Population characteristics of *Vachellia erioloba* woodland near Kathu, Northern Cape

E Cronje

orcid.org/0000-0003-4116-9374

Dissertation accepted in fulfilment of the requirements for the degree *Master of Science in Environmental Sciences* at the North-West University

Supervisor: Prof K Kellner

Co-supervisor: Ms CD Neethling

Graduation May 2023

22833099

ACKNOWLEDGEMENTS

First and foremost, I am thankful for God's goodness!

I would like to thank the following people for their contribution in completing this study:

- My supervisor Prof. Klaus Kellner for his valuable inputs, guidance and support.
- Ms Chrizette Neethling for her guidance, professional input and inspiration for the study.
- Dr Arnim Marquart for his assistance with statistical analysis.
- Ms Daleen Groenewald for the editing of this dissertation.
- Sishen Iron Ore mine for allowing me access to their property and disclosing their biomonitoring data for this study.
- EndemicVision Environmental Services for the opportunity.
- EndemicVision's team for fieldwork and data collecting.
- My family for their love and continuous support.

ABSTRACT

Vachellia erioloba occurs in arid and semi-arid areas. In the dry woodlands of the southern Kalahari, it is often the only large tree in the landscape, implying that it is a keystone species that plays an important role in maintaining ecological processes and species diversity.

Deforestation to make room for mining activities, solar power plants and residential areas poses a threat to *V. erioloba* woodlands, especially because developments of this nature often go hand in hand with increased groundwater extraction and dust pollution. Furthermore, overutilisation of this species for firewood, unsustainable pod removal and seed predation, as well as fire events have been cited as factors that could cause a decline in the health of *V. erioloba* woodland populations.

The Kathu Forest in the Northern Cape Province, predominately consisting of *V. erioloba* trees, was the first to be declared a protected woodland in South Africa in terms of Section 12(1)(c) of the National Forest Act (NFA) (Act 84 of 1998) in 2009. *Vachellia erioloba* has further been declared a protected tree under the NFA. Despite its protected status, a decline in the *V. erioloba* populations surrounding Kathu is still observed. Given that numerous entities own and, by inference, manage a woodland, this study set out to determine the population characteristics of the *V. erioloba* woodlands around Kathu with the intention to ultimately make recommendations for the sustainable management of these populations.

Seven *V. erioloba* woodland sites were investigated over a period of five years to determine the density and size class distribution of each site and its population dynamics (i.e. production, recruitment, establishment and mortality) as well as to observe ecological interactions (i.e. predation by bruchid and wood-boring beetles). In addition, other possible environmental stressors such as dust pollution, past fire events and distance from disturbances and rivers were noted for each population.

Significantly more die-back and mortality were observed at woodlands with fire impact and other disturbances (i.e. close proximity to developments and mining) compared to populations with hardly any disturbances. Furthermore, although present within all populations, the impact of wood-boring beetles on the mortality of these trees seems negligible. In terms of population dynamics, the variation in results indicates that each population should be managed according to its own impacts. For example, public access to woodlands could have a negative impact on pod removal, while high concentrations of game and/or livestock could have a negative impact on sapling establishment.

By understanding the population dynamics of individual *V. erioloba* woodlands, managing agencies would be in a better position to implement effective management strategies to secure their sustainability. It is recommenced that such strategies should include an effective fire management plan and controlling the stocking rate on a property. Furthermore, monitoring of these sites ought to be extended over a longer period of time, since five years can at best only provide short-term information on a long living tree species.

Key terms

Production, Recruitment, Establishment, Die-back, Management impact, Population dynamics

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER 1 INTRODUCTION

1.1 Background and problem statement

Vachellia erioloba (camelthorn tree), previously known as *Acacia erioloba* and *Acacia giraffae*, has been declared a protected tree under the National Forests Act (NFA), (NFA Act No. 84 of 1998). The purpose of the NFA's list of protected trees is to ensure the sustainability of certain tree species by enforcing certain protection measures (NFA Act No. 84 of 1998). According to the Act it is illegal to "cut, disturb, damage or destroy any protected tree or possess, collect, remove, transport, export, purchase, sell, donate or in any other manner acquire or dispose of any protected tree or any product derived from a protected tree" (NFA Act No. 84 of 1998). A licence may be obtained from the Department of Forestry, Fisheries and the Environment (DFFE) to execute some or all of the illegal activities, providing that the conditions to minimise the impact on the sustainability of the species, as stipulated in the licence, be met (NFA Act No. 84 of 1998).

Tree species are declared protected under the NFA, if listed as a protected natural resource in the Red List of Threatened Species as established by the International Union for Conservation of Nature in 1964 (IUCN, 2021). Species can, however, also be included in the NFA's list of protected tree species based on their value as a keystone species, their cultural or spiritual importance, the need to ensure sustainable usage of the species or whether or not the tree is protected through any other legislation (DWAF, 2007). *V. erioloba* is classified as least concern in the Red List of Threatened Species, but are, however, listed as declining (IUCN, 2021). The protective status is rewarded to *V. erioloba* due to its role as a keystone species in the southern Kalahari, its cultural value and the unsustainable use and destruction of this species (Seymour & Milton, 2003; DWAF, 2007; Esterhuyse, 2017)

Vachellia erioloba is regarded as a keystone species in the southern Kalahari due to the critical role these large trees fulfil in maintaining the ecological processes in this landscape (Seymour, 2006; Steenkamp *et al*., 2007). According to Davic (2003), a keystone species can be defined as a "strongly interacting species whose top-down effect on species diversity and competition is largely relative to its biomass dominance within a functional group. In terms of ecological systems, *V. erioloba* plays an important role in maintaining and regulating stable environmental processes and, due to the higher levels of available nutrients found in the soil beneath these trees, provide a microhabitat for several fruited plant species, such as *Boscia* spp. and *Lycium* spp. (Dean *et al.,* 1998). Furthermore, *V. erioloba* provides shade, shelter and nesting sites for various invertebrates, lizards, small mammals, and birds and improves the soil quality, produces edible pods, flowers and gum for animals (Seymour & Milton, 2003; Steenkamp *et al*., 2007).

The Kathu Forest in the Northern Cape Province, predominately consisting of *V. erioloba* trees, was the first to be declared a protected woodland in South Africa in terms of Section 12(1)(c) of the NFA (Mans, 2011). Woodlands were only identified as an important forestry resource that must be managed at the time of launching the National Forestry Action Programme in 1997 with the intention to make the monitoring and reporting of identified woodlands compulsory in terms of the NFA (CSIR, 2002). Note that in terms of this act, a woodland is defined as "a group of trees which are not a natural forest, but whose crowns cover more than five percent of the area bounded by trees forming the perimeter of the group" (NFA Act No. 84 of 1998).

A baseline study conducted in 2002 by the Council for Scientific and Industrial research on woodlands in South Africa found that approximately one-third of South African land is comprised of woodland, of which 11% is protected to some degree (CSIR, 2002). The intention with this study was to develop a classification system on behalf of the then Department of Water Affairs and Forestry (DWAF, now DFFE)[1]. In this regard, the CSIR found that according to Shackleton *et al.*, South African woodlands can be classified as either arid/eutrophic or moist/dystrophic, with eutrophic woodlands occurring in areas with low rainfall that are typically dominated by the genera *Acacia* (now classified as *Vachellia* and *Senegalia*) or *Commiphora* (CSIR, 2002). The *V. erioloba* eutrophic woodland near Kathu is located within the high-altitude *Acacia* woodland class. Only 6% of this woodland class is protected within South Africa (CSIR, 2002).

A woodland is owned and, consequently, managed by numerous entities within South Africa, making it difficult to protect the sustainability of this resource. The *V. erioloba* woodland is being threatened by various new developments, including the expansion of mining and solar power

---

[1] Since South Africa's first democratically elected government came into being in 1994, the mandates of the state departments tasked with the conservation and management of the country's natural resources have changed on several occasions. Consequently, the names of these departments have also changed more than once to reflect the regrouping and/or expansion of their respective functional areas.

