed from time to time to allow for better calibration and algorithm refinements, resulting in more reliable data (Figure 2.8) (Viirsfire, 2018). Each active fire location represents the centre of a 1 km² pixel that is flagged by the algorithm as containing a fire within the pixel. This is seen as point data and can be used to map the movement and occurrence of fires (Viirsfire, 2018). This technology has been applied and is easily accessible from the Fire Information for Resource Management System (FIRMS), to provide near real-time active fire detection for managing natural resources. These records date back to the early 2000s and are easily accessible on the website (Viirsfire, 2018). An example of this technology being applied is on the Advanced Fire Information System (AFIS) website (McFerren & Frost, 2009). The system provides a tool to manage the risk of wildfires that may be close to high-value infrastructure such as plantations. The interactive site allows you to use MODIS and VIIRS data and shows near real-time fires on the ground using information collected from these sensors.

2.5.1.1 A case study comparison between the accuracy of MODIS and VIIRS sensors and fire detection algorithms

This is a small case study on several fires that burned in Idaho in early September 2012. Both the Suomi-NPP (VIIRS) and Aqua (MODIS) satellite sensors had a very similar timing in overpass and viewing angle, therefore a good comparison can be made between the two. Figure 2.9 shows and compares the two satellite sensors within four 2° x 2° cells, and the important information to take away from this is the white numbers above each cell. These represent and summarize the number of coincident detections between the two satellite sensors. The first number (N/x/x) is the number of VIIRS thermal detections in that specific cell. The second number (x/N/x) represents the number of VIIRS pixels with a coincident

MODIS fire detection, and the third number (x/x/N) is the number of MODIS thermal detections in the allotted cell (Viirsfire, 2018). VIIRS detected more fires than the MODIS sensor, and of the thermal anomalies detected, VIIRS detected almost all the fires that MODIS detected, whilst MODIS detected on average 20 % less fires than the VIIRS sensor.

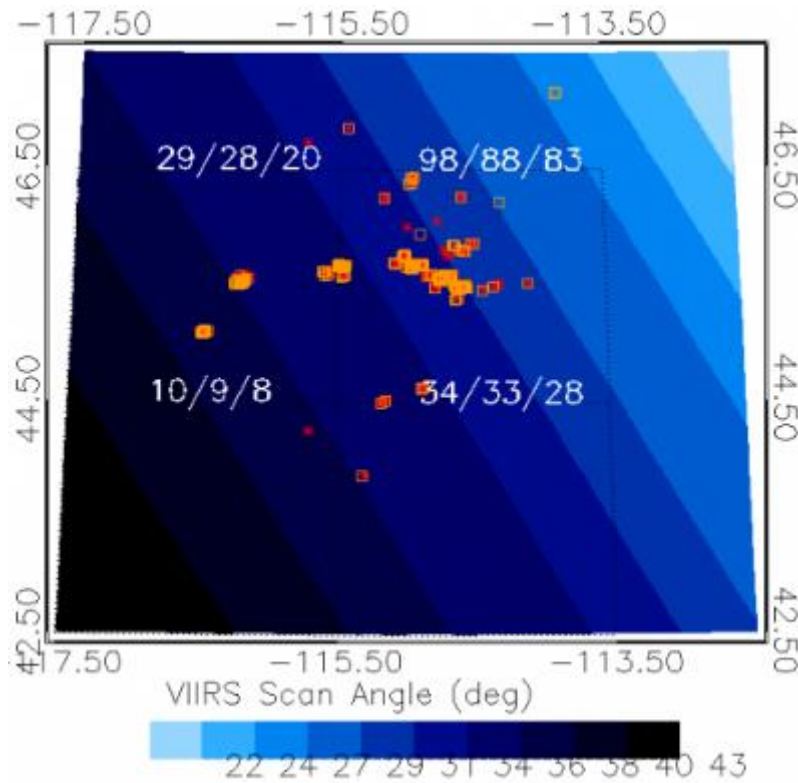


Figure 2.8: VIIRS vs MODIS fire detection comparison (Viirsfire.geog.umd.edu, 2018).

2.5.1.2 The limitations of detecting fires using heat signatures

The MODIS and VIIRS are some of the most reliable and freely available sensors that provide accurate and near real-time data on fire activity. However, they do not come without their limitations. Hantson *et al.*, (2012) conducted a study that compared MODIS fire hotspots to burned areas extracted from Landsat images. It was found that burned patches detected through Landsat and not detected by MODIS hotspots had a success of between 36 and 86%. However, this decreased with larger fires (up to 20% for burned areas larger than 500 ha). There was a linear relationship found between the number of hotspots and total burned area, and grasslands were found to have a larger mean burned area associated with each hotspot detected by MODIS (Hantson *et al.*, 2012). MODIS and VIIRS can only detect fires/thermal anomalies during the time of the satellite overpass, which means that hotspots are only a sample of the total number of fires occurring, whilst the detection of burned patches relies on reflection changes which remain present in the landscape for weeks to months (Hantson *et al.*, 2012). The temporal

limitations of hotspot detection are not the only downfall of MODIS and VIIRS; sensors may struggle to detect fires through thick clouds and smoke produced by fires, smouldering and misidentifying warm bare patches of land surrounded by contrasting green vegetation, limiting the use of the data (Justice *et al.*, 2002; Schroeder *et al.*, 2008). Whilst MODIS and VIIRS are useful for real-time fire data as in the case of AFIS, the sensors do not provide a detailed representation of the total burned area of a fire, and rather fill pixels where fire was detected whilst the satellite was overhead (Justice *et al.*, 2002; Schroeder *et al.*, 2008; Hantson *et al.*, 2012). Alternatively, multispectral imagery can be used to accurately distinguish burn scars in the landscape

2.5.2 Utilising multispectral imagery to identify burn scars

Remote sensing has been used to detect fire burn scars that can be used to calculate the spatial extent of fires. The availability of multispectral bands on many satellite sensors, namely the red, near-infrared and short-wave infrared bands are used to detect the difference between burned and unburned vegetation. Fire scar detection varies greatly depending on the characteristics of the scene, and some band combinations for Landsat 8 that have been found to be successful in studies are bands 4, 5, 7 by Chuvieco and Congalton, (1989) in Spain, bands 3, 4, 5 by Pereira and Setzer, (1993) in the Amazon and bands 4, 7, 1 by Koutsias and Karteris, (1998) in Greece. Landsat produces moderate-resolution images (30 m), which are ideal for burn scar identification at a catchment level as they provide enough accuracy to determine the spatial characteristics of fire regimes. A study undertaken by Hudak and Brockett, 2004, considered the spectral, spatial and temporal characteristics concerning fire scar mapping. Twenty-two annual fire scar maps were produced from 1972-2002 using historical data from Landsat imagery in the savannah landscape on the South African- Botswana border. Multispectral bands, spatial extent, temporal range, frequency and historical records of the Landsat platform allow it to be used to map fire scars in a variety of landscapes (Hudak & Brockett, 2004).

While fire scars may be obvious and easy to identify to the interpreter, mapping fire scars can be difficult as technology and automated classification techniques do not possess the pattern recognition capabilities we as humans do (Hudak & Brockett, 2004). Manual digitizing of burn scars was found to be more accurate than automated fire scar mapping techniques, however, the characteristic patchiness of fires makes this a subjective as well as a time-consuming job. Visual interpretation is also limited to single bands or three-band composites (RGB). However, since the launch of Landsat 7 and 8, image processing capabilities have improved, and it is uncommon to limit classification to three bands (Hudak & Brockett, 2004).

Satellite technology is advancing at a rapid rate, with smaller, less costly and higher resolution sensors coming onto the market constantly. A new product developed by Planet Labs launched in 2014 (Dove satellites) can collect a complete coverage of the Earth's entire land area, at a high resolution (3-5 m), compared to MODIS (1 km) and Landsat (15 m). The Dove satellites revisit areas daily and allow for

an almost real-time picture of the globe (Boshuizen *et al.*, 2014). This is possible as the satellites are released in 'flocks' of up to 50 small satellites that work together to collect imagery and add to the database (Boshuizen *et al.*, 2014). Dove satellites only weighing a few kilograms and occurring in low orbit, not more than 500km above the surface of the planet (Planet Labs, 2015). This new technology has potential to revolutionize fire monitoring remote sensing with constant high-resolution imagery of the entire globe.

2.6 Review of literature conclusions

Fire regimes are a complex part of many ecosystems all over the world, and the advancements in remote sensing and satellite technologies have allowed us to further understand and monitor their impacts on the natural and man-made environments (Justice *et al.*, 2002; Hudak & Brockett, 2004). Fires are an integral part of grassland systems in South Africa and play an important role in maintaining natural ecosystem functioning and are a useful tool for land managers. However, it is evident from the literature that was covered in this study that fires are a very controversial topic and there are positive and negative effects that fires have on the landscape, coming down to a variety of factors. It has been proven that fires do have adverse effects on erosion potential (DeBano *et al.*, 1979; Cawson *et al.*, 2016; Moody *et al.*, 2016; Magomani & van Tol, 2018), and can degrade landscapes, but there are also many positive influences fires have for commercial and communal land managers, preventing bush encroachment, clearing moribund grass, cycling nutrients and allowing new vegetative growth (Bond, 1997; O'Connor & Bredenkamp, 1997; Mucina & Rutherford, 2006; Kraaij *et al.*, 2011). The aim of this study is to assess the fire regimes in the UTRC, but the research also aims to evaluate the effects of fires, hopefully identifying if and how fires are negatively or positively influencing the landscape and add to the knowledge debate around the effects of fires.

3. Materials and Methods

3.1 Introduction

In order to fulfil the objective 1 and 2 of this study, extensive satellite imagery, and field data were collected and processed (section 3.2 and 3.3). Fire regime characteristics were determined using remote sensing and desktop analysis, whilst the effects of fires on the landscape were analysed using data collected from study sites within the catchment area. Landsat 5 and 8 imagery were accessed and processed in order to produce maps and results on fire frequency, size and extent whilst MODIS data were used to determine fire seasonality. Objective 3 (Determining the effects of fire frequency on soils) was accomplished by conducting field measurements and collecting soil samples to analyse in the laboratory, covered in section 3.4, 3.5 and 3.6.

3.2 Burn scar detection and extraction

Landsat imagery was used to identify burn scars within the landscape of the Tsitsa River catchment. The images collected were from the end of the burning season and were analysed in ArcMap 10 to provide an indication of fire frequency and extent. Fire frequency was expressed as the Fire Return Interval (FRI) and the extents of fires were calculated in kilometres squared (km²). Using a combination of point data and imagery, burn scars and Fire Return Interval can be calculated which gives an indication to the frequency of fires within the catchment. Landsat data were obtained from the United States Geological Survey (USGS) database (<http://earthexplorer.usgs.gov/>). The specified study area was selected, and filters used to identify images with dates ranging from August to November each year from 1980 to 2017. The ideal times to identify fire scars were arrived at visually and it was determined that the end of winter (burning season) was most suitable, before the summer rains and regrowth of vegetation. In some cases, cloud cover obscured the images and these images were excluded from the study. From 37 years of data, only 20 years of reliable imagery were collected from the Landsat 5 and 8 satellites. Landsat 7 data was not used as the imagery had been distorted by blank bands across the image due to the pathing of the satellite and was, therefore, deemed unusable. The specifics of the images can be found in Appendices 3 & 4.

Downloaded images came in a zipped file format and were unzipped using Winzip software which produced a file containing multiple .tif images, each with a specific frequency band from electromagnetic radiation captured by the satellite's sensor. The bands suitable for detecting fire scars were a shortwave infrared band (2.107-2.294 nm), a near-infrared band (0.851-0.879 nm) and a red band (0.636-0.673 nm). On the Landsat 8 platform these were bands 7, 5 and 4 and on the Landsat 5 platform bands 7, 4 and 3. Using ArcMap the relevant bands were compiled using the image analysis tool, after which the Landsat 5 images were mosaiced as the study area was located on the boundary of two images. This was accomplished using the Data management toolbox in ArcMap by copying the

raster images and removing the cells that contained no data. The two images could then be mosaiced to a new raster that covered the study area.

The Spatial Analyst tool (Segment mean shift) was used to classify and group pixels based on their spectral differences. Using the segmented image, an unsupervised classification process was run to further classify the image into based on their segmented classes. The image was then reclassified, grouping burnt and un-burnt pixels (Shao & Duncan, 2007; Gong *et al.*, 2013; Fox *et al.*, 2016). Further supervised reclassification was performed for incorrectly designated pixels, resulting in an accurate classification of burnt and un-burnt areas (Hudak & Brockett, 2004; Olivia & Schroeder, 2015). This allowed for a more accurate and reliable indication of the number and size of fires occurring each year, especially for years where the only available cloud-free image was several months after the fire season, e.g. November images. Once fire scars were identified, the image was converted to a vector layer using the Raster to polygon tool which created multiple burn scar polygons. The “burn scar” polygons were selected and extracted to produce a file which only contained the relevant polygons that were identified as burned areas for that year.

3.3 Detecting burn scars using Landsat imagery

Two different methods were used to determine the fire frequency of the Tsitsa River catchment, both utilising Landsat 5 and 8 remotely sensed imagery. These methods were compared to one another to contribute to the completion of Objective 1: What is the best way to detect the frequency and extent of fires in the study area using GIS and remote sensing?

3.3.1 Vector

Once the initial extraction of fire scars was complete there were a total of 20 years’ worth of data in the form of polygons with spatial data (Shape area). These data were used to create a fire frequency map, showing where fires occur and the number of times, they have occurred in the 20-year period observed. To undertake this task, the ArcMap geoprocessing tool ‘Union’ was used to create a single shapefile containing fire scar data from each year for which satellite imagery was collected. Once this was complete the spatial analyst tool was used to do a spatial join on the union file, utilising the same feature class as target class, and joined the features using the “join one to many” option (ESRI, 2019). The spatially joined file was then dissolved with the output file from the previous step. The features selected to dissolve were “TARGET FID” this was done by adding the “Join_Count” by selecting “SUM” in the statistics column. The resulting output attribute table contained a new field which was created during the spatial join and was named “Join_Count”. The field calculator was used to divide the number of spatial joins by the number of years (20), which showed the number of times polygons from separate years overlapped. This produced a fire frequency map which once complete can be altered using the symbology and a graduated colour scale, utilising the join count as the defining category.

3.3.2 Raster

Another technique that was used to show the fire frequency in the catchment was analysing raster data. This data was created during the process of burn scar extraction and is the layers of burn scars before they were converted to vector data. It is a simpler method and results can be achieved in very little time. Extracted burn scar raster layers were combined using the “Cell statistics” tool in ArcMap and the statistics type used was “SUM” as all the cells had a value of 1. This produced a frequency map showing where fires had occurred and how many times they had occurred within the 20 years of study. This is because with each cell having a value of 1, adding the multiple layers together resulted in a value which represented the number of times a fire had burned that specific cell. E.g. If a fire had occurred in a specific area 13 out of 20 years, the cell value would be 13.

3.4 Using satellite sensors to detect fire seasonality and intensity through thermal signatures

An alternative method for detecting fires using remotely sensed imagery is utilising satellite sensors (MODIS and VIIRS) which are on the Aqua and Terra platforms. These data were collected from the Fire Information for Resource Management System (FIRMS) database <https://firms.modaps.eosdis.nasa.gov/download/>. These data are point data with spatial and temporal properties which can aid in the identification of burn scars and time of fire events. This was not used to determine fire frequency as one fire can produce many points, as well as the fact that the satellites only detect thermal anomalies whilst they are passing overhead which occurs once a day between 08h00 and 14h00 due to the satellite being sun-synchronous. However, the burn scars will be detected in the Landsat Imagery as mentioned above. Due to the spatial and temporal properties of the data, we were able to extract the time, date of acquisition and FRP which is measured in Megawatts per pixel (500m x 500m). This aided in understanding when, where and at what intensity fires occurred, adding to the greater understanding of fire regimes. Using a combination of ArcGIS and QGIS, data from 2000-2018 were used to produce accurate seasonality graphs and fire intensity maps of commercially farmed and communal areas within the Tsitsa catchment.

3.5 Calculating fire size

To calculate the size of fires, burn scars were identified and extracted from remotely sensed Landsat imagery (mentioned above for fire frequency). The burn scar data vector files were analysed using a simple ArcGIS statistical analysis, calculating the total area for each polygon (burn scar) which could be exported into Microsoft Excel for statistical analyses.

3.6 Field methods

Soil samples were collected from sites in the UTRC from predetermined sites identified using fire frequency maps that were generated using Landsat data and ArcMap (see Section 3.3). Areas with a high (greater than 10 fires over 20 years – $FRI < 2$) and low (less than 5 fires over 20 years – $FRI > 4$) fire frequency (Govender *et al.*, 2006; Archibald *et al.*, 2010), were identified on maps and study sites

were selected using a buffer tool, ensuring road accessibility (500 m distance from roads). The fire frequency maps were used to select random sample sites of, keeping within predetermined parameters (mudstone and sandstone on north facing slopes within 500 m of the road). There were 7 different categories of sites selected and they are listed as follows: High fire frequency on sandstone, high fire frequency on mudstone, low fire frequency on sandstone, low fire frequency on mudstone, areas that burned in 2017 on sandstone and mudstone, as well as areas that burned recently (2018) on mudstone and sandstone (Table 3.1). Ten sites were selected at high (1-3 year FRI) and low (3> year FRI) fire frequencies on sandstone and mudstone to test the influence of fire frequency, and five sites were selected on mudstone and sandstones that burned in 2017 and in 2018 to determine if soil properties recover post-fire.

Table 3.1: Study site names and descriptions

Site name	Number of sites	Description
Effects of fire frequency on soil characteristics sites		
HS1-10	10	High frequency of fires on sandstone soils
LS1-10	10	Low frequency of fires on sandstone soils
HM1-10	10	High frequency of fires on mudstone soils
LM1-10	10	Low frequency of fires on mudstone soils
Post-fire soil recovery sites		
2017S1-5	5	Fires burned in 2017 on sandstone soils
2017M1-5	5	Fires burned in 2017 on mudstone soils
RS1-5	5	Recent fires occurring on sandstone soils
RM1-5	5	Recent fires occurring on mudstone soils

3.6.1 Desktop analysis: selecting sampling sites

Sample sites were randomly selected using the burn frequency raster data. A shape file with the main roads within the catchment was used to create a buffer zone which was easily accessible for data collection. The fire frequency raster data were clipped using this buffer zone and converted to point data. The point data attribute table contained information on the number of times fires had occurred in that area. A new float field was created in the attribute table and was populated using the field calculator. The *parser* was changed to python and the following code was inserted into the *Pre-Logic Script Code* text block:

```
import random
def rand():
    return random.random()
Rand()
```

was typed into the Random text box and the new field was populated with random numbers. Once the new field had random values, the *Select by Attribute* function was used under *Table Options*. The following expression was typed into the text box where *layer* was the name of the new field created random numbers and *num* is the percentage of features selected.

“layer”<num

The function mentioned above allowed for the random selection of points within the study area that could be used. However, these points had no spatial information, and this was added to the data layer by using the “Add XY coordinates” tool in ArcMap. The tool added spatial properties to the attribute table in the form of X and Y coordinate columns. These columns were then converted to decimal degrees by calculating their geometry in the attribute table which gave us exact coordinates of the points. These points were then converted to .gpx format and sent to a handheld GPS for field use.

Due to the complexity of fires, samples were only collected from high and low-frequency burn sites, as well as recently burned sites to assess what impact fires have after fires have burned. In order to control variables of the selected sites, further screening was completed on ArcMap 10.0. This included identifying the aspect and geology (EGIS, 2019) of the potential sampling sites. Only north-facing slopes were assessed due to time limitations, as well as sites that fell within the sandstone and mudstone lithology. The spatial join tool in Arc was used to identify north- and south-facing slopes within the selected geologies. Random sampling points were overlaid onto the areas that met both criteria and those sites were selected for sampling (Figure 3.1).