In this dissertation, the current names of those departments responsible for the respective functional areas will be reflected in the text. However, when referring to a source published under the auspices of a previous department, the former name of that department will be used.

In this regard, note that two state departments were initially tasked with the conservation and management of South Africa's natural resources, namely the Department of Environmental Affairs and Tourism (DEAT) and the Department of Water Affairs and Forestry (DWAF).

In 2009, the Department of Water Affairs and Forestry divided and the Department of Agriculture, Forestry and Fisheries (DAFF) were founded. In the same year the DEAT divided and the Department of Environmental Affairs (DEA) were formed.

In the interim, matters relating to forestry, fishery and the environment in general were consolidated under one state department, resulting in the DEA changing its name to the Department of Environment, Forestry and Fisheries (DEFF). More recently, the department opted to change its name to the Department of Forestry, Fisheries and the Environment (DFFE). This name and acronym will be used in the text where applicable, unless reference is made to a source that should be credited to either the former DWAF, DAFF, DEA or DEFF.

Matters relating to the conservation and management of water as a natural resource currently resorts under the Department of Water and Sanitation (DWS). In this regard, too, the name of the department and its current/former acronym will be treated as explained above.

facilities in the Northern Cape Province of South Africa (Mans, 2011; Van Rooyen *et al.,* 2016). In addition to the threats posed by deforestation as a result of the afforestation, these trees are also affected by overutilisation for firewood (Van Rooyen *et al*., 2016). *V. erioloba* wood is favoured for firewood in the area and was even used as fuel for the Kimberley diamond mines before coal was produced (Sim, 1921 as cited by Barnes *et al*., 1997). Given the slow growing nature of this species, many trees have thus been lost due to unsustainable harvesting rates (Anderson & Anderson, 2001) and it is, therefore, not surprising that a decline in the camelthorn population near Kathu has been observed over the past years, resulting in a loss of biodiversity and species richness (Powell, 2005).

Studies conducted by Anderson and Laan in 1992 and 1998 respectively postulated that mortality among *V. erioloba* around Kathu might be ascribed to the continued extraction of groundwater resources by various developments in the areas and that this practice might have a negative effect on the environment in future (Van der Merwe, 2001). A comprehensive eco-physiological study was conducted in 2001 by Van der Merwe to determine the possible cause of the decline in the population of these trees. This study revealed that the lowering of the water table was not the sole reason for the mortality of *V. erioloba*, but that factors such as an increase in pod removal by animals (livestock and game) leading to a reduction in the seed bank and the use of arboricides to control the densification of woody species such as *Senegalia mellifera* (black thorn) possibly also cause a decrease in the *V. erioloba* population. Densification of/encroachment by woody species is regarded as a major problem in the area, and extensive bush control has been applied over the years to increase fodder for livestock production and other agricultural activities. In the same study, Van der Merwe (2001) also postulated that seed predation by beetles belonging to the family Bruchidae might also inhibit the re-generation of *V. erioloba*, thereby causing a decline in populations.

Zens and Peart (2003) pointed out that to determine the dynamics of a population, it is important to also understand the process of mortality of a specific species. However, as Menges (2000) found, this can be quite difficult due to the timeframe needed to study long-living species, such as trees, especially when it comes to collecting data on causes of mortality that develop gradually over time. In 1991, Clark as cited by Van der Merwe, 2001, postulated that the structure of the population in terms of size class and density could be used as a measure to determine the persistence of a specific species in unfavourable conditions. Li *et al.* (2014) concurred with this finding, adding that knowledge of at least two structural characteristics of a population could help to understand the population characteristics of a woodland, and that such knowledge could lead to enhanced management of individual species. With specific reference to *V. erioloba* populations, a study conducted by Barnes *et al*. in 1997 found canopy die-back to be a good indicator of the severity of stress in the physiological condition of adult *V. erioloba* trees and postulated that this can be used to monitor a decrease in populations of these trees.

Clearly, if the die-back and recruitment rate of *V. erioloba* trees are understood better, development agencies will be in a more favourable position to implement effective management strategies over the long term. The proviso, though, is that changes in terms of the utilisation of the population by game and livestock should also be taken into account, especially in areas where low recruitment occurs, since this has a direct impact on increasing the seed bank and allowing saplings to survive.

1.2 Aims, objectives and hypotheses

The aim of this study was to determine the population characteristics of the *V. erioloba* woodland population around Kathu in the Northern Cape Province of South Africa.

The objectives included:

- To determine the structure of the population by monitoring the die-back of the adult trees, and the recruitment dynamics of saplings; and
- to make recommendations to mining and development agencies regarding the temporal changes in *V. erioloba* populations.

The hypotheses for this study were:

- Die-back, as an indicator of environmental stress, will differ between populations subject to different impacts, such as fire, dust pollution and de-watering.
- Viable pods are assumed to have viable seed thus, populations with more viable pods present will have higher recruitment rates.
- The presence of Bruchidae beetle predation will affect the recruitment rate of a population negatively.

1.3 Structure of the thesis

Chapter 1: Introduction

Chapter 1 serves as a general introduction to this study, focussing on the problem statement and previous research conducted on the topic. This chapter also sets out the aims and objectives of the study as well as the hypotheses.

Chapter 2: Literature Review

A comprehensive overview and discussion of existing literature relevant to the research topic are provided in this chapter.

Chapter 3: Study Area

Details regarding the location of and an in-depth discussion of the study area (i.e. vegetation, climate, soils and hydrology) are presented in this chapter.

Chapter 4: Materials and methods

All methods applied and equipment used during this study are discussed in this chapter.

5. Results & Discussion

In Chapter 5, data derived during the study is presented and analysed, culminating in a detailed discussion of the results.

6. Conclusion & Recommendations

In addition to summarising this study, this chapter presents a succinct synopsis of the conclusions that have been derived in the process. Limitations regarding this study and its methodology, assessments and results are further discussed, while some recommendations are made with regards to future research on the subject.

Bibliography

All literature consulted in the course of this study are listed here for referencing purposes.

CHAPTER 2 LITERATURE REVIEW

2.1 Characteristics of *Vachellia erioloba*

2.1.1 General description

Vachellia erioloba, previously known as *Acacia erioloba* or *Acacia giraffae* is commonly called the camelthorn tree. *Acacia* is derived from the Greek word *akis* with the meaning ‘sharp point’, referring to the spines present on all African *Acacia* species, whereas *erioloba* is derived from *erio*, meaning ‘hairy’ and *loba* meaning ‘pod’, thus referring to the very distinctive pods carried by the camelthorn tree (Thomas *et al.,* 2009). The genus, *Vachellia*, was named after a chaplain and plant gatherer from China, George H Vachell (1789-1839) (Becking, 2020). *V. erioloba* is part of the Mimosoideae subfamily which forms part of the Fabaceae family. The Fabaceae family (legume family) consists of 12 000 predominantly pod-bearing species (Esterhuyse, 2017).

The camelthorn is a very slow-growing, semi-deciduous tree that can reach up to 16 m in height (Palgrave, 2002) or grows as a shrub, depending on water availability and soil depth (Barnes *et al*., 1997). The tree is long-lived and can reach ages of up to 300 years (Timberlake, 1980 as cited by Barnes *et al.,* 1997). It displays a wide spreading crown and is commonly the tallest tree in its range (Thomas *et al.,* 2009). New shoots are smooth and reddish brown, growing in a zigzag manner, where older branches and stems are covered with a dark brown, deeply furrowed bark (Venter & Venter, 2012). The camelthorn has a remarkable taproot which can reach a depth of up to 68 m (Jennings, 1974).

The white, paired, straight spines (often swollen at the base) can reach a length of up to 6 cm and are a well-known feature of the camelthorn (Venter & Venter, 2012). The leaves, growing in groups of one to seven, are twice-pinnately compound and bluish-green in colour with prominent veins (Thomas *et al.,* 2009). Bright yellow, round inflorescence up to 15 mm in diameter, growing in groups of one to four, can be seen from July to September (Palgrave, 2002; Moustakas *et al.,* 2006) (Figure 2.1).