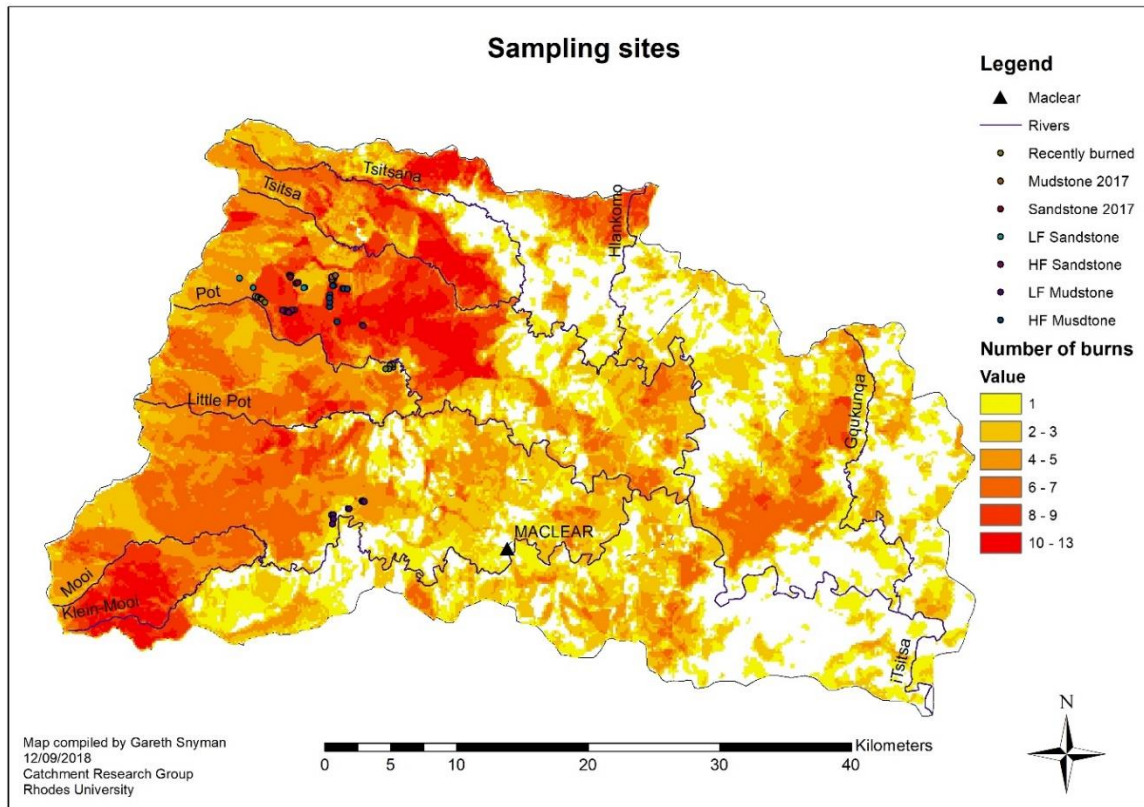


Figure 3.1: Selected sampling sites based on fire frequency and geology.

3.6.2 Sampling design

At each sampling site, a 5 m x 5 m quadrat was laid out using a tape measure, rope and marker poles (Figure 3.2). Readings for crust hardness, biomass and basal cover were taken at each corner and the centre of the quadrat (Figure 3.3). Soil water repellency measurements were taken at the surface at the centre of the quadrat. A soil sample was also taken from the centre of the quadrat to do further analysis and tests in the laboratory. The sampling design for the field samples was unique as the spatial resolution of Landsat 5 and 8 imagery is 30 m, which limited the way in which samples could be taken. A handheld GPS that was used was accurate by up to 5 m which was determined and tested beforehand, meaning that anything larger than a 5 m x 5 m quadrat had the risk of falling into another pixel identified by the Landsat imagery which would have skewed the results. Another factor to consider when designing the sampling method was the topographic nature of the catchment. Some of the sites were located on steep slopes that did not suit a larger quadrat or a transect, as pixels may cover a steep slope or cliff. The benefits of sampling at five points within the 5 m x 5 m quadrat allows for good coverage of data.



Figure 3.2: An example quadrat set up and soil cores being taken with gouge corer during fieldwork.

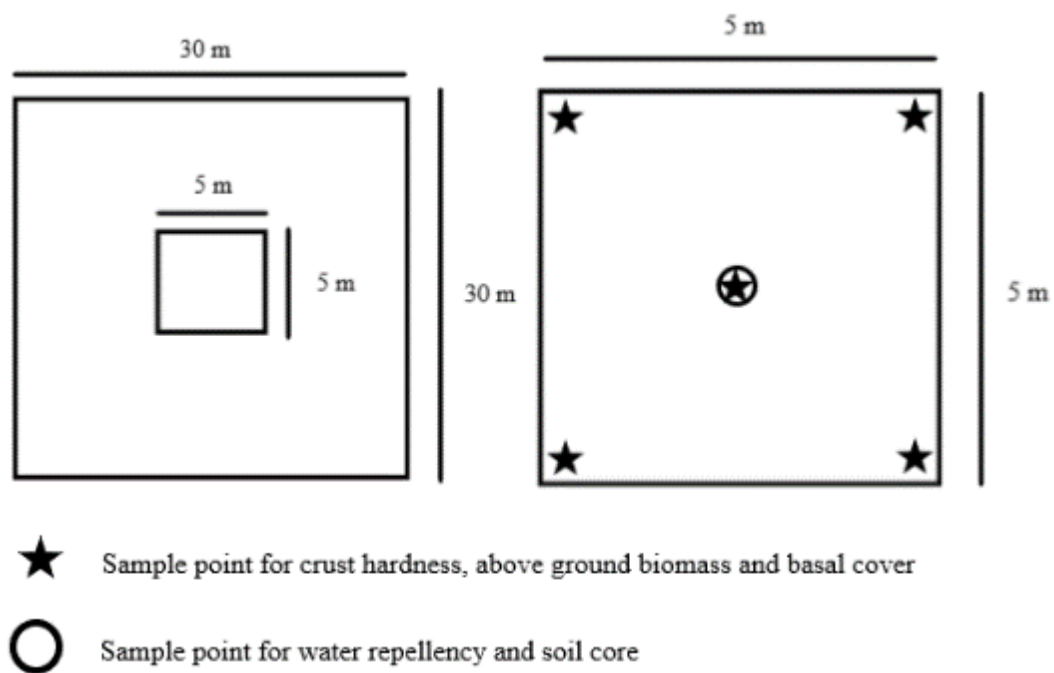


Figure 3.3: Quadrat layout within 30m pixel with sampling points.

3.6.3 Field soil analysis

This section serves the purpose of explaining all methods and measurements that took place in the field and hence classified as field analysis rather than laboratory analysis which was performed on soil samples collected during fieldwork.

3.6.3.1 Infiltration/repellency

The method used to assess soil hydrophobicity or water repellency is the Water Drop Penetration Time WDPT test (Doerr *et al.*, 1996; Steenhuis *et al.*, 2001; Dekker *et al.*, 2009; Rodríguez-Allereset *al.*, 2012; Badía-Villas *et al.*, 2014). The test requires an eyedropper, deionised water, a stopwatch and an undisturbed soil surface. Deionised water was applied to the soil surface using the eyedropper from a height of 1.5 cm (Figure 3.4). The time taken for the water drop to infiltrate the soil surface is used to quantify the water repellency of the soil (Table 3.2). The test was performed in each corner and centre of the quadrat to calculate an average water repellency for each site and to detect variations within the sampling area. Water repellency for below the surface was measured using a gouge corer and dropping distilled water on the cores at 1cm intervals to a total depth of 20 cm (Dekker *et al.*, 2009).



Figure 3.4: Water Drop Penetration Time test being carried out in the field at a recently burned sampling site.

Table 3.2: Degrees of soil hydrophobicity measured in seconds (Adapted from Steenhuis *et al.*, 2001).

Class	Water infiltration time (s)	Severity
0	0-5	Wettable
1	5-60	Slightly water repellent
2	60-600	Moderately water repellent
3	600-3600	Severely water repellent
4	>3600	Extremely water repellent

3.6.3.2 Soil surface hardness

A 1 cm² hand-held soil surface penetrometer was used to measure the soil crust hardness. Five measurements were taken at each sampling site, one in each corner and one in the centre of the quadrat. All measurements were recorded in kilopascals (kPa) (Malvar *et al.*, 2017). The crust penetrometer was set to 0 and firmly pressed into the soil. Once the tip of the penetrometer broke through the soil surface, pressure applied was stopped and the reading was recorded.

3.6.4 Measuring above-ground biomass

Measuring fuel loads was accomplished with a Disk Pasture Meter (DPM), which measures above-ground biomass (Bransby & Tainton, 1977; Trollope & Potgieter, 1986). The DPM was used to take five measurements in each quadrat. Readings from the DPM were then converted using a calibration table (Appendix 6) used to calculate the above-ground biomass per hectare of veld (kg ha⁻¹) (Zambatis *et al.*, 2006).

3.6.5 Ground truthing

Ground truthing was done in the field during early October of 2018, which falls into the end of the burning season, to verify whether fire scars identified during the desktop analysis of satellite imagery were accurate. Ground truthing was done by observation, looking specifically for signs of disturbance by fire such as ash, burned vegetation and post-fire regrowth compared to the unburned landscape.

3.7 Soil analysis from collected samples (Laboratory work)

Soil samples that were collected at each site in the field were brought back to the laboratory for analysis and underwent a standard soil preparation procedure. The samples were air-dried in an oven at 50°C for 48 hours. Aggregates were broken up using a mortar and pestle and sieved through a 2 mm sieve (Darby, 1980; Armstrong & Rowntree, 1992). This process was used to prepare all soil samples for textural analysis, pH and weight loss on ignition.

3.7.1 pH

To determine the pH of the soil samples collected in the field, approximately 20 g of soil was weighed out on a two decimal point balance scale and placed into a 100 ml beaker, after which, 50 ml of distilled water was added. The sample was stirred vigorously with a glass rod, ensuring the sample was completely wet and well mixed. The sample was then left for 50 minutes to reach equilibrium and was stirred again. The sample was left a further 10 minutes before it was vigorously stirred, and the pH probe was placed in the solution. The solution was continuously stirred whilst the probe was submerged for a more accurate measurement (Armstrong & Rowntree, 1992). The temperature of the solution was also measured as there is an inverse relationship between temperature and pH and the temperature of the solution was not constant. The pH readings were adjusted according to the temperature recorded using a calibration table.

3.7.2 Soil organic matter

Loss on ignition is a technique used to determine the amount of organic matter contained in the soil samples collected during fieldwork. The procedure is as follows (Darby, 1980; Armstrong & Rowntree, 1992): small porcelain crucibles were heated at 105°C for two hours and left to cool in a desiccator until room temperature. The porcelain crucibles were weighed on mass balance at a resolution of 0.0001 g (W1). Pre-prepared soil samples were then placed in the crucibles and reweighed to obtain the weight of the dry sample and the weight of the crucible (W2). Crucibles were placed on trays in a specific order with identification numbers to prevent the mixing of samples. The tray was then placed in a furnace at 450°C for 24 hours (Darby, 1980; Armstrong & Rowntree, 1992). After 24 hours the soil samples in the crucibles need to be removed from the furnace and placed in a desiccator to cool to room temperature. Each crucible was weighed after cooling down on a mass balance at a resolution of 0.0001 (W3). This final measurement will calculate the amount of organic that was lost in the soil samples whilst in the furnace. All weights were recorded in a table and the calculations were done in Microsoft Excel (Table 3.3). The equation used to calculate the percentage mass lost on ignition used was:

$$\% \text{ Organic matter} = \frac{(W2 - W3)}{(W2 - W1)} \times 100\%$$

Table 3.3: An example of the table and process used to record loss on ignition measurements.

Sample ID	Crucible (g) (W1)	Crucible + dry sample (g) (W2)	Crucible + Sample after ignition (g) (W3)	Soil		
				mass (g)	Mass lost (g)	Mass lost %
HM1	18.1555	32.0308	31.3493	13.8753	0.6815	4.91%

3.7.3 Textural analysis- determining soil particle size

A textural analysis was performed on the soil samples collected from each site. Soil texture was determined using the soil texture triangle (Brown, 1998; Whiting *et al.*, 2002). To identify the soil texture, a particle size analysis using a sieving technique was performed following Gregorich & Carter's (2007) guidelines. Soil samples were disaggregated with a mortar and pestle until the clumps had been broken down. The sample was then weighed to record the total mass and placed into the top of the sieve stack. Sieves were stacked on top of one another starting with the largest aperture down to the smallest (2 mm, 1 mm, 0.5 mm, 0.25 mm, 0.125 mm, 0.063 mm), with a base pan to catch the finest particles. Individual empty sieves were massed before the samples were added and this mass was used in calculations for all the samples. The sieve stack was then placed on a vibrating sieve shaker for 15 minutes. Once the 15-minute cycle was complete, each sieve was massed to two decimal points and the measurements recorded. The empty mass of the sieve was subtracted from the mass of the sieve, plus the sample to calculate the fraction of the sample caught (Table 3.4). This was then used to calculate the makeup of the soil samples using the soil texture triangle.

Table 3.4: An example of how the measurements for the soil sieve particle size analysis were recorded.

Sample	Empty sieve (g)	Example 1 Sieve + fraction (g)	Sample mass Fraction (g)	220.29 g Percent of total (%)
2 mm	405.47	405.53	0.06	0.03
1 mm	369.3	373.59	4.29	1.95
0.5 mm	300.47	321.88	21.41	9.72
0.25 mm	278.7	326.22	47.52	21.57
0.125 mm	254.38	312.55	58.17	26.41
0.063 mm	262.88	308.75	45.87	20.82
Base	242.86	285.83	42.97	19.51
Total			220.29	100

3.8 Statistical analyses

Data were analysed by Analysis of Variance (ANOVA) using Genstat 19 and Statistica 13.4. ANOVA tests were performed at a 95 % confidence level were used because of its ability to determine whether there are any statistically significant differences between three or more independent variables. In this study ANOVA proved useful as there were more than three variables being tested against one another. E.g. fire frequency (high or low) was compared to soil type (mudstone or sandstone) and a variable that was measured, such as soil surface hardness. In some instances, skewed data were log₁₀ transformed to allow for a normal or near normal distribution, but original data are presented in the results section. Where significant differences were observed, Fischer's protected Least Significant Difference (LSD) was applied. A Principal Component Analysis (PCA) was performed on all the data. Where appropriate, a regression analysis was performed and R² values were presented. The above statistical techniques

aided in identifying whether soil properties are affected by fire frequency and were crucial in completing Objective 3.

3.9 Summary

In summary, the methods above describe how Objective 1 and 2 were accomplished. This was by collecting and processing remotely sensed data from Landsat, MODIS and VIIRS. Study sites were selected according to pre-determined exposure to fires (fire frequency), and were mapped according to accessibility, geology and aspect. Soil properties (WDPT, soil surface hardness, slope, above ground biomass) were measured at selected study sites and soil samples collected for further analysis in the laboratory (pH, LOI, particle size). Statistical analyses were performed on the soil measurements which address Objective 3 and are presented below in the results section.

4. Results

4.1 Introduction

The following section describes what key fire characteristics (seasonality, frequency, size and intensity) were found in order to determine the type of fire regimes present in the UTRC. Following the description of fire regime characteristics, statistical analyses and results of the effects of fire frequency on soil properties are presented, as well as details of soil recovery post fire.

4.2 Fire regime characteristics

Fire regimes are defined utilising key characteristics; i.e. fire frequency, season, intensity and size. Fire frequency and size were determined by extracting burn scars from Landsat images, while seasonality and fire intensity were measured using a combination of MODIS and VIIRS satellite sensors (Figure 4.1). The flow diagram below displays an outline of the processes that took place in order to assess the key components of a fire regime as defined by Archibald *et al.*, 2010. The data were analysed using a combination of ArcGIS, QGIS and Microsoft Excel. The UTRC is approximately 2000 km² and has been split into two land management groups for the purpose of this study (Commercially and communally managed land). Detecting fire seasonality via satellite sensors

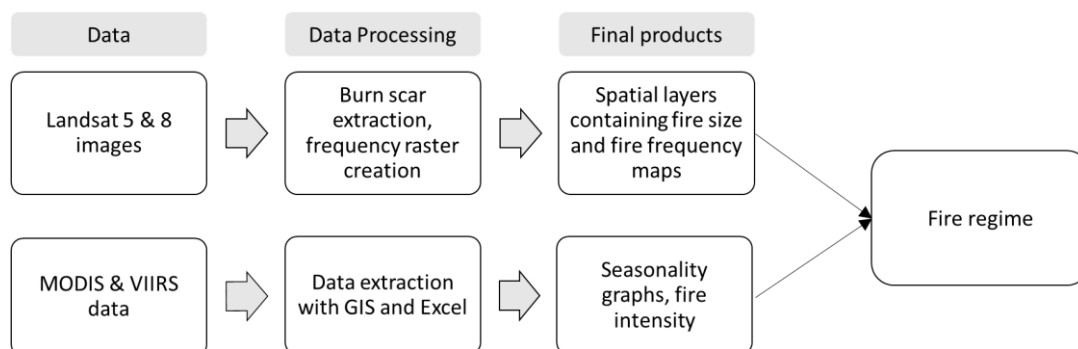


Figure 4.1: A depiction of the data source, processing and products for determining fire regimes in the UTRC.

Fire seasonality was determined using data collected from MODIS and VIIRS satellite sensors and analysed using QGIS date/time maps. The commercially managed area of the catchment is larger than the communally managed portion of the catchment, approximately 1227 km² compared to 770 km² respectively, with more fires occurring annually on the commercial land (Figure 4.2). It is also evident that a difference exists between burning seasons in the commercial and communal areas of the catchment. The burning season for the communally farmed portion of the catchment starts in June, peaking in July and ending in September, with a few fires in October and November (Figure 4.3). Fires in the commercially farmed area of the catchment seem to start as early as March and the fire season carries on through until November, with the peak of the season occurring in August (Figure 4.4). The circular heat maps (Figure 4.3 & 4.4) show that the commercially farmed areas have experienced a greater number of fires than the communally managed areas. This is shown in the legend which indicates that over 19 years, 20 fires have occurred on a specific day compared to the 13 of the communal areas. It is also evident that the highest number of fires occur during the dry, winter period (June- October).

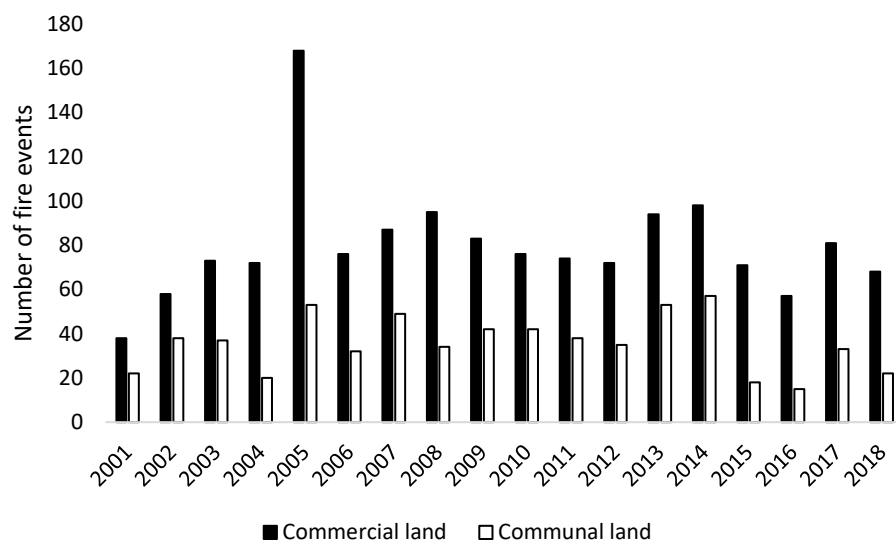


Figure 4.2: Fire events between commercially and communally managed land detected by MODIS and VIIRS between 2001 and 2018.

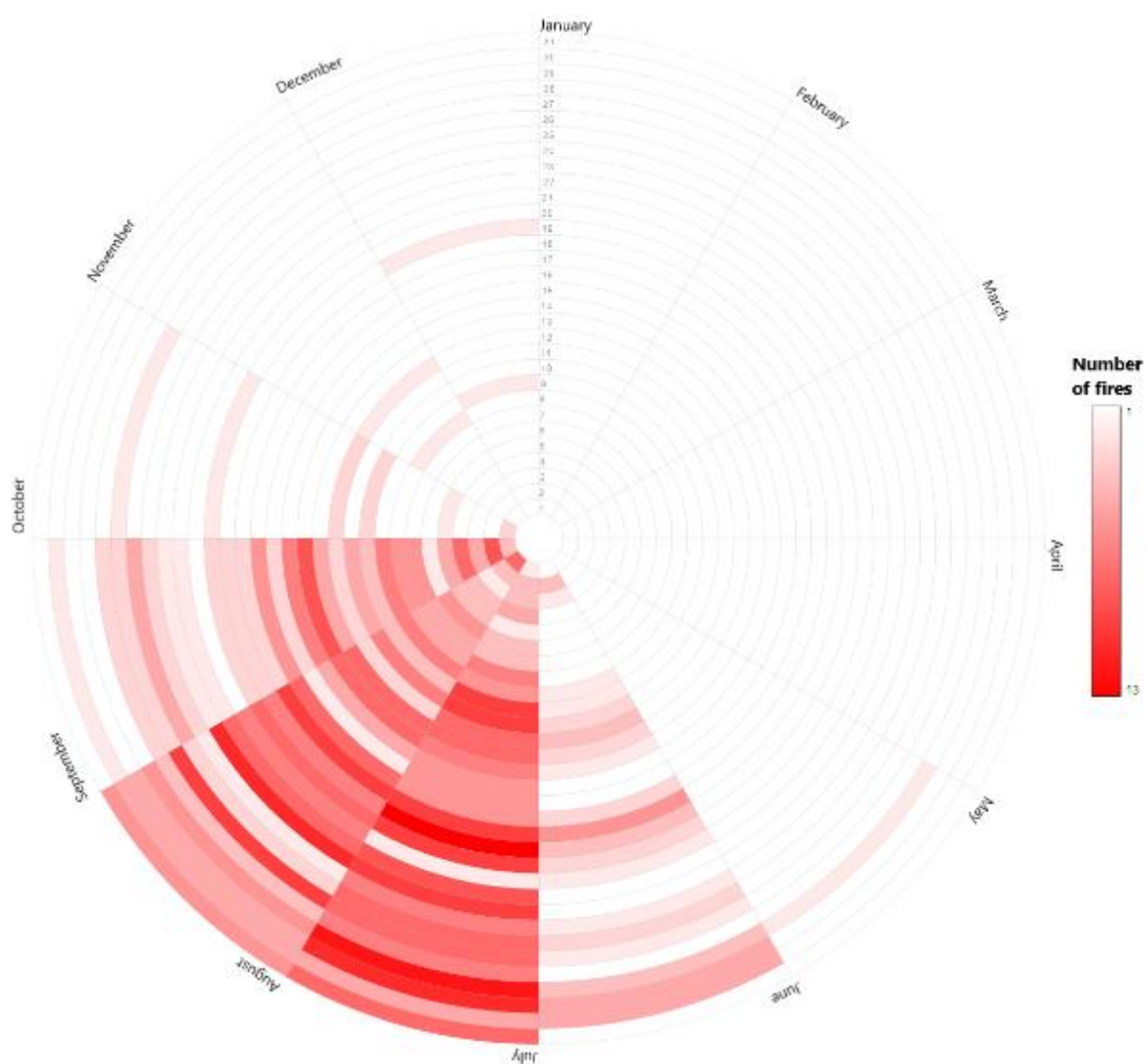


Figure 4.3: Heat map showing fire seasonality in communally managed grasslands of the UTRC.

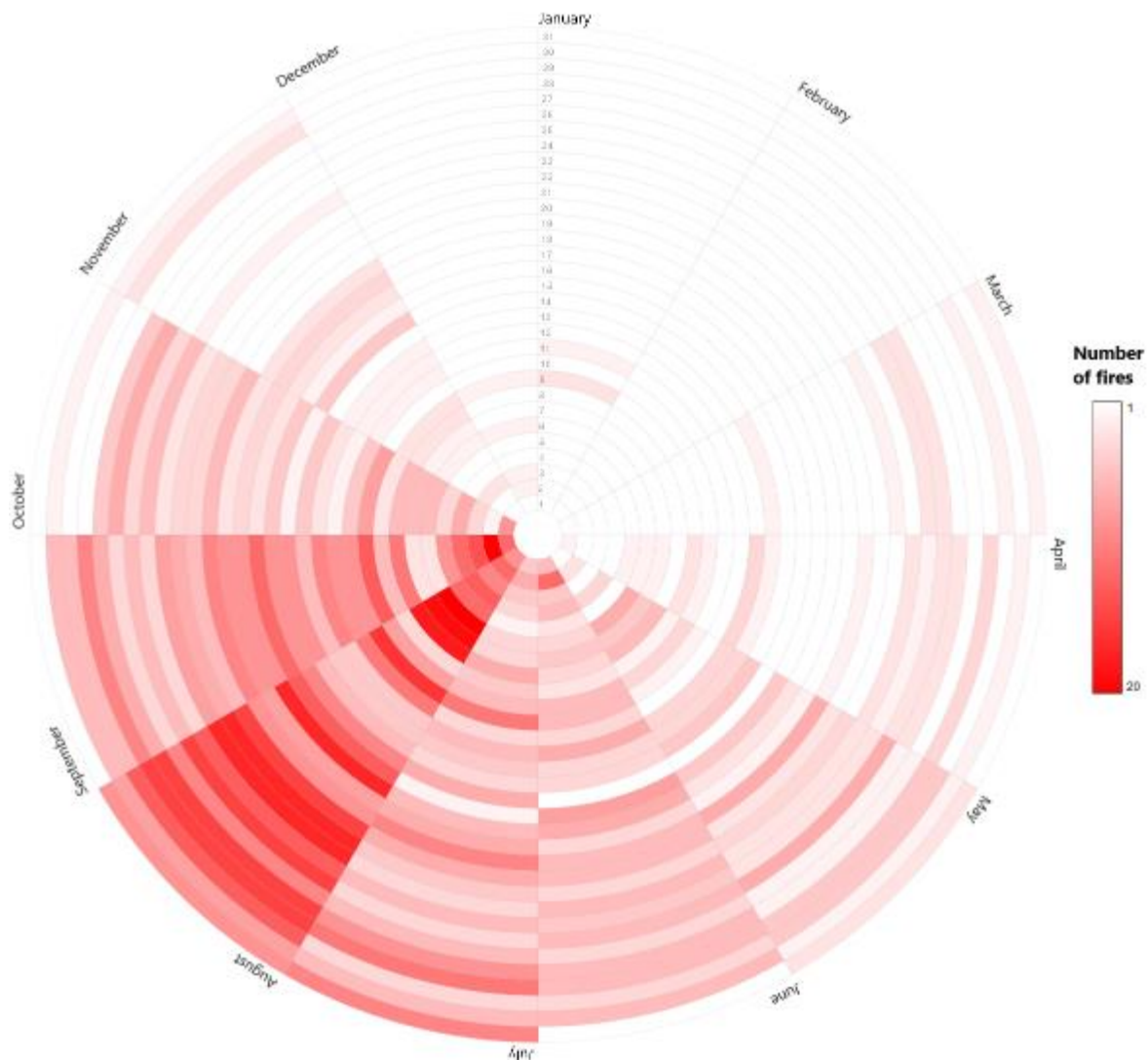


Figure 4.4: Heat map showing fire seasonality in commercially managed grasslands of the UTRC.

4.2.1 Comparing the spatial extent of fires in commercial and communally managed land within Tsitsa River catchment.

The spatial extents of fires were determined using Landsat imagery which was processed using ArcGIS to produce burn scar layers for a total of 20 years between 1984 and 2018. Only 20 years of data were used as some imagery was obscured by cloud cover, or cloud-free data were not available. As mentioned above, the commercially managed land was approximately 1227 km² compared to 770 km² of the communal area. The total annual area burned was calculated for the commercial and communal areas (Table 4.1), however, due to the difference in size of the two land management areas, the overall percentage of the burned area was used to compare the two areas (Figure 4.5). On average, the total burned area and percentage area burned was 13 % higher in commercially farmed areas than the communal areas.

Table 4.1: A comparison of the commercial and communal annual burned area and % burned area from years where data were collected (1984-2018).

Year	Commercial annual burned area km ²	% burned	Communal annual burned area km ²	% burned
1984	545,6	44,5	111,3	14,5
1989	556,4	45,3	111,0	14,4
1991	279,8	22,8	105,9	13,7
1992	457,7	37,3	72,4	9,4
1994	487,2	39,7	34,2	4,4
1995	332,8	27,1	117,3	15,2
1996	156,5	12,8	42,8	5,6
1997	479,7	39,1	149,7	19,4
1998	250,6	20,4	102,4	13,3
1999	238,8	19,5	194,6	25,3
2000	509,1	41,5	37,0	4,8
2001	207,4	16,9	176,5	22,9
2005	264,1	21,5	70,3	9,1
2006	220,9	18,0	151,1	19,6
2007	316,3	25,8	109,1	14,2
2013	226,2	18,4	76,3	9,9
2014	172,9	14,1	50,6	6,6
2015	107,5	8,8	43,8	5,7
2017	150,6	12,3	53,2	6,9
2018	199,0	16,2	34,9	4,5

On average, commercially managed areas experienced larger fires the communally managed areas, however findings show that fires in communal areas increased in size in the early 2000's (Figure 4.5). However, over the past 30 years the annual burned area across the catchment has decreased.

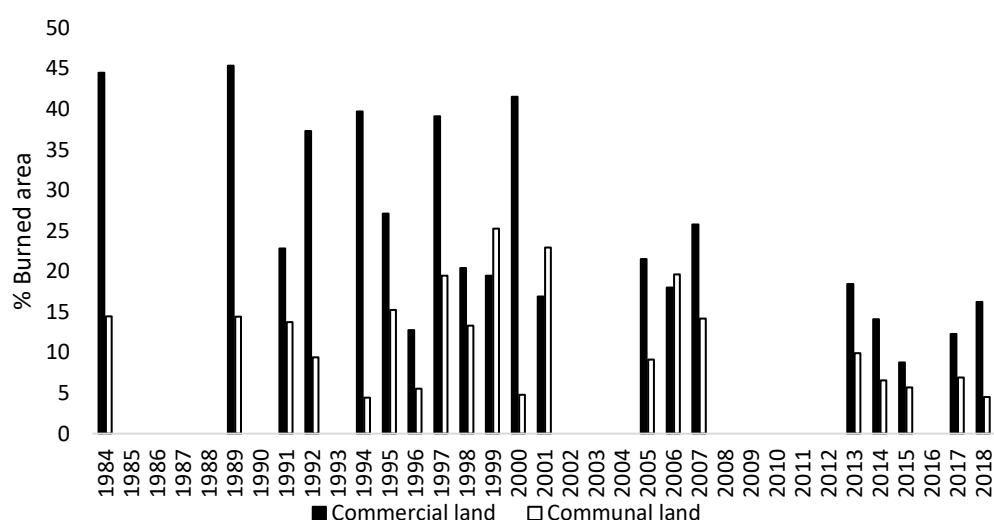


Figure 4.5: A visual representation of the percentage area burned annually, extracted from Landsat 5 and 8 imagery.

A change in total annual burned area is a result of a change in fire size. Fire size is determined by fire scar size, and over the monitoring period, there was a decrease in maximum burn scar and average burn scar size (Table 4.2).

Table 4.2: A summary of the change total burned area and maximum and average burn scar size.

Year	Total burned area km²	Maximum burn scar size km²	Average burn scar size km²
1984	657	388.37	16.84
1989	667.48	360.04	16.28
1991	133.95	24.43	3.72
1992	530.35	193.32	16.57
1994	522.29	275.46	14.12
1995	450.17	132.2	10
1996	160.75	106.95	6.43
1997	626.62	277.19	10.27
1998	30.46	8.79	2.03
1999	433.51	183.91	4.93
2000	546.41	296.26	10.31
2001	383.01	114.35	3.84
2005	301	143.01	5.01
2006	102.66	31.77	6.03
2007	425.47	229.51	10.91
2013	302.64	96.58	6.3
2014	90.41	29.85	4.11
2015	125.82	28.91	2.46
2017	62.2	19.91	2.39
2018	220.53	43.32	1.83

4.2.1.1 The influence of rainfall and climatic cycles on fire scar size

The total annual rainfall for Maclear was compared to the average change in Sea Surface Temperature (SST) which is indicative of the changing climatic cycles (ENSO). ENSO influences local climate and in turn the extent of fires, as rainfall directly determines the available fuel load. No direct correlation between annual rainfall and the change in average SST due to ENSO was identified (Figure 4.6).

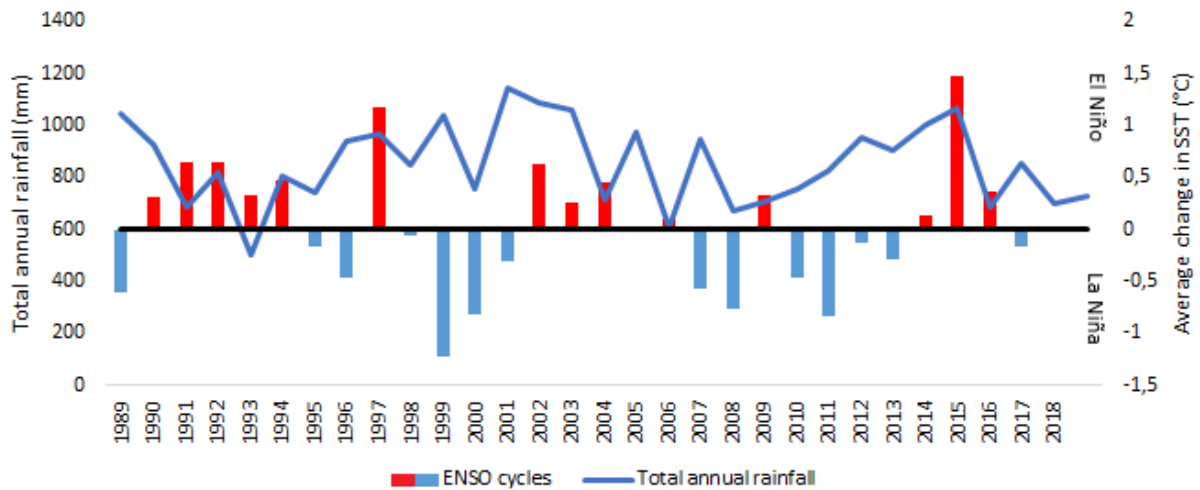


Figure 4.6: A comparison between total annual rainfall in Maclear and changes in average SST(°C), correlated to ENSO, between 1989 and 2018.

Rainfall is known to strongly influence fuel loads and fire size; therefore, total burned area was compared to total the annual rainfall for Maclear to determine whether there was a relationship between the two (Figure 4.7). Trendlines show that annual rainfall is relatively constant, whilst annual burnt area is decreasing over time. This result was not expected and suggests that there is no conclusive relationship between the two over a prolonged period. However, there is a general decrease in total annual burned area in the UTRC from 1989 to 2018.

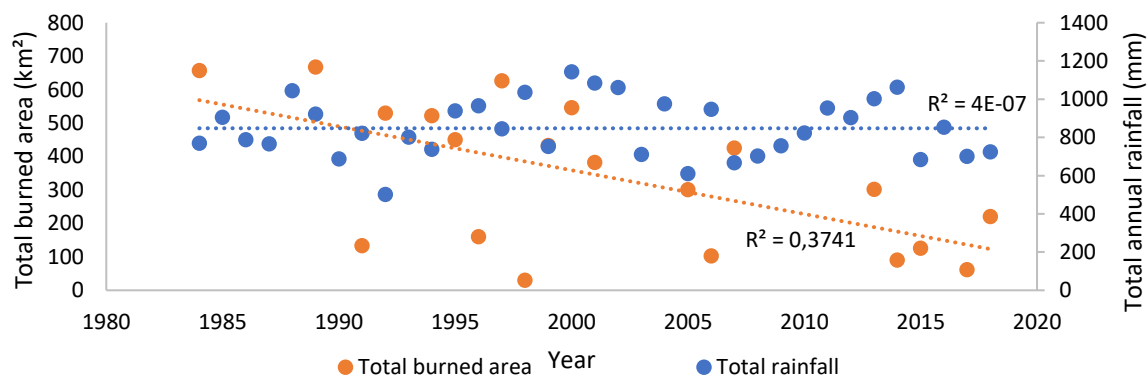


Figure 4.7: A comparison of annual rainfall for Maclear and the annual burned area in the UTRC from 1984-2018.

4.2.2 Identifying fire frequency of the upper Tsitsa catchment using Landsat imagery

Landsat imagery was collected from between 1984 and 2018 to compile a fire frequency map using ArcMap 10, comprising fire scars for 20 sampled years (Appendix 1) to describe the fire regimes in the UTRC. Missing data were due to cloud cover and not included in the mapping process. Fire frequency varies throughout the catchment, with some areas experiencing fires 13 years of the 20-year sampling period, and other areas experiencing none. It is evident that there are areas the western side of the catchment that experiences a relatively higher occurrence of fires, whilst the eastern part of the catchment experiences fewer fires (FRI 3-5 years). The western side of the catchment which, is predominantly commercial land, has areas that are regularly burned, with an FRI of 1-2 years in frequently burned areas (Figure 4.8). These are grassland areas used for grazing livestock, surrounded by commercial plantations. Land cover and management type varies across the catchment and is affecting the spatial patterns of fire.

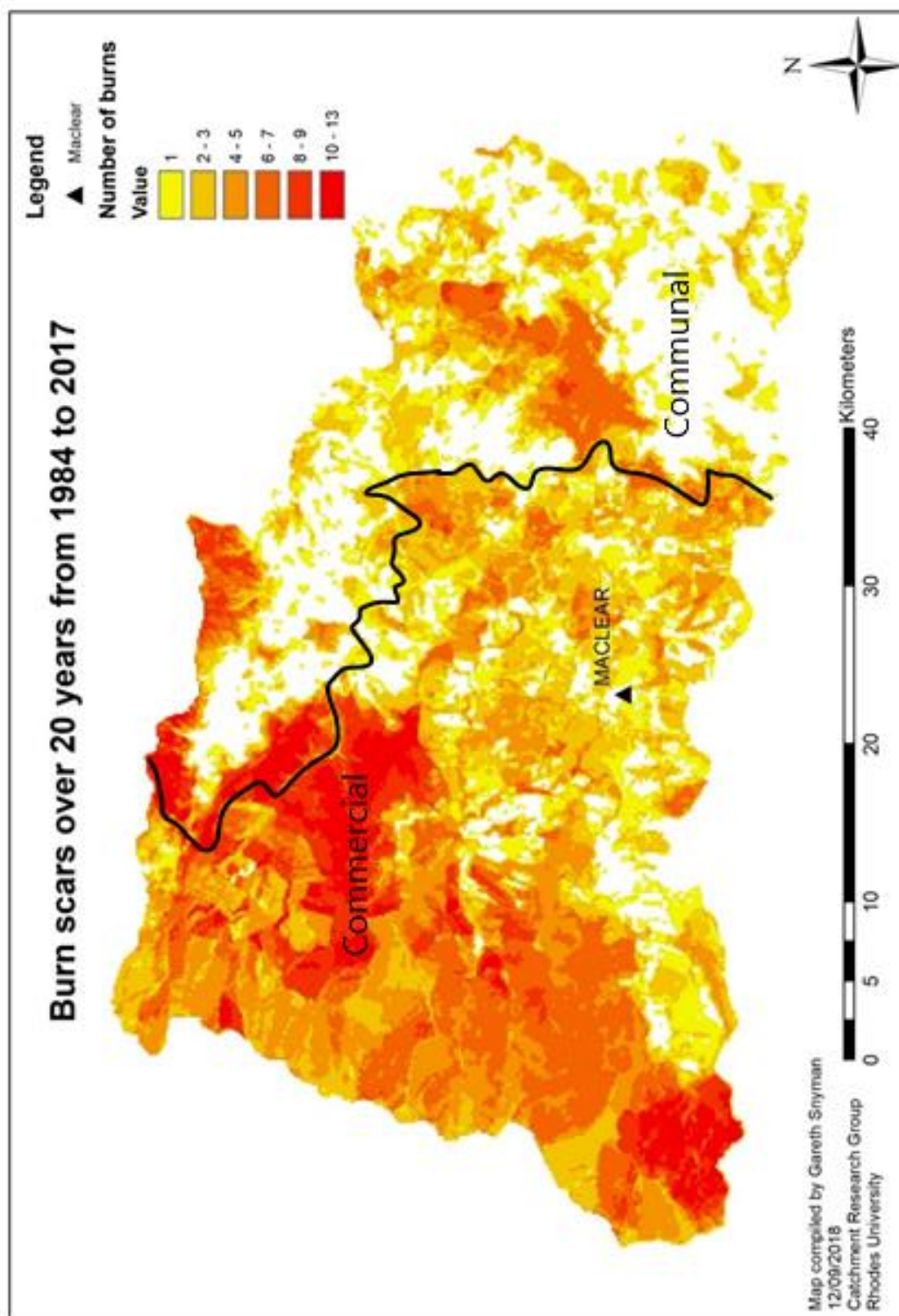


Figure 4.8: Fire frequency in the UTRC from 1984 to 2017. Fire frequency is indicated by graded colours which represent the number of burns, with no colour being a value of 0.