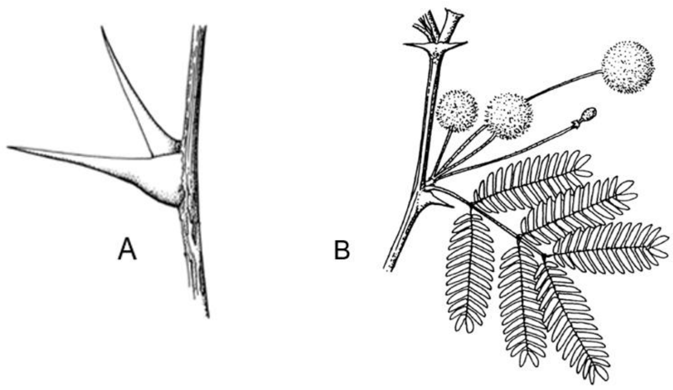

Figure 2-1: (A) Spines with swollen basis ("domatia") of *V. erioloba*; (B) Leaves and inflorescence of *V. erioloba* (adapted from Barnes *et al.,* 1997)

From December to March, the camelthorn bears fruit. These indehiscent pods are kidney shaped, hard and covered in velvety grey hairs (Palgrave, 2002). Each pod contains multiple (8-14) elliptic seeds which are dark brown when ripe (Barnes *et al.,* 1997) (Figure 2.2). One of the most common variations between individual trees is the shape (structure), size and production of the pods. Pods can be as small as 50 x 20 mm or as large as 130 x 70 mm, and their shape can vary from cylindrical to flat and crescent shaped (Barnes *et al*., 1997).

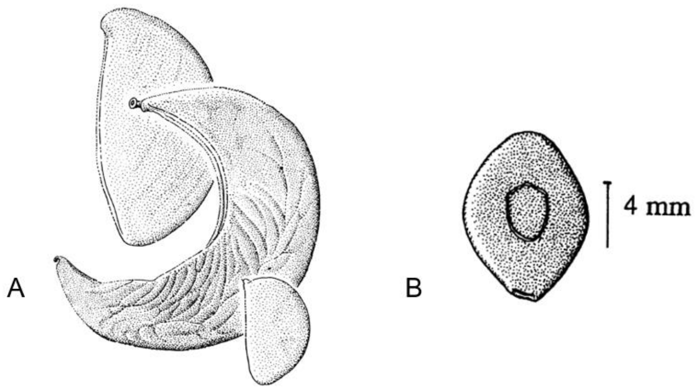

Figure 2-2: (A) Pod variation of *V. erioloba*; (B) Seed of *V. erioloba* (adapted from Barnes *et al.*, 1997)

2.1.2 Distribution

The tree occurs in dry woodland arid and semi-arid areas, and is distributed from Angola and Zambia to the north-western parts of South Africa (Palgrave, 2002; Venter & Venter, 2012). Distribution is largely dependent on the presence of deep sandy soils and not so much on precipitation. Consequently, distribution varies from areas with very low rainfall (< 40 mm/a) typical of the Namib Desert to areas with high rainfall (900 mm/a) such as those found in Zambia

(Barnes *et al.,* 1997). In areas with less than 250 mm/a rainfall, the presence of camelthorns could indicate the presence of underground water (Timberlake, 1980 as cited by Barnes *et al.,* 1997). Notably, dense stands of camelthorn trees in arid areas such as the Northern Cape Province are possible due to the availability of sufficient underground water present in the aquifers below the calcrete substrate and red Kalahari sand (Liversidge, 2001). The camelthorn's distribution ranges over mean daily temperatures from <15°C up to >45°C, as well as areas where frost is a common occurrence (Barnes *et al.,* 1997). In more arid regions, the camelthorn is often the only large tree in the landscape (Barnes *et al*., 1997).

2.2 Importance of *Vachellia erioloba*

2.2.1 *Vachellia erioloba* as a keystone species

A keystone species can be described as a species with a top-down effect, meaning that one species controls the population dynamics (i.e. species diversity, patterns and processes) as well as the sustainability of a community (Paine 1969 as cited by Seymour, 2006). As one of the few large trees present in the southern Kalahari, *V. erioloba* plays an important role to maintain the ecological processes in the landscape and is thus regarded as a keystone species (Seymour, 2006; Steenkamp *et al.,* 2007; Prayag *et al.,* 2020). Large trees play a very different role in arid environments than small trees, as they provide more shade and shelter for larger biodiversity (Dean *et al*., 1998). Seymour (2006) found that species richness per one square metre beneath a large tree's canopy is significantly higher than species richness beneath a small tree's canopy or the diversity within the surrounding grass layer.

Shade, being one of the most important contributions of large trees (Naudé *et al*., 1987), supports specific plant assemblages including shade-tolerant species. Within an otherwise arid and semi-arid area, the large tree canopy maintains soil moisture due to lowered evaporation rates and stem flow (Belsky & Canham, 1994; Dean *et al.,* 1998; Seymour, 2006; Anderson, 2009). Larger trees are also capable of hydraulic lift, whereby water is transported from deep roots to lateral roots to increase the moisture in the soil surface (Richards & Caldwell, 1987).

Higher levels of available nutrients are found in the soil beneath camelthorn trees, providing a microhabitat for several fruited plant species (Dean *et al.,* 1998; Murovhi, 2003). These heightened nutrient levels are most likely due to the increased animal activity beneath the tree, where the soil is enriched by bird droppings, dung from animals that rest under the tree, fallen nests and the remains of prey left by raptors, owls or vultures (Dean *et al*., 1998). Ultimately the seed bank beneath these large tree canopies is more diverse than the seed bank in the surrounding open grassland and could aid in the recovery of vegetation impacted by high browsing pressure (Tessema *et al.,* 2017).

A large tree also serves as an important vertical habitat structure (Seymour, 2006). The shade, shelter and nesting sites provided by these large trees attract various animals (Belsky & Canham, 1994), including antelope, canines, birds, predators and lizards. Predators, such as leopards, use large trees as resting sites as well as a place where they can safeguard their prey against scavengers (Dean *et al.,* 1998). Owls, vultures, raptures and other birds of prey also prefer these large trees for perching and nesting sites (Dean *et al.,* 1998; Anderson & Anderson, 2001; Anderson, 2009). Older trees with natural holes and deep fissured bark provide nesting sites for certain birds, mammals and lizards, such as the striped skink *(Mabuya striata*) (Dean *et al.,* 1998). Large camelthorn trees are the predominant host tree of the sociable weaver (*Philetairus socius*), supporting their large communal nests which could measure up to 7.2 m in diameter (Maclean, 1967). In turn, these nests provide shelter for various other bird species, including the pygmy falcon (*Polihierax semitorquatus*) and the giant eagle owl (*Bubo lacteus*), as well as provide food for predators such as snakes and honey badgers (*Mellivora capensis*).

Besides providing shade and shelter for various animals, the camelthorn is also an important food source for both humans and animals. The pods are highly palatable and nutritious with a protein content of almost 14% (Mannheimer & Curtis, 2009). The high protein and fat content make these pods the perfect fodder, especially during periods of drought. The ripe pods are dropped from the tree during the winter and provide an invaluable food source for antelope (such as gemsbok), various rodents, such as the karoo rat (*Parotomys brantsii*), other game (such as giraffes or elephants) and large and small livestock (Dean *et al.,* 1998; Anderson, 2009; Mannheimer & Curtis, 2009). Palgrave (2002) as well as Venter and Venter (2012) reported that cattle's milk production increased where they have been fed with the pods of *V. erioloba*.