4.2.3 Determining fire intensity

Fire intensity was determined using a combination of data from MODIS and VIIRS satellite sensors. The sensors detect thermal anomalies as they pass over the study area and record the time, date, location and FRP of the event. This was extracted using a combination of ArcMap and QGIS to produce a summary of the intensity of fires in the commercial and communal areas within the UTRC from 2001 until present. There are a few hotspots scattered throughout the catchment, however the most intense fire (>1700 MW per pixel) is located in the lower west region, which is primarily commercial forestry land, and could have potentially been caused by an unplanned forest fire, but is more likely to be burning to clear land after harvesting the timber crop (Figure 4.9). Commercially farmed areas (plantations, livestock and crops) experience more intense fires with a median FRP of 23,5 MW per pixel, compared to communally managed areas with a median FRP 14,2 MW per pixel (Figure 4.10).

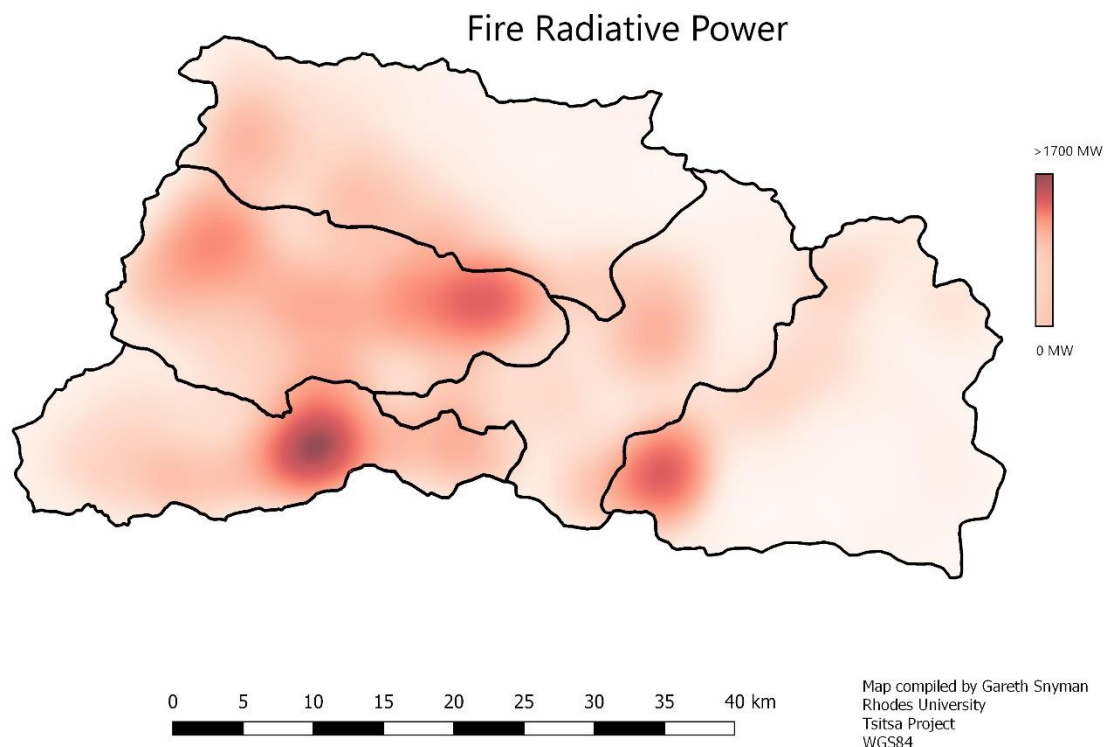


Figure 4.9: Fire intensity hotspots in the UTRC. Data collected by MODIS and VIIRS satellite sensors and analysed in QGIS.

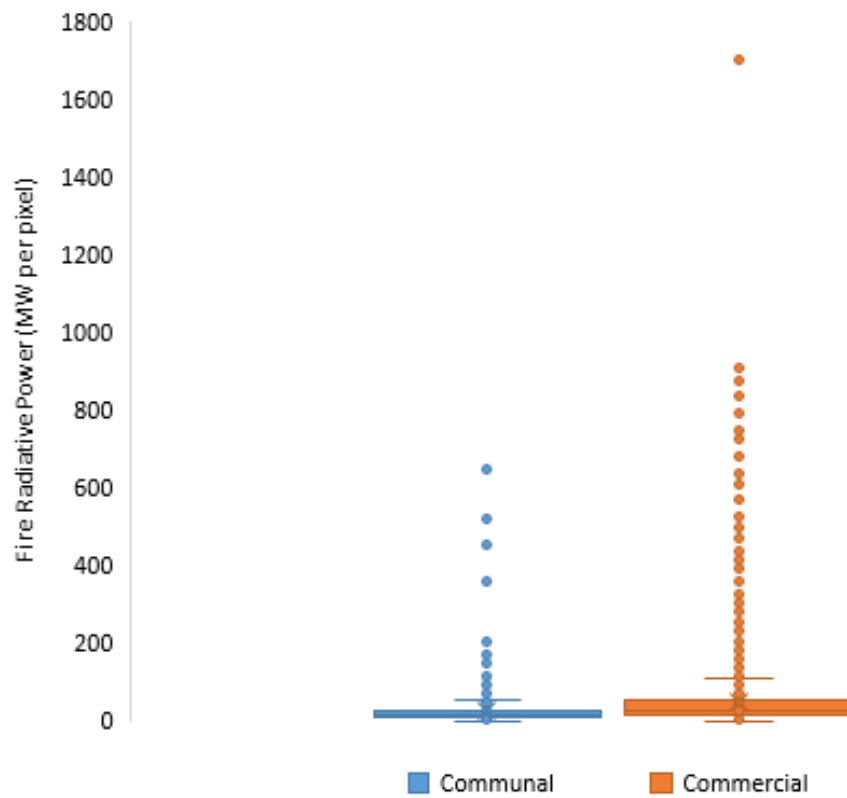


Figure 4.10: Box and whiskers representing the mean FRP detected by MODIS and VIIRS from 2001 to 2018. Whiskers show that there are many outliers and that fires can be extremely intense.

4.3 The effect of fire frequency on soil properties and vegetation

One of the objectives of this study was to quantify how fires affect soils and in turn erosion potential/degradation of the landscape. Multiple tests and measurements were carried out in the field, as well as soil samples collected to be further analysed in a laboratory. These tests were done in order to assess how and to what extent fires are affecting soil properties in the UTRC.

4.3.1 The effect of fire frequency on WDPT

Soil hydrophobicity, or water repellency, was measured in the field using the WDPT (Doerr *et al.*, 1996). This required dropping water droplets onto the soil surface and recording the time which they took to penetrate. The time it took for the droplet to be completely absorbed was recorded as the penetration time and allows us to rank the severity of water repellency of the soil. Water repellency was recorded because hydrophobic soils generate more runoff due to lack of infiltration. Increased or excess runoff promotes erosion due to higher volumes of water travelling across the soil surface leading to degradation of the landscape. A statistical analysis by ANOVA determined that there was a significant interaction ($p < 0.001$) between fire frequency and WDPT, which means that on average, a high fire frequency resulted in a higher WDPT than low fire frequency (Table 4.3). There was also a significant difference ($p = 0.031$) between the soil types, with mudstone having a higher WDPT than sandstone. Interestingly, there was no significant interaction ($p = 0.05$) between fire frequency and the soil type (mudstone and sandstone) which means that regardless of soil type, a higher fire frequency will have a higher WDPT than low fire frequency. However, it is important to point out that despite the high variability within the data ($CV \% = 94.2$), the ANOVA was still able to indicate significant differences. The variation in the data explains why the interaction between fire frequency and soil type was not significant ($p < 0.05$).

Table 4.3: The effect of fire frequency and soil type (mudstone and sandstone) on WDPT. Data were analysed by ANOVA ($n = 10-40$; $p < 0.001$).

Variate: WDPT (s)					
Relationship between:	d.f.	s.s.	m.s.	v.r.	F pr.
Fire frequency and WDPT	1	240.1	240.1	27.28	<.001
Soil type and WDPT	1	44.4	44.1	5.01	0.031
Fire frequency and soil type	1	36.1	36.1	4.1	0.05
Residual	36	316.8	8.8		
Total	39	637.1			

A second analysis was performed on the data in the form of a multiple regression model (Figure 4.11). The test used the actual number of burns rather than the frequency range allocated in the previous analysis, and the two soil types were combined. The regression analysis indicated that there was no difference in the response to the number of burns between the two soil types (mudstone and sandstone), and that WDPT of soils increased exponentially with the number of fires they were exposed to.

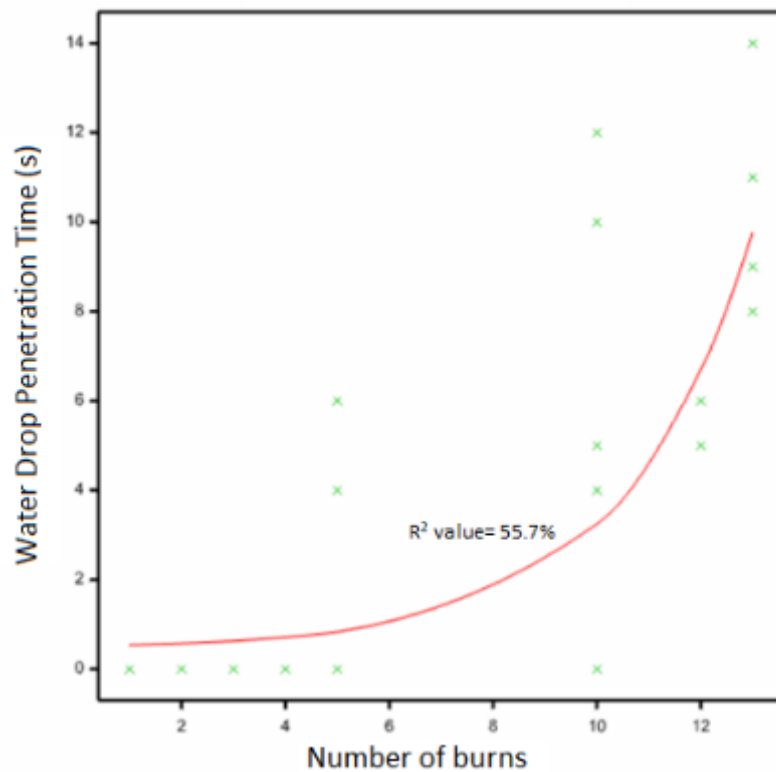


Figure 4.11: The interaction between number of burns and WDPT on two soil types (mudstone and sandstone combined).

4.3.2 The influence of particle size on WDPT

Soil particle size was measured on mudstone and sandstone soil samples in the lab using a sieve stack to determine whether it influenced WDPT on soils that experienced high and low fire frequency. A regression analysis was conducted to determine the linear relationship between WDPT and soil particle size (Figure 4.12). The regression analysis showed that as particle size increased, WDPT decreased, however there is a very poor relationship between the two and that there was no significant interaction ($p = 0.4080$).

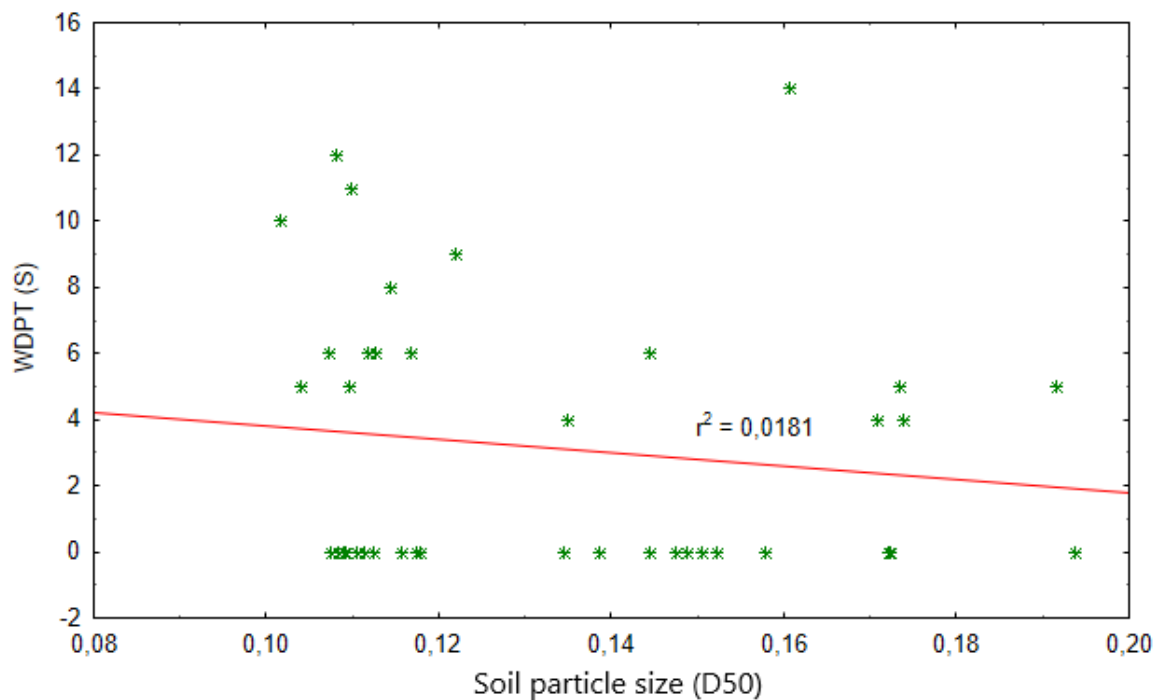


Figure 4.12: The relationship between soil particle size (mm) and WDPT ($r^2 = 0.0181$).

4.3.3 The effect of fire frequency and soil type on soil surface hardness

Soil surface hardness was measured to investigate the effect of fire frequency on two different soil types (mudstone and sandstone). This was done in order to assess whether fire frequency is contributing to soil crusting in the UTRC grasslands. There was a significant difference between soil exposed to a high and low fire frequency ($p < 0.001$), and there was no significant interaction between fire frequency and soil type (Figure 4.13; $p = 0.715$). This relationship was further demonstrated by a regression analysis which considered the number of burns a site experienced over 30 years and grouped the soil types to show how the number of burns influences soil surface hardness. The regression analysis showed a

positive moderate relationship ($r^2 = 36.8\%$) between number of burns and soil surface hardness (Figure 4.14), which complements what was found with the ANOVA.

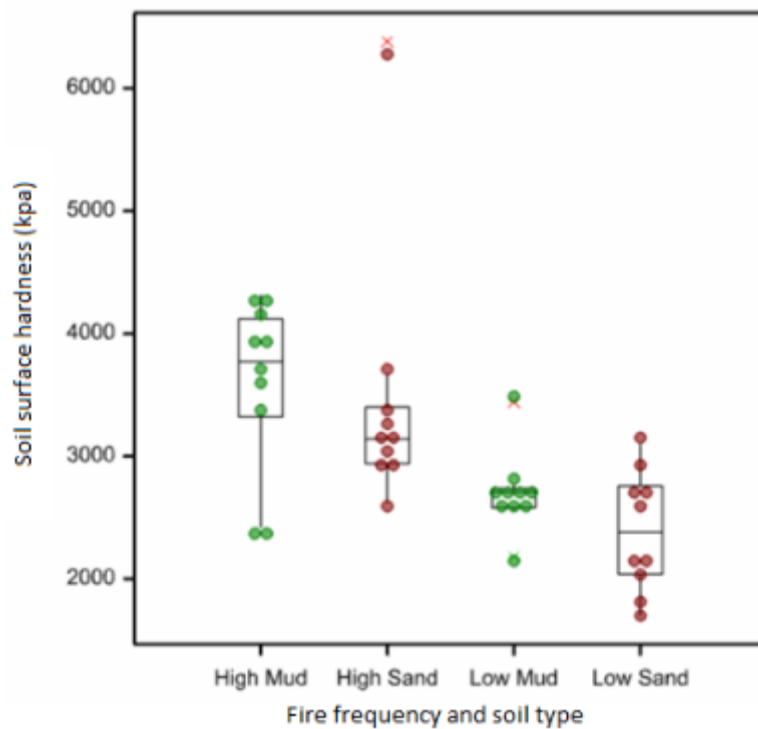


Figure 4.13: The effect of the number of burns (grouped into high and low frequency for >10 burn and 1-9 burns, respectively) on soil surface hardness for two soil types - mudstone and sandstone. Data were analysed by ANOVA ($n = 10-40$; $p < 0.001$).

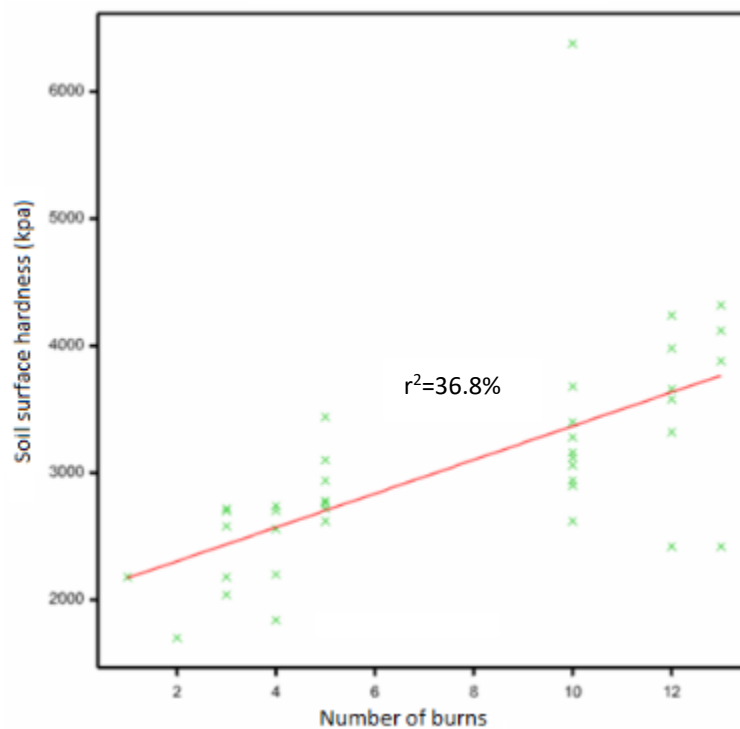


Figure 4.14: The effect of number of burns over 30 years on soil surface hardness ($r^2 = 36.8\%$; $p < 0.001$).

4.3.4 The effect of fire frequency on soil organic matter

Soil samples were taken from each sampling site covering two dominant soil types in the catchment (mudstone and sandstone) and analysed in the laboratory in order to determine the effect of fire frequency on soil organic matter. The data were analysed by an ANOVA and it was found that whilst there was no relationship between fire frequency and soil organic content, soil type determined the amount of organic matter as sandstone had a significantly higher organic content (11 %) than mudstone (4,9 %) as seen below (Table 4.4; $p < 0.001$). The data suggests that on average there is approximately double the proportion of organic matter in sandstone soils when compared to mudstone soils (Figure 4.15).