The leaves of the camelthorn tree are also high in protein (17%) (Murovhi, 2003) and serve as a food source for various browsers. Young leaves and unripe pods however contain prussic acid which can cause poisoning in livestock (Murovhi, 2003; Thomas *et al.,* 2009). Young shoots are eaten by donkeys and goats and flowers are utilised by game, livestock and the larvae of the topaz blue butterfly (*Azanus jesous*) (Mannheimer & Curtis, 2009; Venter & Venter, 2012). Coffee can be made from the seeds (Mannheimer & Curtis, 2009; Venter & Venter, 2012), and the gum can be eaten by humans, animals and birds (Palgrave, 2002; Venter & Venter, 2012).

The camelthorn tree has many traditional medicinal uses, such as the treatment of cuts and wounds (Mannheimer & Curtis, 2009). The gum can be used for stomach ails, the bark is used to treat headaches, powdered pods for ear infections and a tea can be made from the leaves to treat lung infections, while the Mbukushu native community use the roots for cough medicine (Palgrave, 2002; Mannheimer & Curtis, 2009; Thomas *et al.,* 2009; Venter & Venter, 2012).

The wood of the camelthorn is very hard and durable and has been utilised by humans for centuries (Palgrave, 2002). Diamond mining used the wood as raw material, for the likes of mine props, poles, bearings and wagons (Palgrave, 2002; Pakenham, 2007; Esterhuyse, 2017). The wood is also used to build houses, as fence posts, for utensils' handles, knobkieries and pestles and as firewood (Pakenham, 2007; Mannheimer & Curtis, 2009).

Some ant species nest in specialist hollow structures formed from plant matter, like spines or leaves, called domatia, which are specifically engineered to house ants (Wheeler, 1942 as cited by Cambell *et al.,* 2013) (Figure 2.1). When these domatia are present in a specific plant, the plant will benefit from the ants in various ways, e.g., protection against herbivory (Trager *et al.,* 2010). The older, hard and swollen spines of the camelthorn tree serve as domatia for ant colonies (Cambell *et al.,* 2015). Other invertebrates also utilize these swollen thorns, including spiders, moth larvae and wood-boring beetles (Barnes *et al.,* 1997; Cambell *et al.,* 2013; Palgrave, 2002).

Another invertebrate commonly found on the camelthorn tree is the southern African wild silk moth *Gonometa postica* (Raath *et al.,* 2017). *Gonometa postica* is endemic to the Kalahari and Namibia regions of Southern Africa and is mostly found on *V. erioloba* and *Senegalia mellifera* (Van der Merwe *et al.,* 2017). The cocoons of this moth are economically important since they are collected and after processing, the fibre is sold as silk (Veldtman *et al.,* 2002). The silk derived from these cocoons is fine, of a high quality with a natural gold colour and easy to dye (Hartland-Rowe, 1992).

Furthermore, the camelthorn tree has great aesthetic value, featuring in many travellers' photographs (Anderson & Anderson 2001), and is an icon of the Kalahari region, giving a sense of place (Seymour & Milton, 2003; Esterhuyse, 2017).

2.2.2 Conservation status of *Vachellia erioloba*

As mentioned, *V. erioloba* has been declared a protected tree under the National Forests Act (NFA), (NFA Act No. 84 of 1998), rendering any utilisation thereof without a permit to be illegal whether the tree is dead or alive. The act stipulates that "no person may (a) cut, disturb, damage, destroy or remove any protected trees; or (b) collect, remove, transport, export, purchase, sell, donate or in any other manner acquire or dispose of any protected tree, except under a licence granted by the Minister." (NFA Act No. 84 of 1998). The over-utilisation of this tree (especially for firewood), its ecological importance as a keystone species and its cultural value have led to its inclusion in the list of protected species (Seymour & Milton, 2003; DWAF, 2007; Esterhuyse, 2017). As much as this protection does not prevent the utilisation or destruction of this species, it sets certain rules to make sure the species is utilised sustainably.

Permits are sometimes granted for destruction of the trees, provided the permit holder compiles with certain compensation methods. When there is a net loss in biodiversity due to this destruction, an offset agreement is entered into between the Department of Forestry, Fisheries and the Environment (DFFE) and the permit holder to compensate for this loss (DAFF, s.a.). One such example of a biodiversity offset agreement, is research conducted by Van Rooyen *et al.* in 2016, for the Adams solar energy facility project near Hotazel.

The majority of *V. erioloba* trees found in and around Kathu form part of a woodland and not a forest. A woodland consists of a grass-dominant herbaceous layer and separate individual trees at varying distances whereas a forest is mostly contiguous with no grass layer (DAFF, s.a.). Of the ± 40 million ha of woodland in South Africa, only 5 million ha is protected (DAFF, 2015b). Section 3(b) of the NFA (Act 84 of 1998) requires the conservation of a minimum area of each type of woodland. The implementation of this section is, however, being delayed due to budget constraints (DAFF, 2015b).

The *V. erioloba* woodland populations monitored for this study are situated within the high-altitude *Acacia* woodland class (as classified by the CSIR in 2002). Only 12% of the high-altitude *Acacia* woodland is protected within South Africa (DAFF, 2015b). The “Kathu Forest” (predominantly consisting of camelthorn trees) forms part of this 12% and has been declared a protected woodland in South Africa in terms of Section 12(1)(c) of the NFA (Mans, 2011). The Kathu Forest lies to the north of the town Kathu in the Northern Cape Province and is almost 4 000 hectares in extent (Anderson & Anderson, 2007). It was declared protected for scientific and aesthetic reasons, as well as for its contribution to biodiversity and the eco-tourism opportunities it offers (Anderson & Anderson, 2007). Various species included in the IUCN’s red list of threatened species (2021) can be found in the Kathu Forest, including *Hoodia gordonii* and *Harpagophytum procumbens*, as well as other protected species (e.g. *Nerine laticoma*) and endemic or near endemic species (e.g. *Panicum kalaharense*) (Anderson, 2009). Furthermore, the Kathu Forest is rich in archaeological history given that the teeth of an extinct elephant species (*Elephas recki*) as well as silica-coated hand axes have been found within the boundaries of the forest (Anderson, 2009).

This woodland was originally declared a State Forest in 1920 but was deproclaimed in 1956 for the development of the town Kathu (Anderson & Anderson, 2007). In 1995, the Kathu Forest was proclaimed a Natural Heritage Site and a commitment was made by the municipality to only allow new developments to the south of the town (Anderson & Anderson, 2007). To this end, the Strategic Environmental Assessment and Integrated Environmental Development Plan for the Kgalagadi District Municipality recommended that no development should take place within the Kathu Forest and that it should be regarded as a conservation area (Anderson & Anderson, 2007). The Spatial Development Framework (SDF) for the Gamagara Local Municipality also recommended that the area should only be used for tourism purposes (Anderson & Anderson, 2007). Despite all of these being in place, developments within the Kathu Forest still took place, resulting in 27%-34% of the forest already being transformed (Anderson & Anderson, 2007). Considering that the threshold at which a forest ecosystem becomes critically endangered stands at 70% of the natural area remaining, the Kathu Forest may already be considered critically endangered (National Spatial Biodiversity Assessment of South Africa as cited by Anderson & Anderson, 2007). In 2009, the Kathu Forest was finally declared a protected woodland in South Africa in terms of Section 12(1)(c) of the NFA (Mans, 2011).

2.3 Disturbances threatening *Vachellia eriolob*a woodland populations

To determine the dynamics of a population, it is important to also understand the process of mortality of a specific species (Zens & Peart, 2003). Known disturbances may alter the size and structure of a woodland and will, ultimately, alter the biodiversity patterns and processes within a population (Seymour & Milton, 2003). The increasing mortality of *V. erioloba* woodlands within the Kalahari is of growing concern (Schachtschneider & February, 2013). The decline of camelthorn populations has been observed since the 1980's (Laan, 1998 as cited by Van der Merwe, 2001), and in 2011, Mans specifically reported on the death of multiple camelthorn trees, of all ages, in and around Kathu.