Table 4.4: ANOVA analysis showing the interactions between fire frequency, soil type and soil organic content.

Variate: Soil organic matter content					
Relationship between:	d.f.	s.s.	m.s.	v.r.	F pr.
Fire frequency and organic content	1	2.45E-05	2.45E-06	0.03	0.874
Soil type and organic content	1	0.03972	0.03972	41.07	<0.001
Fire frequency and soil type	1	0.000187	0.000187	0.19	0.663
Residual	36	0.034813	0.000967		
Total	39	0.074745			

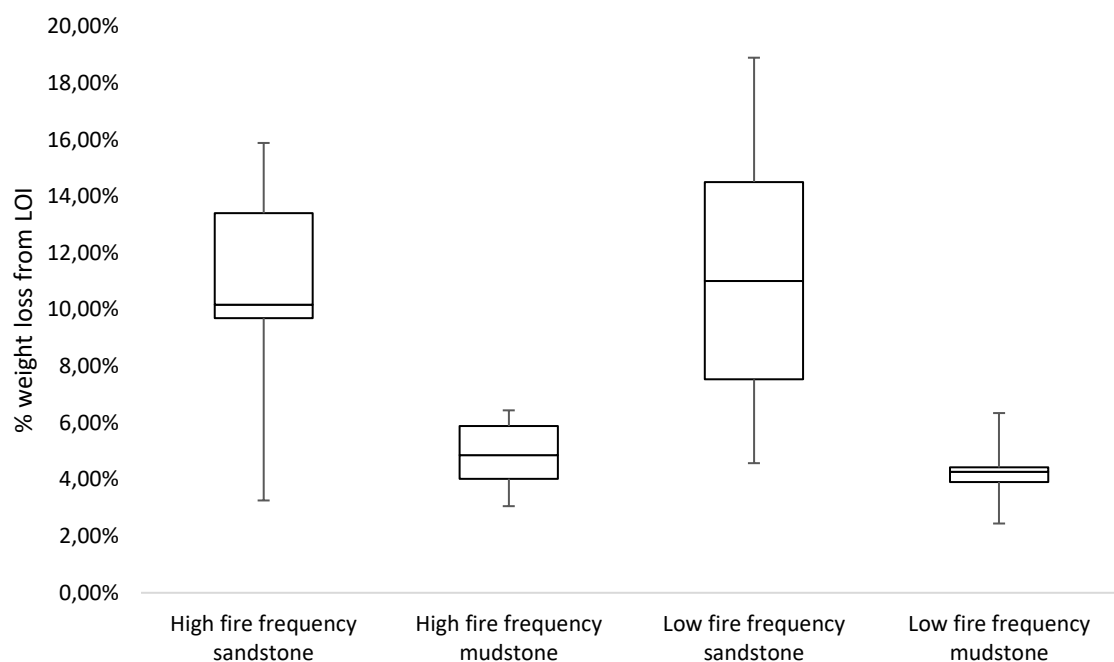


Figure 4.15: Comparing organic content loss in soils to fire frequency and soil type (mudstone and sandstone), where sandstone had a significantly higher organic matter ($p < 0.001$).

4.3.5 The effect of fire frequency on soil pH

Soil pH was measured in samples collected from each site in the field and analysed in the laboratory using a pH probe (Eutech pH 700 Meter). This was to assess the impact of fire frequency on soil pH on two different soil types (mudstone and sandstone). An ANOVA test was performed on the data captured which showed that on average, mudstone has a higher pH than sandstone, and that mudstone soils exposed to a high frequency of fires have a higher pH compared to soils exposed to a low frequency. There was a significant difference between mudstone and sandstone at a high fire frequency (Table 4.5; $p < 0.001$), however there was no significant difference between the two soil types at a low fire frequency (Figure 4.16).

Table 4.5: ANOVA analysis showing the interactions between pH, fire frequency and soil type (n=10-40; $p = 0.01$).

Variate: Soil pH					
Relationship between:	d.f.	s.s.	m.s.	v.r.	F pr.
Fire frequency and pH	1	0.50456	0.50456	7.38	0.01
Soil type and pH	1	2.66772	2.66772	36.41	<0.01
Fire frequency and soil type	1	1.0923	1.0923	14.91	<0.01
Residual	36	2.63769	0.073237		
Total	39	6.93828			

A Fisher's protected (LSD) test was also performed to test the differences between high and low fire frequency on pH on mudstone and sandstone sampling sites. Significant differences are indicated by a, b, ab, and c below (Figure 4.16). The analyses show that sandstones exposed to a high fire frequency (a) were significantly different from both high (c) and low (b) fire frequency fires on mudstone. Mudstone samples that experienced a high fire frequency (c) are significantly different from sandstones at a high (a) and low (ab) fire frequency as well as low fire frequency mudstone sites (b).

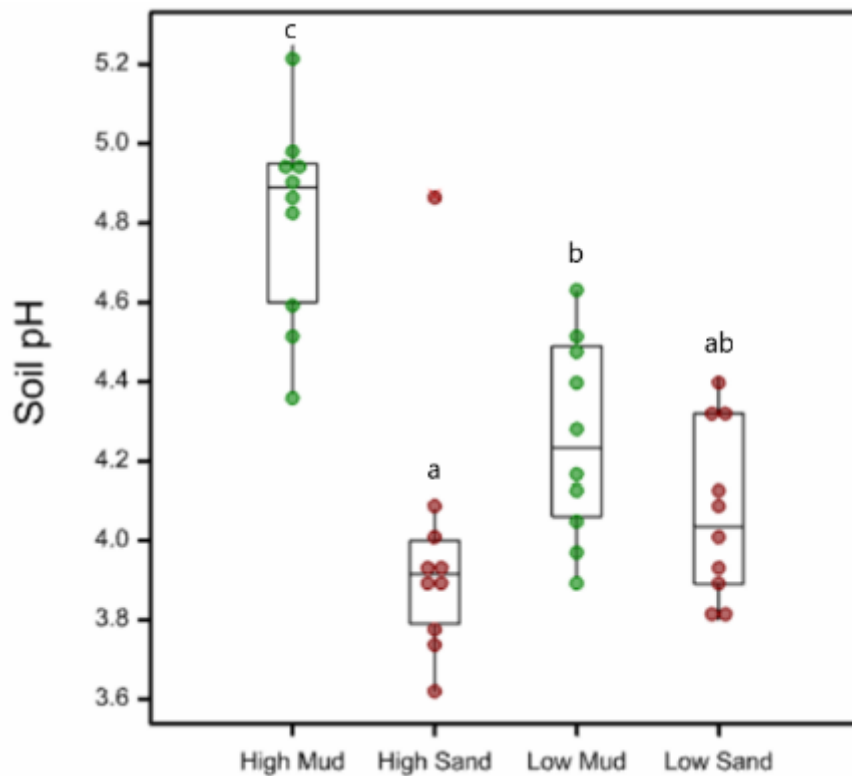


Figure 4.16: The effect of fire frequency (high and low) on the pH of mudstone and sandstone soils. Data were analysed by ANOVA and Fisher's protected least significant difference test (n=10-40) Significant differences indicated by a, b, ab, c.

4.3.6 The effect of fire frequency on above-ground biomass

Above-ground biomass was measured with a DPM at a total of 40 sites in the field. The data were used to determine whether fire frequency effects above-ground biomass on two different soil types (mudstone and sandstone). The data were Log10 transformed and analysed by ANOVA. A Fisher's LSD was also performed on the data, and it shows that above-ground biomass at a low fire frequency on sandstone (a) was significantly different from the rest of the samples; high fire frequency sandstone and mudstone, low fire frequency mudstone (b) (Figure 4.17). Whilst there was no interaction between fire frequency and above-ground biomass, there was a significant interaction between fire frequency and soil type. There was no difference between mudstone and sandstone sites exposed to a high fire frequency, however, at a low fire frequency, sandstone has a significantly higher above-ground biomass than mudstone ($p= 0.002$)

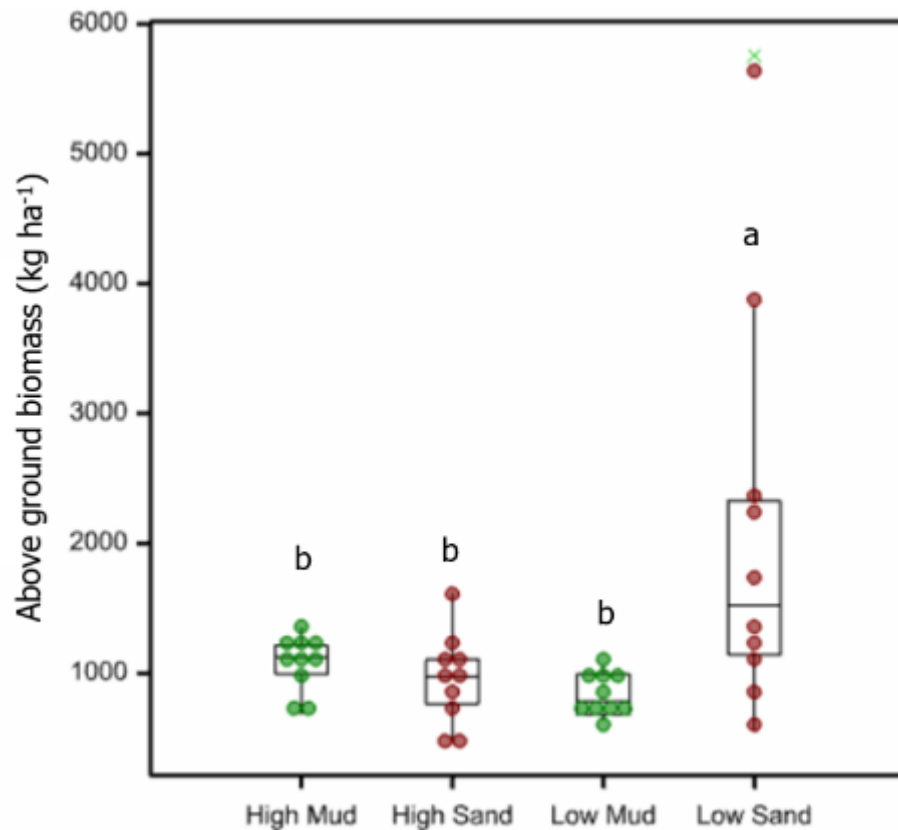


Figure 4.17: The effect of fire frequency and soil type on above ground biomass. Data were analysed by ANOVA and Fisher's protected least significant difference test (n=10-40) Significant differences indicated by a, b.

4.3.7 The influence of fire frequency on soil particle size

Soil particle size was measured by putting samples from the field through a sieve stack. This was then compared to fire frequency to assess whether more frequent fires influenced the median soil particle size in mudstone and sandstones in the study area. The data were analysed by ANOVA and it was found that there is a significant interaction between fire frequency and soil particle size ($p=0.026$) and that soil type had no influence on median soil particle size (Table 4.6). A box and whisker plot were produced, showing that a higher fire frequency resulted in a smaller median particle size, and a low fire frequency resulted in a larger median soil particle size (Figure 4.18).

Table 4.6: ANOVA analysis showing the effects of fire frequency and soil type on the median soil particle size

Variate: Median soil particle size					
Relationship between:	d.f.	s.s.	m.s.	v.r.	F pr.
Fire frequency and particle size	1	0.003720	0.003720	5.409	0.026
Soil type and particle size	1	0.000080	0.000080	0.116	0.735
Fire frequency and soil type	1	0.000019	0.000019	0.027	0.870
Residual	36	0.024758	0.000688		
Total	39	0.028478			

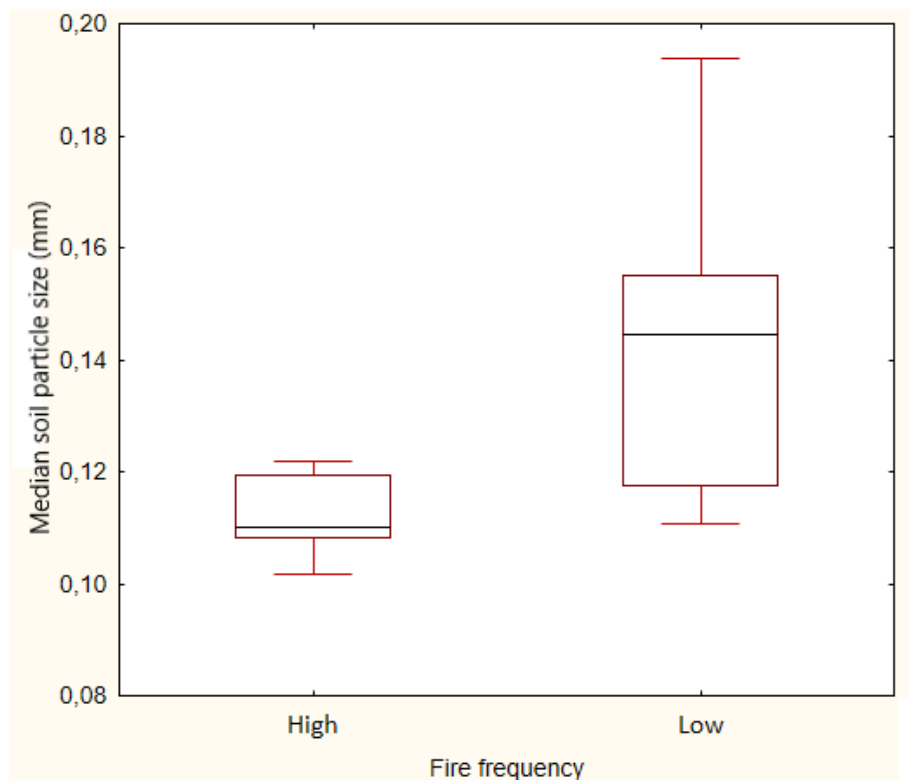


Figure 4.18: Relationship between the number of burns and median soil particle size.

4.3.8 Principal component analysis

A PCA was performed to better understand the data and the relationships between fire frequency and the variables measured during the study. Principal component 1 explains 44.61 % of the variation, whereas component 2 explains 17.75 % of the variation within the data (Figure 4.19). The PCA showed a strong positive correlation between WDPT, soil surface hardness and fire frequency, whilst there was a negative correlation between biomass and fire frequency. These correlations show that WDPT, soil surface hardness and biomass are all directly affected by fire frequency, whilst soil organic content and soil pH have a very weak interaction with fire frequency, and that changes in either are a product of soil type (soil parent material) or other variables that were not considered during the study. The location of points within the PCA biplot indicate the differences between them (high fire frequency on mudstone- black; high fire frequency on sandstone- red; low fire frequency on mudstone- green; low fire frequency on sandstone- blue) with sandstone and mudstone points exposed to a high fire frequency having a higher soil surface hardness and WDPT than soils exposed to a low fire frequency. The vertical difference between mudstone and sandstone points are due to the soil organic content and pH (likely a product of soil parent material).

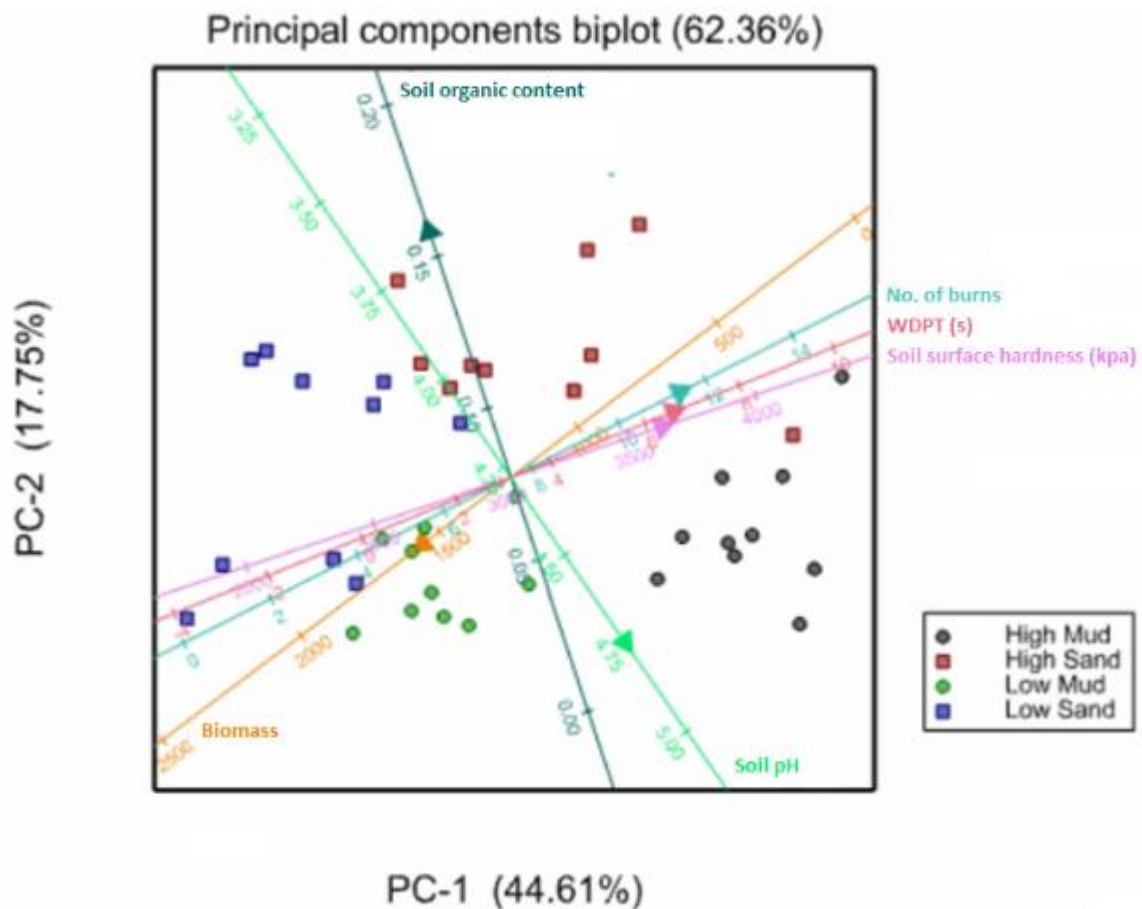


Figure 4.19: Principal Component Analysis showing correlations throughout the data collected during the study.

4.4 Comparing soil characteristics between sites that burned in 2017 and 2018

Soil samples were collected from 20 sites that had burned recently; 10 sites that burned in 2017 (1 year prior to data collection) and 10 sites that burned in 2018 (weeks- days before data collection). These two sampling groups were compared to one another to determine whether the soil and vegetation characteristics regenerates over time. There were no significant differences observed between above-ground biomass (Table 4.7; $p = 0.055$) and pH (Table 4.8; $p = 0.239$) between 2017 and 2018, however there were observed trends between soil surface hardness, WDPT and soil organic content which will be explored below.

Table 4.7: The difference between the effects of fire on above-ground biomass over a one-year recovery period.

Variate: Biomass (kg. ha⁻¹)					
Relationship between:	d.f.	s.s.	m.s.	v.r.	F pr.
Year and biomass	1	4503574	4503574	4.3	0.055
Soil type and biomass	1	3964061	3964061	3.79	0.069
Year and soil type	1	844523	844523	0.81	0.382
Residual	16	16751420	1046964		
Total	19	26063578			

Table 4.8: An ANOVA summary comparing between the effects of fire on soil pH over a one-year recovery period.

Variate: Soil pH					
Relationship between:	d.f.	s.s.	m.s.	v.r.	F pr.
Year and pH	1	0.07688	0.07688	1.5	0.239
Soil type and pH	1	0.00392	0.00392	0.08	0.786
Year and soil type	1	0.02312	0.02312	0.45	0.512
Residual	16	0.8224	0.0514		
Total	19	0.92632			

4.4.1 Comparing soil surface hardness between sites that burned in 2017 and 2018

Data were analysed by ANOVA to compare soil surface hardness between 2017 and 2018 burned sites. (Table 4.9). It was found that there was no significant difference in between the time since burning and the soil surface hardness on either mudstone or sandstone (Figure 4.20). A general comment that can be made is that mudstone seems to be slightly harder than sandstone and this is evident throughout the study and is noted in other soil surface hardness tests.

Table 4.9: ANOVA showing the interactions between soil surface hardness and fire on two different soil types between sites that burned in 2017 and 2018.