2.3.1 Deforestation

Deforestation can be defined as "the cutting down of trees in a large area, or the destruction of forests by people" (Cambridge University Press & Assessment, 2022). Deforestation can have a devastating effect, since forest (contiguous tree layer with no grass layer) or woodland (grass-dominant herbaceous layer and separate individual trees) ecosystems can already be classified as critically endangered if only 70% of the natural ecosystem remains (National Spatial Biodiversity Assessment of South Africa cited by Anderson, 2009). A woodland is owned and subsequently managed by numerous entities within South Africa, making it difficult to protect the sustainability of this resource.

Land being cleared for the expansion of mining, solar power facilities and housing are some of the greatest threats to the *V. erioloba* woodland in the Northern Cape Province of South Africa (Anderson & Anderson, 2007; Mans, 2011; Van Rooyen *et al*., 2016). In the Kathu Bushveld area (Mucina & Rutherford, 2006) researched by Van Rooyen *et al.* in 2016, developed land increased from 8 418 ha in 2002 to 36 561 ha in 2014 (an increase of more than four times). The *V. erioloba* woodland is also being cleared to improve grasslands for fodder production, especially in areas where the misconception exists that this species can cause bush encroachment (densification of woody species) (Seymour & Milton, 2003).

2.3.2 Over-utilisation

With its low moisture content and high heat capacity, the wood of the camelthorn tree makes excellent firewood, which led to unsustainable harvesting of the species over the years (Anderson & Anderson, 2001; Raliselo, 2003; Van Rooyen *et al*., 2016). Since the selling of firewood can create a much-needed income for farmers, especially in periods of drought, Anderson and Anderson (2001) reported that up to 60 tons of firewood were collected on some properties near Kathu per month. The over-utilisation of camelthorn wood is, however, not limited to privately owned land, since Liversidge (2001) reported that dead camelthorn wood was even collected within nature reserves to be sold as firewood.

In the past, the wood was also used as fuel for the Kimberley diamond mines before coal was produced which led to the removal of many large trees between Olifantshoek and Kathu and the area surrounding the city of Kimberley in the Northern Cape Province of South Africa (Sim, 1921 as cited by Barnes *et al.,* 1997). More than 160 years ago, even the missionary Robert Moffat wrote about “the remains of ancient forests of the camelthorn” which he ascribed to over-utilisation by the “Bantu” (non-Europeans) (Anderson & Anderson, 2001).

As pointed out by Morkel (2013), over-utilisation of these trees brings short-lived economic relief and needs to be managed better if sustainable utilisation is to be achieved. Even though Morkel (2013) found that cut stumps could coppice, Seymour & Milton (2003) pointed out that these stumps are very sensitive to browsing, leading to the failure of regeneration.

Furthermore, over-utilisation of the pods of *V. erioloba* is common, especially close to towns where the pods are also collected on a large scale to sell as fodder (Anderson, 2009). According to Bosch (personal interview, as cited by Van der Merwe, 2001), 60 tons of pods are collected in and around Kathu in good pod-bearing years. The number of pods produced each year differs and can be as much as 200 kg per annum per adult tree (Schatschneider, 2010). Due to the use of the pods by humans and animals, seed produced is often not recycled into the seed bank causing a decline in sustainable *V. erioloba* populations (Eriksson & Ehrlen, 1992; Van der Merwe, 2001). This was evident in the study conducted by Van der Merwe (2001) where no seeds were found in soil samples taken in and around Kathu.

2.3.3 Dust pollution

Dust deposited on trees can cause damage to the leaf surfaces, inhibit proper pod development (Powell, 2005) or alter the soil chemistry which causes problems with the germination of seed (Farmer, 1993). Depending on the size of the dust particles, the dust may smother the leaves by blocking the stomata (Farmer, 1993). Various types of dust will have different effects on different types of vegetation and cannot be compared with one another (Farmer, 1993). It is however evident that chemical effects from different types of dust are more damaging to vegetation than physical effects caused by the dust particles (Farmer, 1993).

Dust originating from the open pit mines in the area is believed to be one of the causes of the mortality of the camelthorn populations near Kathu (Anderson, 2009). The red layer of mine dust can be seen on the vegetation alongside the roads (Van der Merwe, 2001). Camelthorns are specifically vulnerable to dust pollution since the leaves do not have any form of protection, preventing the dust to embed in the leaves as for example the waxy layer present in the leaves of the *Ziziphus mucronata* woody species (Powell, 2005). The leaves of camelthorns are also rough, which aid in the accumulation of the dust on the laminae (Van der Merwe, 2001). The study conducted by Van der Merwe (2001) found that although the dust damaged the leaves it did not have a negative effect on the chlorophyll content and ultimately the photosynthetic ability of the camelthorns. The study also found no significant negative effect on the transpiration rate or protein contents of the leaves. It was also determined that the dust did not inhibit the germination or the growth of seedlings of camelthorn trees.

2.3.4 De-watering

Various entities, such as the Kathu town, Golf course, townships and mines, are utilising groundwater within the study area, which includes abstraction from the Kathu aquifer and the Khai Appel aquifer (Golder Associates, 2014). In 2015, Shangoni reported that approximately 14,523,463 m^3 of groundwater was being abstracted annually by Sishen Mine, of which approximately 7,711,8486 m^3 was used for mining operations and mining facilities (Shangoni, 2017). The surplus groundwater was either being discharged into the Lategan Dam or used to supply water to the Gamagara Local Municipality, to Dingleton and to the Vaal Gamagara pipeline (Shangoni, 2017). In 2013, Geo Pollution Technologies estimated that Khumani Iron Ore Mine, which is located within the dewatering zone of Sishen Mine, will be intersecting the groundwater table by 2028 or later, thus increasing the abstraction rate by this date (Geo Pollution Technologies, 2013). In the past 10 years (2012-2022), the Kathu town expanded by 61%, the industrial area expanded by 28% and the Mapoteng township expanded by 185% (Google Earth Pro, 2012; Google Earth Pro, 2022). This is a total expansion of approximately 474 hectares, with an additional 400 hectares of residential expansion currently in development to the north-west of the Kathu town (Google Earth Pro, 2022).

With the growing demand for groundwater by various mines in the area, and given that Kathu is an expanding town, concerns have been raised that the continued abstraction of groundwater resources from developments near Kathu would cause the mortality of *V. erioloba* trees (Van der Merwe, 2001; Anderson & Anderson, 2007), as these trees develop very deep tap roots to utilise underground water sources in arid areas (Moustakas *et al.,* 2006; Anderson 2009). In 2001, Van der Merwe, reported that de-watering was not the primary cause for the mortality of *V.erioloba* woodlands surrounding Kathu. This study has however only taken data into account from 1994 to 1998, after which tremendous mining expansions and increased de-watering took place in the area (Van der Merwe, 2001). Similar findings were reported by Van der Merwe *et al*. in 2020 of a study conducted where long-term observation (1982-2016) of tree die-back in the Kalahari Gemsbok National Park led to the conclusion that increased die-back cannot be ascribed to groundwater extraction alone. This study was however situated within a National Park where de-watering is limited to watering points, it cannot be compared to the scale of dewatering taking place in this study area (Van der Merwe *et al*., 2020).

Schachtschneider and February (2013) as well as Meintjies and Van der Westhuizen (2017) estimated that should the water table be lowered permanently by more than 4 m, canopy die-back in *V. erioloba* would increase. In this regard, Meintjies and Van der Westhuizen (2017) also cautioned that since the effects of de-watering will only become evident once the ecosystem has been stressed beyond its critical threshold, lowering of the water table should not be ruled out as a possible cause of *V. erioloba* mortality.

The study conducted by Woodborne in 2004 found that surface recharge events (e.g. excessive rainfall) could trigger the use of shallow water in an attempt to save energy by halting the camelthorn tree’s investment in deeper root development. However, should the water table lower during the period where sufficient surface water is available (which could be up to 10 years), the roots will not be able to reach the much deeper water sources since root growth has been restrained to a certain depth. The tree will go into a state of dormancy and, eventually, die as a result of water stress unless the water table rises again. Furthermore, while the tree is in a state of dormancy, its vulnerability is heightened since it can no longer grow new plant tissue to recover from insect or bacterial impacts and may, consequently, die from causes other than water stress.