Variate: Soil surface hardness (kpa)					
Relationship between:	d.f.	s.s.	m.s.	v.r.	F pr.
Year and SSH	1	2158245	2158245	5.42	0.033
Soil type and SSH	1	3256245	3256245	8.17	0.011
Year and soil type	1	804005	804005	2.02	0.175
Residual	16	6374560	398410		
Total	19	12593055			

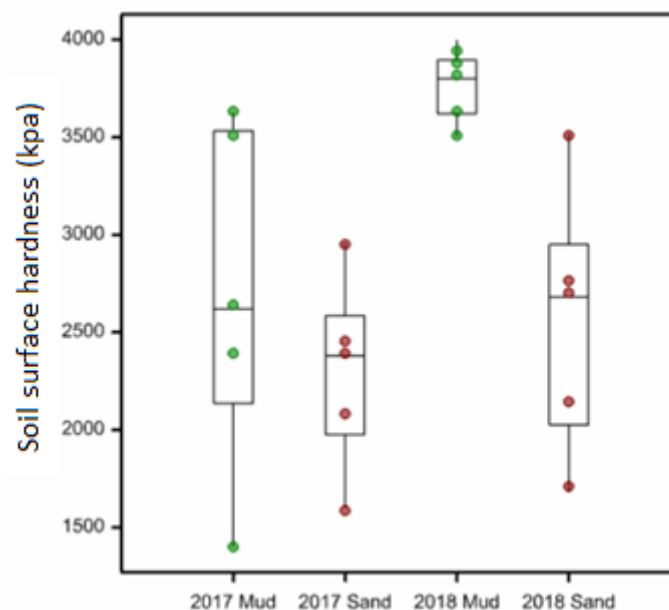


Figure 4.20: A comparison of soil surface hardness between sites that burned in 2017 and 2018.

4.4.2 A comparison of WDPT between 2017 and 2018 burn sites

Measurements of WDPT were taken in the field and the data analysed by ANOVA and a post-hoc Fisher's protected least significant difference test. There was no significant change in the WDPT between the sites that burned in 2017 and 2018. However, there was a significant difference between the two soil types ($p = 0.013$) in 2018. Mudstone sites that burned in 2018 had a significantly higher WDPT than sandstone sites that burned at the same time and this is illustrated by Fisher's protected least significant test (Table 4.10).

Table 4.10: Significant differences between WDPT, soil type and year burned identified by Fisher's protected least significant difference test. Significant differences are indicated by a, b, ab.

Site	Mean	Significance
2018 Sandstone	0.000	a
2017 Sandstone	1.719	ab
2018 Mudstone	1.832	b
2017 Mudstone	3.306	b

4.4.3 The difference between soil organic content in two different soil types in 2017 and 2018 burn sites

Soil samples from sites that burned in 2017 and a year later (2018) were analysed for SOM loss. The results were analysed by ANOVA and there was no difference in organic content loss for mudstone soil samples between 2017 and 2018. The 2018 sandstone soils had a significantly higher organic content than the 2017 sites that had burned one year prior (Figure 4.21; $p < 0.01$). Fishers LSD was performed on the data and displays the differences between the samples, with the mudstone sites that burned in 2018 (a) being significantly different from the rest of the sampling groups (b) (2017 mudstone and sandstone and 2018 mudstone).

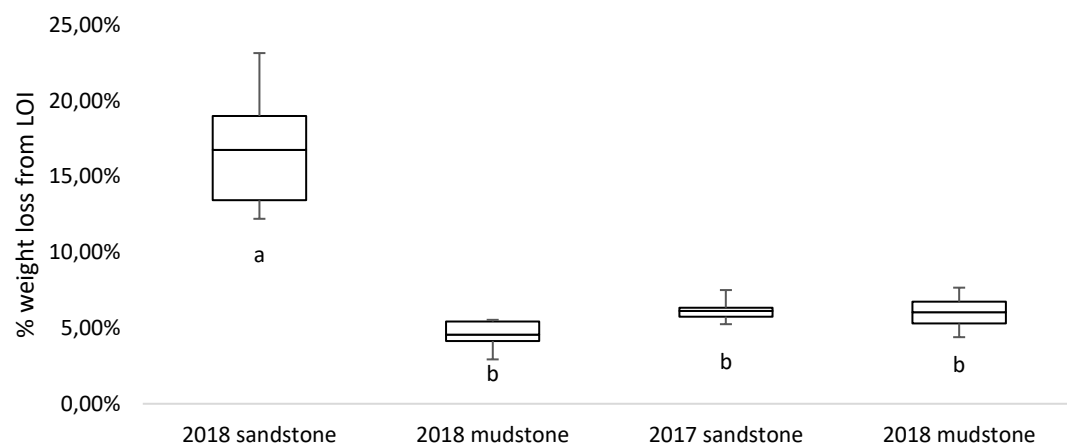


Figure 4.21: A modified Box and Whisker plot showing the differences in soil organic content between samples that burned in 2017 and 2018 on two different soil types (mudstone and sandstone), with significant differences between samples indicated by a, b.

4.5 Summary

A summary of the results serves to link what was found in terms of the research questions and objectives laid out at the beginning of the study. It was found that there are more frequent fires in the commercially managed part of the catchment compared to the communally managed land. The fire season was also longer by two months in the commercial areas compared with the communal areas. Fires on the commercially managed land also tend to be larger than fires in the rest of the catchment. Overall, the commercial land is experiencing larger fires, more frequent fires and a longer burning season than the communal land, and this will aid in describing the fire regimes of the catchment set out in Objective 2. Statistical analysis on the measurements taken from soil samples show that there is a significant relationship between fire frequency and water repellency as well as soil surface hardness. There were no evident relationships between fire frequency and organic content loss, soil particle size and pH. The results obtained in this section will assist in completing Objective 3 (The effects of fire frequency on soil properties).

5. Discussion

5.1 How are fire regimes best defined and detected in the upper Tsitsa?

A review of the literature found that remote sensing, in combination with GIS was the most effective method to assess and monitor fires and fire regimes across a large landscape, such as the UTRC, as it allows for an efficient desktop process that can be accomplished remotely at a relatively low cost (Justice *et al.*, 2002; Hudak & Brockett, 2004; Archibald *et al.*, 2013). It was essential in collecting twenty years of data, over a period of 34 years from 1984-2018 which were used to produce fire scar maps and graphs which enables us to determine the fire regimes of the upper Tsitsa (Appendix 1).

There have been a variety of studies undertaken on fires in southern Africa, which utilised an assortment of remote sensing techniques. However, this study looked at fire regimes as defined by Archibald *et al.*, (2013), who took into account a wide array of variables that can be detected remotely. Archibald *et al.*, (2013) broke fire regimes down into key characteristics; frequency, seasonality, extent, size and intensity, which can all be measured using remote sensing and combined to define a “fire regime”.

There is an assortment of satellite sensors that are freely available over the internet that were applicable to this research, however, this project was designed around papers that found success using Landsat’s moderate-resolution imagery (Hudak & Brockett, 2004; Joyce *et al.*, 2009; Roy *et al.*, 2014; Schroeder *et al.*, 2015), as it was the most appropriate for the temporal and spatial scale of this study. Whilst Landsat imagery was used to assess the fire frequency, size and extent of fires within the UTRC, it could not accurately detect fire intensity and seasonality due to its onboard sensors and therefore was limited in this regard. MODIS and VIIRS sensors were more appropriate to determine fire seasonality and fireline intensity as the data captured incorporates time, date and FRP (Justice *et al.*, 2002; Archibald *et al.*, 2013; Giglio *et al.*, 2016).

The timing of data collection for Landsat imagery was determined by fire seasonality obtained from MODIS. Images were collected from each year between August and October (peak of burning season; Figure 4.3 & 4.4), and imagery from any earlier or later in the year would not have the full extent of fire scars or grasses had turned vibrant green from the first rains after burning, making fire scars difficult to detect. Field observations confirm this and many areas that were burned earlier in the year had already started going green with new shoots and vegetation when imagery was collected. For this reason, a combination of unsupervised and supervised classification was used during image processing. On average 68% of burn scars were identified using unsupervised classification. Supervised classification was essential for years where cloud cover obscured images from during the peak fire season (August to October), and images from November were used. This number is relatively low as some of the imagery collected from later in the year (November) was more difficult to classify during the unsupervised classification process than data collected earlier in the year (September and October). In this case, supervised classification was the preferred method as it introduced a human factor and was more efficient in detecting fire scars that were undetected or incorrectly classified by unsupervised classification. In some cases, there was too much cloud cover for the images to be used, and these years were left out of the study. Further limitations arose due to the striped/banded nature of Landsat 7 imagery resulting in data gaps, and for this reason these data were left out of the study. Whilst MODIS and VIIRS were effective in providing fire hotspot intensity and seasonality, they only detected active fires and could not provide accurate data on spatial patterns, hence the need for Landsat imagery. Another limitation of the MODIS and VIIRS sensors was that they only started collecting data in 2001, whilst Landsat imagery dates to the early 1980s (Table 2.2), and this meant that fire seasonality and intensity was only recorded from 2001 onward. Even though Landsat has a larger data span, the purpose of this study was to identify fire regimes and their characteristics in the UTRC and Landsat, MODIS and VIIRS were reliable sources of data and information for doing so.

5.2 Spatial and temporal dynamics of fire regimes in commercially and communally managed land in the upper Tsitsa River catchment

Fire occurrence is regular throughout the upper Tsitsa catchment; it occurs naturally through lightning strikes during intense weather events (mostly in summer) and is ignited by humans for planned burns and, in some cases, unintentionally. Approximately 65% of the upper Tsitsa is comprised of mesic grasslands (Mucina & Rutherford, 2006), which are essential to the livelihoods of many communities as they serve as grazing land for commercial and communal livestock farmers. With fire being such an integral part of the grassland landscape in the UTRC, it was important to assess the different components of fire regimes and determine how, when, where and why fires were being used. Throughout the literature and this study, it was evident that fires and fire regimes are complex systems that are affected by a multitude of different ecological, anthropogenic and climatic factors (Archibald *et al.*, 2010;

Morgan *et al.*, 2002; Govender *et al.*, 2006), with many of the key characteristics of fire regimes closely interlinked to one another.

Using remotely sensed imagery, it was determined that fire regime characteristics differ between the two different land management classes in the catchment (commercial and communal management). A high number (Figure 4.2), of large (Table 4.1), frequent fires (Figure 4.8) occurred in the commercial areas and smaller, less frequent fires occurred in the communal areas (Figure 4.8; Table 4.1). This is aligned with what Archibald *et al.*, (2010) found, with FIL and FCS pyromes dominating grasslands around the world. Archibald *et al.*, (2010) stated that the difference between FIL and FCS pyromes is the Human Impact (HI) score, which integrated population density and land transformation (Bowman *et al.*, 2009; Archibald *et al.*, 2010; Aldersley *et al.*, 2011). This information is relevant to the UTRC as there are two clear different land management types, and the influence humans are having on fires is evident; in the commercially managed areas of the catchment; fires are being effectively used as a land management tool. This may be due to the abundance of relatively unfragmented sour veld which yields a high biomass per unit area which is ideal for frequent fires (Mucina & Rutherford, 2006). Commercial livestock farmers burn every 1-2 years to clear old grasslands for new grazing, and all commercial farms (livestock, agriculture and plantations) burn fire breaks yearly to prevent the destruction of crops and other assets (D. Sephton, personal communication, 10 October 2018). This is in line with the SANBI guidelines for managing grassland biodiversity and livestock production (SANBI, 2014). Alternatively, communal farmers are not utilising fires as much as commercial farmers and fires only burn the same area every 3-5 years with some areas experiencing even larger fire return intervals (Figure 4.8). This is likely due to the difference in the scale of farming (Figure 1.8), and that communal lands are shared between communities and villages. This study hoped to identify changes in fire frequency over the monitoring period, however, it was unable to do so due to the gaps in data and this future research could investigate these shifts in fire regimes.

Communally managed areas of the catchment are made up of sweet veld which receives less rainfall than the commercially managed areas, resulting in lower fuel loads. Low biomass combined with the high grazing pressure and a high degree of grassland fragmentation (Figure 1.6) results in grasslands that cannot support large and frequent fires (Mucina & Rutherford, 2006). Archibald *et al.*, (2009) stated that there were perceptions that more fires were ignited on communal land than privately owned commercial farms and found that conversely, more fires occurred on commercial land than on communal land, and this finding was shared by this study. However, it may be the case that more unplanned ignitions take place on communal land than commercial land due to negligence (car exhausts, household fires, cigarette buds) and poor fire control practices. There are negative perceptions in some communities about the use of fire, and it is not always seen as a viable tool as it can cause loss of grazing land and in the worst cases property and life if not correctly controlled (B Gusha, personal communications, 9 October 2019). This may be one of the reasons we see a difference between the fire

frequency in commercial and communally managed land across the catchment. Another hypothesis could be that due to the mismanagement and overgrazing of the communally managed land, there are insufficient fuel loads to support larger and more frequent fires, and this can be investigated in further research.

Fire size is another key aspect of fire regimes which is linked to Archibald's global pyromes and was closely examined during this study. Burn scars extracted from Landsat imagery (used for assessing fire frequency), were analysed to assess the annual spatial extent of fires within the UTRC. The annual burned area was much higher in the commercial areas than the communal areas throughout the study (Table 4.1). However, there is a difference in the size of the two land management types in the catchment, with the communally managed land covering approximately 1200 km² and the commercially managed land 770 km² and the annual burned area was not an accurate measure of how much the land management areas burned compared to one another. For this reason, the percentage of two land management areas was calculated for a more detailed comparison, which found that on average, a much higher portion of the commercially managed land is burned annually (25,1 %) compared to the communally managed land (12 %) (Figure 4.5). This comes down to grassland fragmentation and fuel loads availability, because if there are fuels present, they will burn at some stage regardless of management types and practices as this is one of the inherent principles of grasslands that have been evident in this study (Vogl, 1979). A steady decrease in total burned area was found during the monitoring period (Table 4.3). The decrease in total burned area was related to a decrease in the maximum and average burn scar size. This may be due to the implementation of government actions to prevent risk around settlement areas as well as an increase in agriculture, resulting in smaller, controlled fires. Whilst there was no clear relationship found between rainfall and fire size in this study, literature states that there is a direct relationship between the two. Rainfall controls vegetation growth and therefore biomass and fuel load availability (Mucina & Rutherford, 2006).

5.2.1 Fire intensity and seasonal patterns in the UTRC

Fire intensity and seasonality was assessed in the UTRC using a combination of MODIS and VIIRS point data that was processed to determine when fires occur during the year. The data were available from 2001 until present and were processed in QGIS, which allowed for a comprehensive timeline of fire occurrences within the catchment. For the purpose of this component of fire regimes in this study, the catchment was again split into two management groups- commercial and communally managed land. It was found that in the commercial part of the catchment, fires occur as early as March, likely due to fire breaks being burned to act as buffer zones around plantations and farm boundaries in accordance with the NVFFA, with the fire season starting in June and ending in October, with the peak number of fires occurring in August (Figure 4.4). This is slightly different to the communally managed

land as data shows that fires start occurring in June (2 months later than commercially managed land), with the peak number of fires occurring in July, and the fire season ending in September (Figure 4.3). The seasonality graphs (Figure 4.3 & 4.4) show us that the commercial land has a fire season that lasts as long as 6 months, whilst the communal areas have a fire season that is 4 months long. The longer and earlier fire season could be due to the difference in altitude and that the commercial areas receive frost slightly earlier than the communal areas, curing the grass and increasing ignition potential, as well as early burning of fire breaks whilst fuel loads are still moist to prevent runaway fires (Mucina & Rutherford, 2006). There is also a difference in the number of fires that burn, with the commercially managed land experiencing a much higher number than the communally managed areas (Figure 4.2). Fire seasonality is closely linked to fire intensity and is determined by climatic conditions and the availability of dry fuel loads (Chuvieco *et al.*, 2008). Fires occur in areas with strong seasonal rainfall, allowing vegetation to rapidly grow during wet months which provide ample fuel loads during the dry months (Archibald, 2010). The findings for fire seasonality in this study are in line with what is generally observed for grasslands and savannas all over southern Africa (Bond *et al.*, 2005; Govender *et al.*, 2006; Archibald *et al.*, 2010), with the highest number of fires occurring during the dry season (June- September) when there is little precipitation (Figure 1.2). This was expected due to the low relative humidity and fuel load (moribund grasses) having a low moisture content, the vegetation was easily ignited and burned (Govender *et al.*, 2006).

Fire seasonality is directly linked to fire intensity (Brooks *et al.*, 2004), with more intense fires occurring during the dry season as a result of low fuel moisture content. Dry fuel loads result in more intense fires; however, grasslands were found to be of a relatively low intensity because of the fuel type (grasses) compared to thicket or forests, ranging from 5 up to approximately 1700 MW per pixel. Govender *et al.*, (2006) had similar findings in the Kruger National Park, South Africa, with regards to fireline intensity; however, it must be noted that the although both studies were based around grasslands, the vegetation type and species composition is different. Whilst FRP varies, it can indicate fuel type and is closely linked to pyromes described by Archibald *et al.*, 2013.

5.3 Landscape response to fires in the upper Tsitsa

Due to the nature of the UTRC, with much of the communally managed land being severely degraded (gullyng, slope erosion and lack of vegetative cover) compared to the commercially managed land, this study assessed a variety of soil characteristics in the more natural parts of the upper catchment in order to determine whether fire frequency is having a negative impact on soils and some aspects of vegetation cover. In order to complete an assessment of the soils, fire regimes were identified in the first portion of the project via a desktop study involving extensive remote sensing and GIS work. Assessing the impacts of fire frequency on soil properties was conducted through measurements in the field and

through laboratory work with soil samples that were collected during fieldwork. Overall, it was found that fires are altering certain soil characteristics in the UTRC and this will be discussed in depth below.

5.3.1 Response of soil characteristics to fires

The response of soils to burning can vary significantly depending on multiple different factors such as parent material, soil particle size, fire frequency, intensity, vegetation cover and climate (Doer *et al.*, 1996; DeBano, 2000; Magomani & van Tol, 2018). The PCA that was performed on the data collected on soils during the study showed that soil surface hardness, water repellency and above-ground biomass were influenced by fire frequency and that pH and soil organic content were influenced more by parent material (Figure 4.19). The variables that were directly affected by fire frequency can result in negative changes in soil characteristics, altering local geomorphological and hydraulic processes, making landscapes more vulnerable to degradation and erosion and this will be further discussed in this section. However, I will first discuss the variables that were not found to interact with fire frequency.

Analysis by ANOVA showed that there was a significant difference between mudstone and sandstone sites at a high fire frequency and on average, mudstone soils had a higher pH than sandstone soils (Figure 4.17). Mudstone and sandstone soils exposed to a high fire frequency exhibited pH values averaging 4,9 and 3,9 respectively. It was also found that when exposed to a high frequency of fire, the pH in mudstones was 13 % higher than sites exposed to a low fire frequency. The findings on mudstone sites supported what has been found in other literature, that states that pH levels in soils increase with fire frequency and intensity (Thomas *et al.*, 2000; Vega *et al.*, 2013). However, the findings for the sandstone sites are not supported by other studies analysed in the literature, and this could be explained by the low- moderate intensity fires that the UTRC grasslands experience. According to Vega *et al.*, (2013), soil pH should increase with the level of burn severity, but there is no change in pH in soils that are exposed to low and moderate-intensity fires. This could explain why there is no change in pH and that changes in pH are a function of fire severity and soil type rather than fire frequency.

Another variable that had a weak relationship with fire frequency was soil organic content according to the PCA (Figure 4.19). It was found through statistical analysis that there was no relationship between fire frequency and SOM, however, sandstone soils had a significantly higher organic content than mudstone soils (approximately double the amount). These findings suggest that SOM is controlled by soil type and vegetation and fire severity rather than fire frequency (Figure 4.16), which is supported by literature that states that SOM will be altered to different degrees at higher intensity fires, which the UTRC does not experience regularly in its grasslands (De la Rosa *et al.*, 2012). Soils with a higher organic matter content tend to be more water repellent due to the release of hydrophobic compounds from vegetation during decomposition and burning, however there are other influencing factors that need to be accounted for, such as soil particle size, soil parent material, soil surface crusting, fire intensity and fire frequency (Jordán *et al.*, 2013).