In a report submitted to AGRI Northern Cape in 2018, Pietersen and Gaffoor cautioned that the cumulative impacts of de-watering and groundwater abstraction are not always taken into account by governing bodies when doing a study in the area and that the current rate of groundwater utilisation in the area is often unsustainable, since local municipalities, the agriculture sector and power stations are all tapping this natural resource. In addition, natural discharge processes such as evaporation, transpiration and seepage (water discharged to wetlands and rivers) also contribute to a decrease in the water table. Of interest here is that as pointed out by Schachtschneider (2010) and as confirmed by the findings of Pietersen and Gaffoor (2018), recharging and maintaining the water table in semi-arid areas are dependent on excessive rain events.

2.3.5 Arboricides

Bush encroachment is a wide-ranging phenomenon seen in savannas where overgrazing suppresses the grass layer, leading to higher densities of the woody shrub/tree layer (Skarpe, 1991). Other factors that could cause bush encroachment include the absence of fires, underutilisation of the grass layer and human disturbance (Van Rooyen *et al*., 2016). Incorrect placement of water points, placement of lick sites, the absence of browsing animals (especially game) and replacement with domestic grazers are just some examples of bad management that could lead to bush encroachment (Van Rooyen *et al.,* 2016).

Bush control is applied in bush-encroached areas for ecological, economical and/or aesthetic reasons and are widely observed in the areas surrounding Kathu (Van Rooyen *et al*., 2016). Once an area is encroached, it can take very long to reverse this phenomenon via better management practices such as the removal of grazers, thus other methods, such as mechanical or chemical control are often used (Jeltsch *et al.,* 1996; Van Rooyen *et al*., 2016; Harmse *et al*., 2016). Application of arboricides is regarded as one of the most cost-effective, chemical methods to clear bush-encroached areas of large extents (Seymour, 2006; Harmse *et al.,* 2016). When applying chemical control, it is important to apply an arboricide that is selective with regard to the target species (Harmse *et al.,* 2016; Van Rooyen *et al.,* 2016; Marquart *et al.*, 2023). Non-selective arboricides not only affects the target species but can have detrimental impacts on the entire woody vegetation layer (Dreber*, et al.*, 2019).

Extensive bush control has been applied over the years to increase fodder for livestock production and other agricultural activities. Arboricides used within the study area, for the control of *Senegalia* *mellifera* (black thorn) causing bush encroachment was not target-specific and led to the demise of many *V. erioloba, V. haematoxylon* and *Boscia albitrunca* trees in the area (Liversidge, 2001; Van der Merwe, 2001; Seymour, 2006; Van Rooyen *et al*., 2016).

2.3.6 Animal impact/herbivory

The study conducted by Van der Merwe in 2001 revealed that one of the main factors increasing the mortality of *V. erioloba* is increased pod removal by animals through herbivory (livestock and game) leading to a reduction of the seed bank. Van der Merwe (2001) further found that the seeds beneath the mean browse height do not reach maturity since they are utilised by the animals too early in the production stage. Herbivores can also hinder the growth of saplings, keeping them in a vulnerable state or completely destroying them as a result of trampling (Barnes *et al.,* 1999 as cited by Morkel, 2013). Saplings may, however, respond to this herbivory by investing in root growth, thereby increasing their chances of survival (Seymour, 2008).

Utilisation of pods for fodder can also have a positive impact on the sustainability of *V. erioloba* populations. Scarification of the seed coat by the digestive system of animals (livestock and game) and an increase in moisture availability, aid in the germination process (Coe & Coe, 1987; Hoffman *et al.,* 1989; Barnes *et al.,1997*). The dispersal of seeds away from mature tree canopies (by means of dung) can also have a positive effect on seedling establishment during dry periods where the competition for moisture is too high beneath tree canopies due to denser vegetation (Jeltsch *et al*., 1999). Removal of the seed from beneath the canopies of *V. erioloba* also lessens their vulnerability to parasitism (Dean *et al.,* 1998).

2.3.7 Fire impact

Due to the lack of sufficient vegetation for fuel loads, fire events are not a general occurrence in arid and semi-arid areas but can occur occasionally, especially after good rainfall events (Steenkamp *et al*., 2007). A general mortality rate of 10% is observed in woody vegetation due to fire, a rate that was also observed in the Kgalagadi National Park (Jeltsch *et al*., 1999). The consequences of a fire will, however, differ every time since each fire is different depending on factors such as weather conditions, available fuel load and the type of fire (Seymour & FitzPatrick, 2007).

Large, mature camelthorn trees and seedlings are most vulnerable to fire, where the rest of the trees usually resprout after a fire (Morkel, 2013; Seymour, 2006; Steenkamp *et al.,* 2007). Smaller trees tend to resprout at the base of the tree (coppicing), whereas larger trees resprout form the trunk or canopy (Seymour, 2006). Old trees often have a lot of accumulated dry material since they house various nests, such as those of sociable weaver or tree rat, and this contributes to the fuel load and, ultimately, their demise in the event of fire (Prayag *et al.,* 2020; Steenkamp *et al*., 2007). Bark damage is also more severe in larger, older trees than damage found in younger trees (Seymour, 2006).

By preventing recruitment of seedlings to become mature trees and eliminating large old trees by fire events can change the structure of an ecosystem and ultimately promote grassland vegetation (Seymour & Huyser, 2008).

2.3.8 Seed predation

African savanna trees produce insignificant seed banks, mainly due to parasitism (Tybirk *et al.,* 1994 as cited by Jeltsch *et al.,* 1999). Van der Merwe (2001) found that seed predation by Bruchidae beetles could cause a decline in *V. erioloba* populations and inhibit the re-generation of this woody species. *V. erioloba* is specifically vulnerable to bruchid predation since these beetles prefer indehiscent pods and large seeds (Barnes *et al*., 1997). The rate of bruchid infestation differs from tree to tree, possibly due to variances in flowering times, chemical differences and/or the presence of *Crematogaster* ant colonies (Barnes *et al.,* 1997; Seymour & Milton, 2003). A study conducted by Traveset in 1990 found that *Crematogaster* ants prey on the eggs of bruchid beetles in the pods of *Vachellia farnesiana*.

Bruchid beetles lay their eggs in the pods of *V. erioloba* trees, while unripe and the young larvae will then feed on the maturing seeds, thereby damaging or completely destroying the seed embryos (Halevy, 1974 as cited by Hoffman *et al.,* 1989). Mature pods will drop to the ground and lay there until they are predated by herbivores or decompose. The longer the pods have been lying on the ground the higher the level of predation, most likely due to the amount of time that has passed for larval development (Hoffman *et al.,* 1989). Bruchid beetles are killed while passing through the digestive system of herbivores, and some of the seed could be predation-free if the larvae have not reached them yet (Hoffman *et al*., 1989). Rapid germination is also expected in seeds with exit holes where the beetle has not damaged the embryo yet, thus leaving the seed coat more permeable for water uptake (Lamprey *et al*., 1974 as cited by Hoffman *et al*., 1989).

The pods of *V. erioloba* trees are also consumed by various animals such as birds, ungulates that feed on the pods from the lower canopy before they are ripe, and rodents foraging the dropped pods (Barnes, 2001). Barnes also found that in areas with large primate populations, foraging could be the main cause of seed predation since the primates are able to forage the entire canopy for ripe and unripe pods. Barnes (2001) also found that even though passing through the digestive system of the animal increases seed germination, seeds that were not fully developed when foraged were unable to germinate.

2.3.9 Insect damage

Most of the camelthorn trees that show signs of die-back, in and around Kathu, show signs of insect damage in the form of tunnels within the trunk or branches which may be due to the presence of Buprestidae (Mans, 2011) and Cerambycidae larvae (Ncedana *et al*., 2005).

The Buprestidae family consists of about 15 000 species of wood-boring beetles, commonly known as the jewel beetles (Evans *et al*., 2007). Most of the species in this family are oligophages which means they target specific genera (Evans *et al.,* 2007). Buprestids mainly feed on living trees that have been weakened by environmental stress, ultimately damaging the trees further or killing the tree entirely (Evans *et al.,* 2007). The female beetle is able to find trees that show weakness in the form of water stress, root damage (due to biotic or abiotic causes) or defoliation, and some species are even have the ability to detect specific parts of the tree that have been weakened (Evans *et al.,* 2007). Healthy trees are therefore able to defend themselves against buprestid attack.

The Cerambycidae family consists of about 25 000 species of wood-boring beetles, commonly known as the longhorn beetles (Evans *et al*., 2007). According to Evans *et al*., the majority of the Cerambycid beetles attack only dying or extremely damaged trees. Nevertheless, most species are very specific as to which part of a tree they will infest (e.g. the roots or the stem) and even though most of the species are regarded as secondary pests (i.e. kill a tree that is already damaged or dying) some species are regarded as primary pests that will be the main cause of tree's demise (Evans *et al*., 2007).

2.3.10 Invasive species

The high densities of the invasive *Prosopis glandulosa* woody plants has been suggested as another reason for the decline of *V. erioloba* trees (Schachtschneider, 2010). *Prosopis* has a dual root system with a very deep tap root and an extensive lateral root (Schachtschneider, 2010) and therefore has the ability to form dense thickets by doubling its area of distribution in just five years (Le Maitre, 1999 as cited by Schachtschneider, 2010).

The impact of this invasive *P. gladulosa* trees is far more significant within the riparian zone than in areas further inland where *V. erioloba* is largely dependent on groundwater, as *P. glandulosa* relies on both deep groundwater and shallow water (Schachtschneider, 2010). In riparian zones *V. erioloba* and *P. glandulosa* compete for the same water source resulting in mortality amongst

V. erioloba since *P. glandulosa* has a rapid transpiration rate, resulting in the lowering of the water table (Schachtschneider & February, 2013). Increased canopy die-back (more than 50%) among *V. erioloba* has also been observed in areas invaded (Schachtschneider, 2010).

2.4 Population characteristics

2.4.1 Production

V. erioloba can start producing large numbers of pods by the age of 20 years (Barnes *et al.,* 1997) and can produce as much as 200 kg of pods per year (Schachtschneider, 2010).

As pointed out by Joubert (2002) though, the number of pods produced depends on the canopy size of a tree and not on its age or height. Pods develop from December to March and will ripen and fall from the tree from April and throughout the winter (Barnes *et al.,* 1997; Palgrave, 2002). In some cases, rainfall can have an effect on the production of pods (Nel *et al.,* 1985; as cited by Barnes *et al*., 1997; Schachtschneider, 2010) as well as biotic factors (e.g. late frost or early winds) that could destroy the developing pods (Nel, 1983 as cited by Barnes *et al*., 1997).

2.4.2 Seed bank

The majority of seed produced and dispersed by *V. erioloba*, become dormant and are incorporated into the upper soil layer. Seeds can remain in a state of dormancy from a few days to a number of decades (Fenner, 1985). A well-established seed bank has many advantages for a population’s persistence. The seed of *Acacia* species (now called *Vachellia* or *Senegalia*) can remain viable within the soil where no rain event triggers germination for up to 50 years (Tybirk *et al.,* 1992) and only germinate when the environmental conditions are optimal, thus aiding in a population’s survival (Sabiiti & Wein, 1987).

Acacia seeds are well adapted to establish a viable seed bank, since these seeds have strong coats and are long-lived (Coe & Coe, 1987; Barnes *et al*., 1997). Furthermore, *Acacia* trees produce a significant number of seeds to compensate for the loss of seed due to predation, decay and utilisation (Sabiiti & Wein, 1987). Nevertheless, even though these trees should have large seed banks, the trees present in the African savanna seem to have insignificant seed banks (Tybirk *et al.,* 1994 as cited by Jeltsch *et al.,* 1999).

Due to the use of the pods by humans and animals, seed produced is often not re-cycled into the seed bank (Eriksson & Ehrlen, 1992; Van der Merwe, 2001) (see Figure 2.3 for the seed population dynamics). Within the study area, up to 60 tons of *V. erioloba* pods per year are collected by humans to sell as fodder (Bosch personal interview, as cited by Van der Merwe, 2001). Unripe seed being consumed by various animals such as birds, primates, livestock and other ungulates also contributes to a further decline in the establishment of a viable seed bank (Barnes, 2001). Van der Merwe (2001) found that unripe seeds consumed were destroyed when passing through the gut of animals, since these seeds did not have a hard coat yet and are, therefore, not taken up in the seed bank. Another limiting factor causing a decline in the number of viable seeds within the seed bank, is bruchid predation (Jeltsch *et al.,* 199). Van der Merwe also found that, for reasons unknown, some pods do not develop fully, and could thus also not contribute to the seed bank.

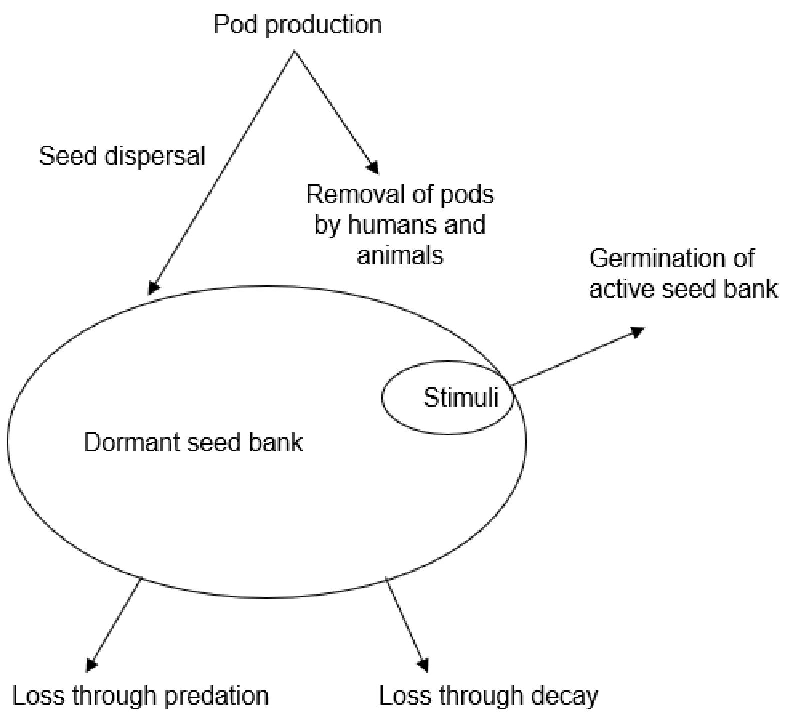

Figure 2-3: Flow chart of *V. erioloba* seed population dynamics (adapted from Harper, 1977)

2.4.3 Recruitment

It is common to see a large number of *V. erioloba* seedlings emerge following on a period of good rains (Barnes, 2001; Seymour, 2008; Van der Merwe *et al*., 2019; Van der Merwe *et al.,* 2020). Germination depends on the number of seeds available as well as soil moisture (Jeltsch *et al.,* 1999). In arid and semi-arid areas, rainfall can vary significantly from place to place within a relatively small area, resulting in various recruitment patterns being observed within one region (Steenkamp *et al.,* 2007).

Population patterns will also differ as a result of seed dispersal through animal dung, often leading to the formation of cohorts in areas of increased animal activity (Barnes *et al.,* 1997). Seed passing through the digestive system of animals are scarified, thereby increasing its permeability that results in quick germination (Hoffman *et al.,* 1989; Barnes, 2001). Seed not eaten by herbivores will eventually germinate when the seed coat is softened or broken down by the micro-organisms and acids within the soil and/or as a result of changing weather conditions (Cavanagh, 1980 as cited by Hoffman *et al*., 1989).