WDPT was used as a proxy for water repellency, and this study found that there was a significant interaction between WDPT and fire frequency (Table 4.3; $p < 0.01$), with water repellency increasing from zero seconds up to fifteen seconds. A regression analysis also showed a moderately strong positive relationship between the two variables ($r^2 = 55.7\%$; Figure 4.12). An increase in fire frequency resulted in an increase in WDPT and therefore soil water repellency. There was also a significant difference between the soil types (Table 4.3; $p = 0.031$), with mudstone sites having on average, a higher WDPT and water repellency than sandstone sites. The results were unexpected, as literature stated that, soil water repellency is generally associated with coarse-textured, sandy soils (Crockford *et al.*, 1991; DeBano, 1991; Doer & Shakesby, 2000). The reasoning behind this is that a limited supply of hydrophobic substances would sufficiently coat larger soil particles due to their smaller surface area per unit volume compared to clay soils with finer soil texture and larger surface area per unit volume (DeBano, 1971; DeBano, 1991). Using the above premise, sandy soils should be more susceptible to water repellency than clay soils, but this is not the case in the UTRC. Grasses do not typically contain high amounts of hydrophobic chemicals would coat soil particles and prevent water infiltration, and the water repellency found in this study could be due to soil parent material and characteristics. Low water repellency seen in sandstone soils during this study could also be attributed to their high organic content, as vegetation and roots can increase water infiltration by creating natural channels for water to flow. Further statistical analyses were performed to explain the interaction between fire frequency, particle size and WDPT, and it was found that the median soil particle size was smaller at sites that had a high fire frequency than sites with a low fire frequency (Figure 4.20). The results show that areas in the UTRC that experience a high frequency of fires have finer textured soils and a higher water repellency. Whilst this is not the norm, further research found that soils with a considerable clay content can exhibit extreme water repellency. Water repellency can occur when clay particles form aggregates, reducing the effective surface area of particles that are coated by hydrophobic layers (Crockford *et al.*, 1991; Dekker & Ritsema, 1996). Aggregation of soils may explain why this study found that soil particle size did not have any significant effect on water repellency (Figure 4.12), whilst there was a significant interaction between fire frequency and WDPT.

Although there was a strong relationship between fire frequency and water repellency, the shift in WDPT values were relatively low and had relatively little effect on the landscape through increased runoff from frequently burned areas. Although WDPT increased from low to high fire frequency by up to ten times, the average WDPT classification for the soils was wettable (0-5 s) and slightly water repellent (5-60 s) (Table 3.4), with no WDPT test lasting over 35 seconds, making the soils fairly wettable. The presence of water repellency means that soil particles are being exposed to hydrophobic substances, but the exact chemical composition responsible for the formation of hydrophobic layers differs, and is generally made up of a mixture of organic compounds that accumulate on and between soil particles from waxy leaves, roots, fungi, microbes and decomposing organic matter in or on the

surface of soils (Dekker *et al.*, 2009). The origin of these compounds is important as some are more hydrophobic than others, for example, eucalypts and pines contain oils that are known to be highly hydrophobic and cause severe water repellency (Zavala *et al.*, 2009). Although this study assessed fire regimes throughout the UTRC, its focus was on the effects of fires on grassland landscapes. The presence of low-intensity fires and lack of waxy/oily volatiles in grasses could explain the occurrence of low severity water repellency found during this research. Along with water repellency, the formation of a hard crust through weathering, raindrop compaction, fires and livestock can contribute significantly to erosive processes (Laker, 2004).

Fire frequency was found to have a significant influence on soil surface hardness, regardless of soil type (Figure 4.14; $p < 0.001$). Soil surface hardness relates to the crust that can form on soils from exposure to fires and subsequent raindrop impact and excessive trampling by livestock, most often seen in overgrazed areas and can limit infiltration via compaction (Laker, 2004; Simorangkir, 2007; Durán Zauzo & Rodríguez Pleguezuelo, 2008; Malvar *et al.*, 2017). Soil compaction and decreased infiltration ultimately lead to increased erosion potential through excess overland flow. Compacted soils coupled with increased water repellency caused by frequent fires would increase erosion during high rainfall events as water infiltration rates are reduced.

5.3.2 Soil recovery post-fire

The impacts of fires on soil properties were assessed during this study in order to determine whether fire frequency is affecting erosion potential and altering hydrological and geomorphological processes. While it is important to measure the impacts that fires and fire frequency have on soils for future management implications, one needs to consider whether the impacts on soils are permanent and severely damaging to the landscape. Therefore, measurements were recorded at specific sites that burned in 2017 (one year prior to sampling) and 2018 (weeks before sampling) to assess the immediate impacts and the recovery of soil properties post-fire. Whilst repeated measurements and monitoring would be the ideal way to determine change over time of the landscape post-fire, this study tried to identify specific characteristics that could be used to indicate post-fire recovery. Sampling sites for this portion of the study were not exposed to frequent fires and therefore did not exhibit the same soil characteristics as soils that were exposed to a high FRI. It was found that there were very minor changes in soil properties after a one-year period, specifically, for pH which was expected due to the occurrence of low-intensity fires in the study area and because it is heavily influenced by soil type and vegetation (Vega *et al.*, 2013). More noticeable changes occurred in soil surface hardness, WDPT and SOM after measurements were taken one year after burning. On average, soil surface hardness decreased after one year, and mudstone was found to be significantly harder than sandstone immediately after burning (Figure 4.20), which is consistent with the findings of this study, and the findings are in line with the literature (Malvar *et al.*, 2017). Similarly, water repellency was also significantly higher on mudstones than sandstones. However, length of time between sampling and the last burn had no effect on WDPT

(Table 4.10). It was expected to find a higher level of water repellency in sites that burned in 2018 than in 2017 as other studies have found that water repellency generally increases immediately after fires and decreases over time (Jordán *et al.*, 2013). Whilst there was a significant difference between mudstone and sandstone soils, there was no evidence of severe water repellency, with all the soils being wettable or slightly water repellent. The findings of this study suggest that there is minimal immediate impact on infiltration post fires at a low fire frequency. This could be attributed to low fire intensity due to low fuel loads and the small number of hydrophobic compounds present in grasses (Jordán *et al.*, 2013; Doerr *et al.*, 2000). Studies in pine and eucalyptus forests found a high degree of water repellency was caused by the high intensity of fires linked to the vegetation type, as well as the waxy, hydrophobic compounds released by the forests during burning (Doer *et al.*, 1996; Prosser & Williams, 1998; Rodríguez-Alleres *et al.*, 2012) which are not characteristic of grassland vegetation.

SOM was another variable that showed noteworthy change over the two years, with sandstone sites that burned in 2018 containing a significantly higher amount of SOM than the sites that burned in 2017 (Figure 4.20). This finding is not supported by the literature as the recently burned sites were expected to contain lower SOM than sites that burned one year prior as organic content should decrease with increasing surface temperature (Keeley, 2009; Wieting *et al.*, 2017). The results that were found in this study may be a result of the low-intensity fires that occur in the UTRC grasslands and natural variability in the landscape that might not be well defined with the small sample size and time for space substitute.

Overall, the variables show that one-year post-fire, the soils are regaining their ‘natural’ properties and the immediate effects of fire are low in the UTRC. The minimal effects of fire can be attributed to the low-intensity fires that occur in grasslands, low hydrophobic compound release by grasses and the fact that the vegetation and landscape are resilient to fires as they are a natural ecosystem driver in the catchment.

5.3.3 Vegetation fire interaction

Vegetation plays a large role in maintaining landscape functions and the prevention of erosion by providing cover against splash erosion and holding soil particles together with roots, improving water infiltration in the soils, slowing the flow of water across the soil surface and shielding soils from erosive rainfall (Vogl, 1979; Mucina & Rutherford, 2006). Above-ground biomass was measured during this study to assess how fire frequency is interacting with vegetation in grasslands and ultimately influencing erosive processes (Vieira *et al.*, 2015; Bond & Keane, 2017). This study found that there was no significant difference between above-ground biomass at sites with high and low fire frequencies. This result was expected as grasslands have adapted alongside fires and grazing and are extremely resilient to these impacts (Trollope *et al.*, 2002; Mucina & Rutherford, 2006). Grasses will regrow stems and leaves after defoliation by fire or grazing by drawing on undisturbed carbohydrate reserves in plant bases (Vogl, 1979). Vegetation regrowth is successful in healthy grassland systems, but excessive and

frequent disturbances can deplete carbohydrate reserves as in the case of overgrazing or annual burning (Mucina & Rutherford, 2006).

A broader observation made by this research was that the commercially managed land was predominantly made up of sour veld, characterised by high biomass and frequent fires (Mucina & Rutherford, 2006; Figure 4.8). Appropriately implemented livestock grazing and controlled burning (D. Sephton, personal communication, 10 October 2018), are resulting in frequent large fires which support high grass production and vegetation cover (Figure 1.5) and low rates of soil erosion (Herd-Hoare, 2019). However, the communally managed areas are dominated by sweeter veld which is more palatable to livestock. Poor management of livestock is resulting in overgrazing and the continuous heavy use of natural resources is decreasing available biomass for burning and creating a fragmented and degraded landscape (Le Roux *et al.*, 2008; Fica, 2011; Mararakanye & Le Roux, 2012). Decreased fuel load densities and increased fragmentation are resulting in infrequent, small and low-intensity fires (Figure frequency; size; intensity). Mucina and Rutherford, (2006) stated that communal grazing lands, particularly in the former Transkei, are stocked at 3 to 4 times the recommended stocking rate for commercial farms which would negatively affect vegetative growth and degrade the landscape over time (SANBI, 2014). Overgrazing can alter plant size and density and negatively affect seed production which could explain the poor NDVI values in the communally managed land (Figure 1.5). Heavy grazing in the former Transkei has also been found to have negative effects on species richness and composition, resulting in less palatable grasses (Owen-Smith & Danckwerts, 1997), possibly driving communities to graze livestock in specific areas. It is thought that areas with a higher fire frequency in communally managed grasslands are where livestock have been excluded, resulting in higher biomass and fuel availability.

Vegetation type is playing a large role in controlling fire regimes in the UTRC, as shown in the difference between the size and frequency of fires in the commercial and communally managed land. Grassland vegetation (sweet/sour veld) seems to be a function of underlying geology and soil parent material in the UTRC, with sandstones in the commercial areas supporting sour veld (high biomass) and the sandstones in the communal areas supporting sweet veld (lower biomass), which is more palatable to livestock. Similar results were found by Fica, (2011), who's research found that two adjacent quaternary catchments supported different levels of biomass-based even though they had similar management strategies and climatic conditions. This leads one to believe that geology is playing a larger role in determining vegetation type and ultimately fire regimes than land management alone.

5.4 Synthesis of fire regimes and their effects on the UTRC

It is difficult to determine the natural fire regimes in the UTRC due to the large scale anthropogenic influence that is already present, however, we know from literature that fires in grasslands are extremely common and they experience a high frequency of fires with a FRI of 1-3 years (Govender *et al.*, 2006; Mucina & Rutherford, 2006; Archibald *et al.*, 2013). Research found that there were two different fire regimes present and these were closely linked to the type of land management. Commercially managed grasslands experienced frequent fires (FRI 1-3 years) and that this number was consistent across the study area, with the highest frequency of fires being around plantations (Figure 4.8), whilst communally managed grasslands had inconsistent fire patterns with an average FRI of 4-5 years, with some areas not supporting burning. We can't assume that these areas didn't burn at all as the data had gaps due to limitations using remote sensing. However, this study gives a general sense of the fire regimes present in the UTRC over the past 30 years. More frequent fires in the commercially managed areas of the catchment resulted in a larger annual burned area than the communal areas. Less frequent fires in the communally managed areas are likely due to low available biomass and landscape fragmentation due to overgrazing. Throughout the catchment there is a decreasing annually burned area trend over the 30-year record, with evidence of decreased maximum and average burn scar size (Figure 4.5; Table 4.2). The findings of this study are in line with Archibald *et al.*, (2013) as it was found that southern African grasslands experience frequent (1-3 years), intense, large and frequent, cool, small fires, with a possible shift to the latter with the increase in plantations and agriculture, shifting away from intensive livestock production in the commercially managed areas. This research determined that the commercially managed areas in the UTRC are experiencing a FIL pyrome and the communally managed areas are experiencing an FCS pyrome as described by Archibald *et al.*, 2013. Fire intensity was variable throughout the catchment, with relatively low-intensity fires burning in the communally managed grasslands and slightly higher intensity fires burning in the commercially managed areas. The difference in fire intensity between the two management areas is likely due to the effect of fuel load, with the commercial areas having higher and more consistent fuel loads than the communal areas. Another notable remark is that high-intensity fires were detected near plantations, likely due to the burning of slash piles before planting new trees.

One of the aims of this study was to assess the impacts of fire frequency on soil properties and land degradation, however, it was found through literature that fire intensity plays a larger role in altering soil characteristics (Vogl, 1979; Doerr *et al.*, 1996; DeBano, 1991; DeBano, 2000; Cawson *et al.*, 2016). The findings of this study are summarised in a flow chart which illustrates the influence of topography, climate, management and most importantly geology determining fuel loads and fire regimes (Figure 5.1). For soils to be negatively altered by fires, they need to be exposed to intense heat for a prolonged period of time, and this is very rare in grasslands. However, a high fire frequency and low-moderate intensity fires did alter soil properties to some extent. It was found that soil surface hardness and water

repellency both increased with the number of burns, indicating a positive trend between the two variables (Figure 4.11; Figure 4.14). Soil surface properties such as soil surface hardness and water repellency are important when studying post-fire hydrology as they can both result in decreased infiltration rates, enhanced runoff and erosion, particularly once the vegetative cover has been removed (Cawson *et al.*, 2016; Alcañiz *et al.*, 2017; García-Comendador, 2017). Whilst soil surface hardness and water repellency are the two biggest factors that increase erosion potential, their impacts on the landscape are relatively low, with soils being classified as slightly water repellent and wettable. Vegetation cover also plays an important role in preventing runoff, and it was found that above-ground biomass was not being significantly affected by fire frequency in the commercially managed areas of the catchment. This again comes down to the severity of disturbances within grasslands and it seems that low and moderate-intensity fires play less of a role in landscape degradation and erosion than overgrazing (SANBI, 2014). In the bigger picture, fires are temporarily removing vegetative cover and altering soil properties, but not to the degree that was previously thought. Grass species are not sensitive to fire intensities as their growing points are located close to the soil surface, away from the release of heat energy (Trollope *et al.*, 2002). The excessive land degradation in the communally managed areas cannot be linked with fires alone, and may be a function of geology, climate and human pressure (road networks, vegetation fragmentation, trampling, etc.). Overgrazing is also evident in the communal areas, forming compacted soils and causing vegetation reduction and fragmentation and opportunities for enhanced runoff. Vegetation type and biomass ultimately determines the natural fire regimes that would be present in a landscape that can be altered by human influence, via change in land cover (e.g. vegetation fragmentation, housing areas, plantations, etc.) and land management (e.g. planned burning, overgrazing, etc.).

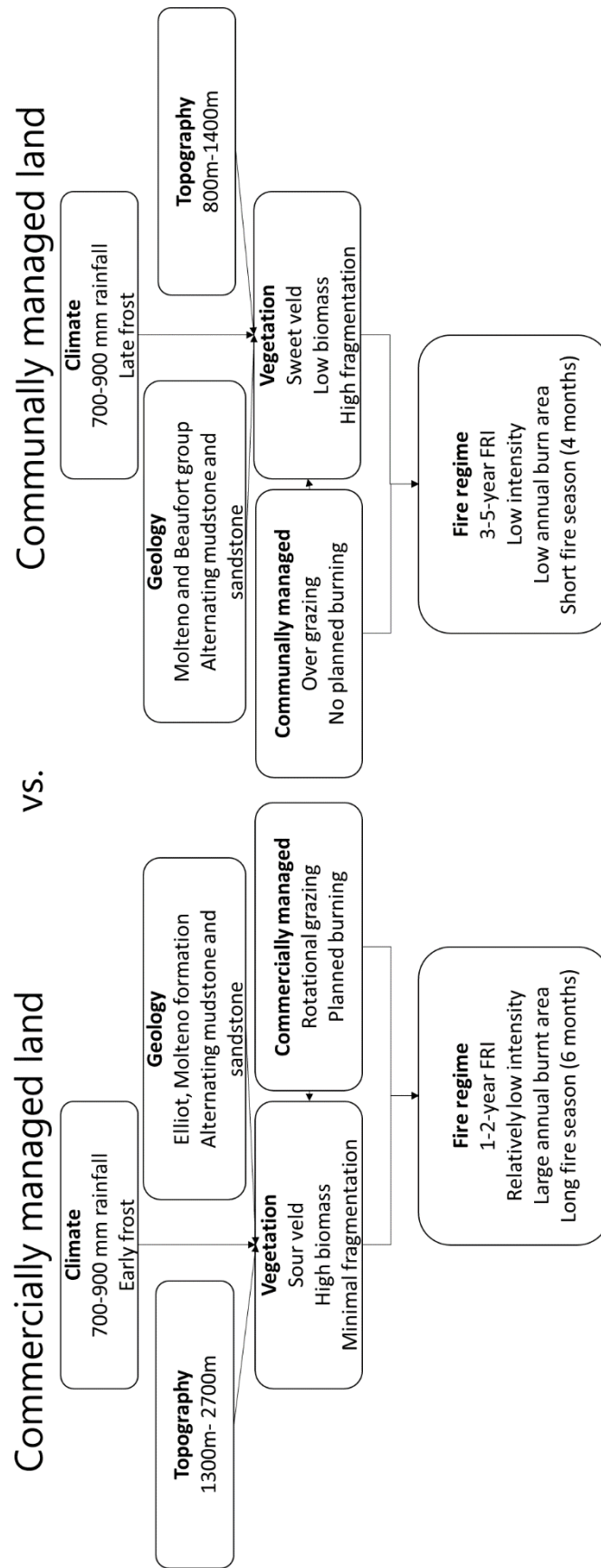


Figure 5.1: A summary flow diagram showing the differences in environmental and anthropogenic factors influencing the catchment, focusing on commercial and communitally managed areas and the resulting fire regimes.

6. Conclusion

This study successfully developed and tested a method to assess fire frequency, size, intensity and seasonality over the past 30 years using remotely sensed data. The data were used to assess fire patterns and changes due to various environmental and anthropogenic factors. A variety of remotely sensed data were available; however, it was found that a combination of multispectral Landsat imagery and MODIS hotspot and radiative data were most efficient in describing the fire regimes present in the UTRC. The temporal scale and spatial resolution of the Landsat data made it reliable and accurate for catchment scale research regarding burn scar size and frequency (Hudak & Brockett, 2004). Limitations with Landsat data were a relatively long revisit period (16 days) and cloud cover, obscuring catchment wide mapping of burn scars. MODIS and VIIRS short return period and thermal sensor data provided a strong grounding for research as its active fire detection algorithm proved beneficial, detecting fire hotspots with time/date capture and FRP (Archibald *et al.*, 2013; Giglio *et al.*, 2006; Chuvieco *et al.*, 2008). The remotely sensed data proved its ability to accurately detect and describe fire regimes and their characteristics within the UTRC.

This research found that there were two distinct fire regimes present in the UTRC. It was found that commercially managed land within the UTRC experienced a high frequency of large, low intensity fires from May to November every year. The communally managed land experienced a lower frequency of small, low intensity fires from June to October (Archibald *et al.*, 2013). The commercial areas have a largely natural fire regime, whereas the communal area of the catchment has a slightly altered fire regime, partly due to human-related pressures, vegetation cover fragmentation, overgrazing, underlying rock types and associated soils. Vegetation type and underlying geology plays an integral role in determining fire regimes, as is the case in the UTRC. Large frequent fires (1-2 years) occur in high biomass, sour veld areas of the commercial part of the catchment, and smaller less frequent fires (3-5 years) occur in the fragmented, low biomass sweet veld of the communal areas.

Fire frequency and intensity were found to be altering landscape and soil physical and chemical characteristics in the UTRC. Fires were temporarily reducing above ground biomass whilst increasing water repellency and soil surface hardness. Removal of biomass and decrease in infiltration has the potential to result in increased runoff and erosion. However, research showed that the degree at which fires affected soil properties were minimal. This is due to the nature of the relatively low biomass of grasslands, supporting low intensity fires, and the resilience of the landscape and fire adapted grasslands. There is a high degree of landscape degradation in the communal areas of the UTRC, and it was assumed that fires were playing a role in this process in the distant past. However, the findings of this research suggest that land management and soil type is affecting vegetation cover and supporting overgrazing in the communally managed areas, increasing landscape fragmentation and degradation rather than fires being a driver of erosion over the past 30 years.

Ultimately, this research found that over the past 30 years, the landscape degradation in the communal areas is not a direct function of fires and is rather due to the surrounding geology, associated erodible soils, overgrazing and human impacts. Whilst much of the communally managed land is recognized as being severely degraded, and it would take a long time to restore the landscape to a level where it could support a more natural fire regime. However, an increase in fire frequency would pose risk to local communities due to the density of people in the area and the associated risks that come with more frequent fires. It is therefore unlikely and not advisable to move to a more frequent fire regime in the communal areas in the near future due to the associated risks and low fuel availability.

It is clear that one needs to take a holistic approach when assessing fire regimes and their impacts on the environment. Fires are an essential part of many systems, however, to suppress or ignite fires in the wrong place or at the wrong time can end in catastrophe. In a natural system, fires will burn at different intervals determined by climate, vegetation type, biomass and ignition sources. Grasslands have coexisted with frequently occurring fires for hundreds of years (Vogl, 1979; Bowman *et al.*, 2011) and fires play a vital role in the functioning of these ecosystems. One needs to consider burnt and recovering grasslands as equally important steps in an endless cycle that is part of a larger continuous process maintaining grassland health and vigour. Using this premise, the assessment of damage and recovery post-fire may seem uncalled for, however one can't deny the negative impacts fires can pose if natural fire regimes are altered by anthropogenic activities (Vogl, 1979; Bowman *et al.*, 2011; Archibald *et al.*, 2013; Alcañiz *et al.*, 2017). With so many factors influencing fire regimes, whether it be climate, vegetation, or anthropogenic influence, they need to be recognised as dynamic, multidimensional systems where there is no "one size fits all" approach.

6.1 Limitations and recommendations for future research

Due to the large temporal scale of this project, one of the main limitations was gaining access to freely available, cloud-free, unobscured satellite imagery during and directly after the fire season. Unobscured images from early in the fire season did not show the full extent of fires that burned in that specific year and burn scars in images from the end of the season after the first rains proved difficult to identify due to the rapid regrowth of vegetation. Therefore, several years were left out of the study due to partial or full cloud cover on available images which would have skewed or provided unreliable data as seen in Appendix 2. Future research could make use of satellites with a shorter revisit time or utilise high-resolution products that may come with a cost. Alternatively, one could assess fires using drone imagery, however this would require the equipment and research to be done on a smaller scale and at a higher cost.

With regards to work in the field, the large spatial scale and the nature of fire seasonality limited the sampling period where soil samples could be collected and analysed, with very specific areas of the catchment being sampled in favour of others. The fact that large areas of the catchment are less accessible by road or foot, given the steep terrain also limited field measurements and data collection. Future research could take a broader approach and assess a larger extent of the catchment, allowing for a better idea of how soils and vegetation are responding to specific fire regimes and patterns (i.e. fire breaks around plantations or abandoned farmlands areas near communal villages). When analysing data, the unique and altered fire regime in the communally managed areas made it difficult to accurately measure the effects of fire frequency on soil characteristics and compare them to the commercially managed land. The way one could go about this in the future is to potentially set up plots to be burned at different intervals and take measurements before and after fires, allowing for a more in-depth idea into how fires are altering soil properties with different geologies, vegetation types and grazing pressure.

The findings of this study could be used to develop fire and grazing guidelines for the communally managed grasslands. This would require community input and willingness to work with researchers and managers, as well as understanding and incorporating traditions and customs around fires and keeping livestock.

7. References

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Gusha, B. 2019, *Communal grasslands, fire and grazing*, Maclear

8. Appendices

Appendix 1: Summary table of Landsat imagery products used for fire scar extraction during the study.

Collection Date	Satellite	Path	Row(s)	Product identifier(s)
01/11/2017	Landsat 8	169	82	LC08_L1TP_169082_20171101_20171109_01_T1
11/10/2015	Landsat 8	169	82	LC08_L1TP_169082_20151011_20170403_01_T1
09/11/2014	Landsat 8	169	82	LC08_L1TP_169082_20141109_20170417_01_T1
05/10/2013	Landsat 8	169	82	LC08_L1TP_169082_20131005_20170429_01_T1
03/09/2007	Landsat 5	169	81	LT05_L1TP_169081_20070903_20161112_01_T1
			82	LT05_L1TP_169082_20070903_20161111_01_T1
03/11/2006	Landsat 5	169	81	LT05_L1TP_169081_20061103_20161118_01_T1
			82	LT05_L1TP_169082_20061103_20161118_01_T1
23/10/2005	Landsat 5	169	81	LT05_L1TP_169081_20050929_20161124_01_T1
			82	LT05_L1TP_169082_20050929_20161125_01_T1
17/08/2001	Landsat 5	169	81	LT05_L1TP_169081_20010817_20161210_01_T1
			82	LT05_L1TP_169082_20010817_20161210_01_T1
09/10/2000	Landsat 5	169	81	LT05_L1TP_169081_20001001_20161213_01_T1
			82	LT05_L1TP_169082_20001001_20161213_01_T1
28/08/1999	Landsat 5	169	81	LT05_L1TP_169081_19990828_20161218_01_T1
			82	LT05_L1TP_169082_19990828_20161216_01_T1
13/11/1998	Landsat 5	169	81	LT05_L1TP_169081_19981113_20161222_01_T1
			82	LT05_L1TP_169082_19981113_20161220_01_T1
22/08/1997	Landsat 5	169	81	LT05_L1TP_169081_19970822_20161230_01_T1
			82	LT05_L1TP_169082_19970822_20161230_01_T1

22/10/1996	Landsat 5	169	81	LT05_L1TP_169081_19961022_20170103_01_T1
			82	LT05_L1TP_169082_19961022_20170103_01_T1
02/09/1995	Landsat 5	169	81	LT05_L1TP_169081_19950902_20170106_01_T1
			82	LT05_L1TP_169082_19950902_20170106_01_T1
15/09/1994	Landsat 5	169	81	LT05_L1TP_169081_19940915_20170112_01_T1
			82	LT05_L1TP_169082_19940915_20170112_01_T1
30/10/1993	Landsat 5	169	81	LT05_L1TP_169081_19931030_20170117_01_T1
			82	LT05_L1TP_169082_19931030_20170116_01_T1
09/09/1992	Landsat 5	169	81	LT05_L1TP_169081_19920909_20170123_01_T1
			82	LT05_L1TP_169082_19920909_20170121_01_T1
22/08/1991	Landsat 5	169	81	LT05_L1TP_169081_19910822_20170125_01_T1
			82	LT05_L1TP_169082_19910822_20170125_01_T1
01/09/1989	Landsat 5	169	81	LT05_L1TP_169081_19890901_20170201_01_T1
			82	LT05_L1TP_169082_19890901_20170201_01_T1
18/08/1984	Landsat 5	169	81	LT05_L1TP_169081_19840818_20170220_01_T1
			82	LT05_L1TP_169082_19840818_20170220_01_T1

Appendix 2: A summary of the years that total burned area and rainfall data were available.

Year	Total burned area (km²)	Total rainfall (mm)
1984	657	769,31
1985		905,12
1986		788,74
1987		767,21
1988		1045,27
1989	667,48	923,25
1990		687,16
1991	133,95	822,02
1992	530,35	502,71
1993		802,24
1994	522,29	738,79
1995	450,17	939,54
1996	160,75	966,12
1997	626,62	845,78
1998	30,46	1037,05
1999	433,51	753,98
2000	546,41	1143,92
2001	383,01	1085,71
2002		1061,88
2003		710,72
2004		975,41
2005	301	610,18
2006	102,66	947,16
2007	425,47	668,68
2008		703,1
2009		756,6
2010		823,5
2011		953,8
2012		904,1
2013	302,64	1003
2014	90,41	1063
2015	125,82	685
2016		853,5
2017	62,2	702
2018	220,53	725

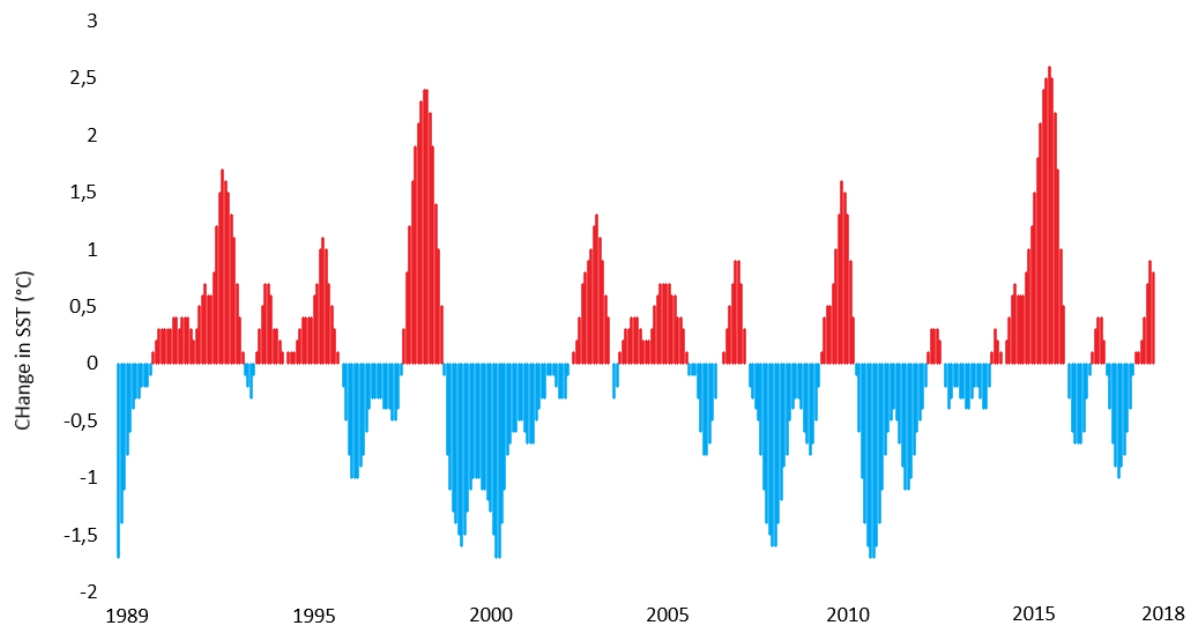
Appendix 3: Available bands, wavelengths and image resolution on Landsat 4-5.

Landsat 4-5		Wavelength	Resolution
	Bands	(micrometers)	(meters)
Thematic Mapper (TM)	Band 1 - Blue	0.45-0.52	30
	Band 2 - Green	0.52-0.60	30
	Band 3 - Red	0.63-0.69	30
	Band 4 - Near Infrared (NIR)	0.76-0.90	30
	Band 5 - Shortwave Infrared (SWIR) 1	1.55-1.75	30
	Band 6 - Thermal	10.40-12.50	120* (30)
	Band 7 - Shortwave Infrared (SWIR) 2	2.08-2.35	30

Appendix 4: Available bands, wavelengths and image resolution on Landsat 8

Landsat 8		Wavelength	Resolution
	Bands	(micrometers)	(meters)
Operational	Band 1 - Ultra Blue (coastal/aerosol)	0.435 - 0.451	30
Land Imager (OLI) and Thermal Infrared Sensor (TIRS)	Band 2 - Blue	0.452 - 0.512	30
	Band 3 - Green	0.533 - 0.590	30
	Band 4 - Red	0.636 - 0.673	30
	Band 5 - Near Infrared (NIR)	0.851 - 0.879	30
	Band 6 - Shortwave Infrared (SWIR) 1	1.566 - 1.651	30
	Band 7 - Shortwave Infrared (SWIR) 2	2.107 - 2.294	30
	Band 8 - Panchromatic	0.503 - 0.676	15
	Band 9 - Cirrus	1.363 - 1.384	30
	Band 10 - Thermal Infrared (TIRS) 1	10.60 - 11.19	100 * (30)
	Band 11 - Thermal Infrared (TIRS) 2	11.50 - 12.51	100 * (30)

Appendix 5: Changes in sea surface temperature due to ENSO from 1989 to present.



Appendix 6: DPM calibration table

Mean DPMH (cm)	Standing crop (kg ha ⁻¹)	Mean DPMH (cm)	Standing crop (kg ha ⁻¹)	Mean DPMH (cm)	Standing crop (kg ha ⁻¹)	Mean DPMH (cm)	Standing crop (kg ha ⁻¹)	Mean DPMH (cm)	Standing crop (kg ha ⁻¹)
0.2	0.0	12.2	3 448.5	24.2	5 575.1	36.2	6 730.1	48.2	7 087.6
0.4	2.1	12.4	3 490.8	24.4	5 605.5	36.4	6 741.3	48.4	7 088.9
0.6	17.2	12.6	3 532.9	24.6	5 635.7	36.6	6 752.3	48.6	7 090.0
0.8	52.1	12.8	3 574.6	24.8	5 665.8	36.8	6 763.0	48.8	7 091.1
1.0	104.2	13.0	3 616.0	25.0	5 695.9	37.0	6 773.6	49.0	7 091.9
1.2	168.6	13.2	3 657.1	25.2	5 725.8	37.2	6 784.0	49.2	7 092.7
1.4	241.0	13.4	3 697.8	25.4	5 755.6	37.4	6 794.2	49.4	7 093.3
1.6	318.3	13.6	3 738.3	25.6	5 785.2	37.6	6 804.2	49.6	7 093.8
1.8	398.3	13.8	3 778.5	25.8	5 814.8	37.8	6 814.0	49.8	7 094.2
2.0	479.5	14.0	3 818.4	26.0	5 844.3	38.0	6 823.6	50.0	7 094.5
2.2	561.1	14.2	3 858.0	26.2	5 881.0	38.2	6 833.1	50.2	7 094.6
2.4	642.3	14.4	3 897.4	26.4	5 904.2	38.4	6 842.3	50.4	7 094.6
2.6	722.8	14.6	3 936.4	26.6	5 927.1	38.6	6 851.3	50.6	7 094.5
2.8	802.3	14.8	3 975.2	26.8	5 949.7	38.8	6 860.2	50.8	7 094.3
3.0	880.6	15.0	4 013.8	27.0	5 972.0	39.0	6 868.9	51.0	7 093.9
3.2	957.5	15.2	4 052.1	27.2	5 994.1	39.2	6 877.4	51.2	7 093.5
3.4	1 033.2	15.4	4 090.1	27.4	6 015.9	39.4	6 885.7	51.4	7 092.9
3.6	1 107.5	15.6	4 127.9	27.6	6 037.4	39.6	6 893.8	51.6	7 092.2
3.8	1 180.5	15.8	4 165.5	27.8	6 058.6	39.8	6 901.7	51.8	7 091.4
4.0	1 252.1	16.0	4 202.9	28.0	6 079.6	40.0	6 909.5	52.0	7 090.4
4.2	1 322.4	16.2	4 240.0	28.2	6 100.3	40.2	6 917.1	52.2	7 089.4
4.4	1 391.5	16.4	4 276.8	28.4	6 120.8	40.4	6 924.5	52.4	7 088.2
4.6	1 459.3	16.6	4 313.5	28.6	6 141.0	40.6	6 931.7	52.6	7 087.0
4.8	1 526.0	16.8	4 350.0	28.8	6 160.9	40.8	6 938.7	52.8	7 085.6
5.0	1 591.5	17.0	4 386.2	29.0	6 180.6	41.0	6 945.6	53.0	7 084.1
5.2	1 655.9	17.2	4 422.2	29.2	6 200.0	41.2	6 952.3	53.2	7 082.5
5.4	1 719.3	17.4	4 458.0	29.4	6 219.2	41.4	6 958.9	53.4	7 080.8
5.6	1 781.6	17.6	4 493.7	29.6	6 238.1	41.6	6 965.2	53.6	7 079.0
5.8	1 843.0	17.8	4 529.1	29.8	6 256.8	41.8	6 971.4	53.8	7 077.1
6.0	1 903.4	18.0	4 564.3	30.0	6 275.2	42.0	6 977.4	54.0	7 075.0
6.2	1 962.8	18.2	4 599.4	30.2	6 293.3	42.2	6 983.3	54.2	7 072.9
6.4	2 021.5	18.4	4 634.2	30.4	6 311.3	42.4	6 989.0	54.4	7 070.7
6.6	2 079.2	18.6	4 668.9	30.6	6 328.9	42.6	6 994.5	54.6	7 068.3
6.8	2 136.2	18.8	4 703.4	30.8	6 346.4	42.8	6 999.9	54.8	7 065.9
7.0	2 192.4	19.0	4 737.7	31.0	6 363.5	43.0	7 005.1	55.0	7 063.4
7.2	2 247.8	19.2	4 771.8	31.2	6 380.5	43.2	7 010.2	55.2	7 060.7
7.4	2 302.5	19.4	4 805.8	31.4	6 397.2	43.4	7 015.1	55.4	7 058.0
7.6	2 356.5	19.6	4 839.6	31.6	6 413.7	43.6	7 019.8	55.6	7 055.2
7.8	2 409.8	19.8	4 873.2	31.8	6 429.9	43.8	7 024.4	55.8	7 052.2
8.0	2 462.5	20.0	4 906.6	32.0	6 445.9	44.0	7 028.8	56.0	7 049.2
8.2	2 514.5	20.2	4 939.9	32.2	6 461.6	44.2	7 033.1	56.2	7 046.1
8.4	2 565.9	20.4	4 973.1	32.4	6 477.2	44.4	7 037.2	56.4	7 042.8
8.6	2 616.7	20.6	5 006.0	32.6	6 492.5	44.6	7 041.1	56.6	7 039.5
8.8	2 667.0	20.8	5 038.9	32.8	6 507.5	44.8	7 044.9	56.8	7 036.1
9.0	2 716.7	21.0	5 071.5	33.0	6 522.4	45.0	7 048.6	57.0	7 032.6
9.2	2 765.9	21.2	5 104.0	33.2	6 537.0	45.2	7 052.1	57.2	7 029.0
9.4	2 814.5	21.4	5 136.4	33.4	6 551.4	45.4	7 055.5	57.4	7 025.3
9.6	2 862.6	21.6	5 168.6	33.6	6 565.6	45.6	7 058.7	57.6	7 021.6
9.8	2 910.3	21.8	5 200.7	33.8	6 579.5	45.8	7 061.8	57.8	7 017.7
10.0	2 957.5	22.0	5 232.6	34.0	6 593.2	46.0	7 064.7	58.0	7 013.8
10.2	3 004.2	22.2	5 264.4	34.2	6 606.7	46.2	7 067.5	58.2	7 009.7
10.4	3 050.4	22.4	5 296.1	34.4	6 620.0	46.4	7 070.1	58.4	7 005.6
10.6	3 096.2	22.6	5 327.6	34.6	6 633.1	46.6	7 072.6	58.6	7 001.4
10.8	3 141.6	22.8	5 359.0	34.8	6 646.0	46.8	7 075.0	58.8	6 997.1
11.0	3 186.6	23.0	5 390.3	35.0	6 658.6	47.0	7 077.2	59.0	6 992.7
11.2	3 231.2	23.2	5 421.4	35.2	6 671.0	47.2	7 079.3	59.2	6 988.2
11.4	3 275.4	23.4	5 452.4	35.4	6 683.3	47.4	7 081.2	59.4	6 983.7
11.6	3 319.2	23.6	5 483.2	35.6	6 695.3	47.6	7 083.0	59.6	6 979.0
11.8	3 362.7	23.8	5 514.0	35.8	6 707.1	47.8	7 084.7	59.8	6 974.3
12.0	3 405.7	24.0	5 544.6	36.0	6 718.7	48.0	7 086.2	60.0	6 969.5