2.4.4 Establishment

Determining the causes that limit establishment of seedlings is important if the population dynamics of a woodland are to be understood, as it determines the sustainability of a species (Barnes, 2001). In a sustainable *V. erioloba* population, a large percentage of seedlings are expected in order to increase the chances of individuals surviving to adulthood (Van der Merwe, 2001). However, as pointed out by Seymour (2006), even with successful germination, the main problem found in *V. erioloba* populations is that these seedlings do not necessarily grow into larger size class saplings and adults.

In general, survival of *V. erioloba* seedlings is very low and numerous factors could influence their establishment (Barnes, 2001; Steenkamp *et al.,* 2007; Van der Merwe *et al.,* 2019; Van der Merwe *et al*., 2020). Sapling survival is susceptible to various environmental and vegetative factors such as competition between plant species, soil moisture (especially in arid and semi-arid regions with sandy soils) predatory invertebrates, rodents, herbivores and the availability of "safe sites" producing a suitable microhabitat for the seed to germinate and survive (Steenkamp *et al.,* 2007). All of these factors influence the sustainability of *V. erioloba* populations.

One reason for the lack of establishment of these seedlings could be drought conditions since moisture is a very important component during the first few seasons of the seedling's existence (Steenkamp *et al*., 2007; Van der Merwe *et al*., 2019; Van der Merwe *et al.,* 2020). In 2001, Barnes found that above-average rainfall is needed for the establishment of *Acacia* seedlings. This is supported by a study conducted by Seymour and Milton in 2003 and a later study conducted by Seymour in 2006 in which it was found that at least 50 rainfall days or at least three consecutive good rainfall years are necessary to reach a seedling establishment of 30%. It can thus be surmised that the amount of rain and rainfall patterns are major factors in the establishment of viable *V. erioloba* populations.

Furthermore, Smith and Shackleton (1988) found that shaded areas underneath tree canopies are not ideal for *Acacia* seedlings to establish and high mortality rates are observed (Smith &

Goodman, 1987 as cited by Barnes, 2001). Open areas where there are less vegetative competition and less fuel load in the case of a fire, are more suitable for the establishment of these seedlings (Van der Walt & Le Riche, 1984). Smith and Shackleton (1998) found that seedlings growing in shade did not invest as much energy in root development as seedlings growing in full sun and were ultimately unable to survive during the dry season. Barnes (2001) also pointed out that seedlings germinating in full sun will not be able to establish during very dry seasons and that, under these conditions, the shade provided by tree canopies will aid in less moisture being lost.

Grazing and trampling by livestock and game also hinder tree establishment (Jeltsch *et al*., 1999; Steenkamp *et al.,* 2007; Van der Merwe *et al.,* 2019). In the study conducted by Van der Merwe (2001), it was clear that very few saplings reached the middle size classes where browser pressure was high. Grazing can, however, be important in aiding with the removal of the grass layer, thereby reducing the competition for light (Seymour & FitzPatrick, 2007). Additional factors influencing establishment include fire impacting on small individuals, arboricides hindering growth and competition amongst vegetation (Seymour, 2006). The smaller individuals may decrease as the biomass of some individuals increases. This can be described as density-dependent self-thinning (Van der Merwe *et al.,* 2020).

2.4.5 Size class distribution

According to research conducted by Clark (1991, as cited by Van der Merwe, 2001), the structure of the population in terms of size class and density could be used as a measure to determine the persistence of the specific species in unfavourable conditions. A healthy, growing population will for instance show an inverse J-pattern size class distribution, where all size classes are represented with the majority of the population falling within the smaller size classes (Morkel, 2013; Van Rooyen *et al*., 2016). Van der Merwe *et al*. (2019) reported on studies conducted in the Kalahari Gemsbok National Park (now called the Kgalagadi National Park), over a course of 38 years where a visual assessment of the size class distribution of *V. erioloba* populations was used to determine the sustainability of these populations. In this study it was evident that one population was changing from a developing population to a mature population with more individuals present in the higher size classes and fewer in the lower size classes. It was also determined that one of the populations were declining but a new population was being, established not far from it.

Individual trees grow at different rates due to factors such as microsite conditions or the underlying genetics of each tree (Stahle *et al*., 1996 as cited by Van der Merwe *et al.,* 2019). Saplings can grow up to one meter in the first year when growing on deep sandy soils but will grow much slower on heavier soils (Barnes *et al.,* 1997). The growth rate will also depend on moisture availability, grazing pressure, fire frequencies and vegetative competition (Belsky, 1984; Scholes & Archer, 1997; Seymour, 2006; Midgley and Bond, 2001). Slow growth is expected in seedlings since most of the growth takes place in the root system in the early stages (Leistner 1967 as cited by Seymour & FitzPatrick, 2007). For example, a tap root measuring 110 cm has been found on a seedling only five cm in height (Seymour & Huyser, unpublished data, as cited by Seymour & FitzPatrick, 2007).

Cohorts, or trees of the same size and age, are a common phenomenon in *V. erioloba* populations. These cohorts form as a result of seed dispersal via animal dung in areas with increased animal activity (Barnes *et al.,* 1997) or due to the simultaneous germination of seeds in the seed bank following a rain event (Barnes, 2001; Seymour, 2008; Van der Merwe *et al*., 2019; Van der Merwe *et al.,* 2020). These cohorts could, however, be trees from different age classes which were suppressed by a grass layer that will grow to the next size class once the competition for light by surrounding grass has been eliminated as a results of grazer activity (Seymour, 2006).

2.4.6 Die-back

Research by Barnes *et al.* (1997) indicates that canopy die-back is an indicator of the severity of stress in the physiological condition of adult *V. erioloba* trees and could be used to monitor the decrease in *V. erioloba* populations. Shadwell and February (2017) found that percentage canopy die-back is not dependent on seasonal changes. Except for environmental factors influencing the growth of a tree, as age may also lead to die-back of branches (Milton & Dean, 1995 as cited by Barnes *et al*., 1997). The presence of wood-boring beetles that make tunnels underneath the bark of the tree, thereby effectively ring-barking the branches, is also common in dead branches on *V. erioloba* trees (Barnes *et al*., 1997).

2.5 Management

It is important to manage *V. erioloba* populations effectively since these populations are under threat due to various factors mentioned above, and because of their legislated protected status and the important ecological role they fulfil within a specific area. As pointed out by Barnes *et al.* (199, as cited by Morkel, 2013), *V. erioloba* is regarded as an important species and where better management can have a huge impact on their survival and sustainability. In this regard though, Dean *et al.* (1998) cautioned that management of the population structure of *V. erioloba* is just as important as protecting individual trees, because only by creating spatial heterogeneity in a landscape can the role of *V. erioloba* as a keystone species be maintained.

Li *et al.* (2014) state that knowledge of at least two structural characteristics of a population can help to understand the population characteristics of a woodland, leading to the better management of a specific species. Furthermore, long-term monitoring of a species can present insightful knowledge of past management outcomes and lead to better management decisions in future (Lindenmayer & Likens, 2009).

Management actions can affect the population structure, distribution and density of *V. erioloba* (Steenkamp *et al.,* 2007). Mismanagement can lead to savanna areas becoming bush encroached or being transformed into grasslands (Seymour, 2006). The correct management of these woodlands will be far more efficient than trying to reverse changes in the vegetative structure resulting from bad veld management (e.g. becoming bush-encroached) (Van Rooyen *et al.,* 2016).

Creating protected areas for *V. erioloba* woodlands will eliminate disturbance factors such as overutilisation and will contribute to better management opportunities where only natural threats need to be studied (Van der Merwe *et al.,* 2019). Furthermore, managing the utilisation by livestock and game can also have a huge impact on the successful germination and establishment of *V. erioloba* seedlings (Hoffman *et al.,* 1989). However, as much as devising appropriate methods to manage *V. erioloba* woodlands is important and must be absolutely clear about who will be held accountable for implementing good management principles and ensuring these woodlands are sustainable over the long term (Liversidge, 2001).

CHAPTER 3 STUDY AREA

3.1 Locality

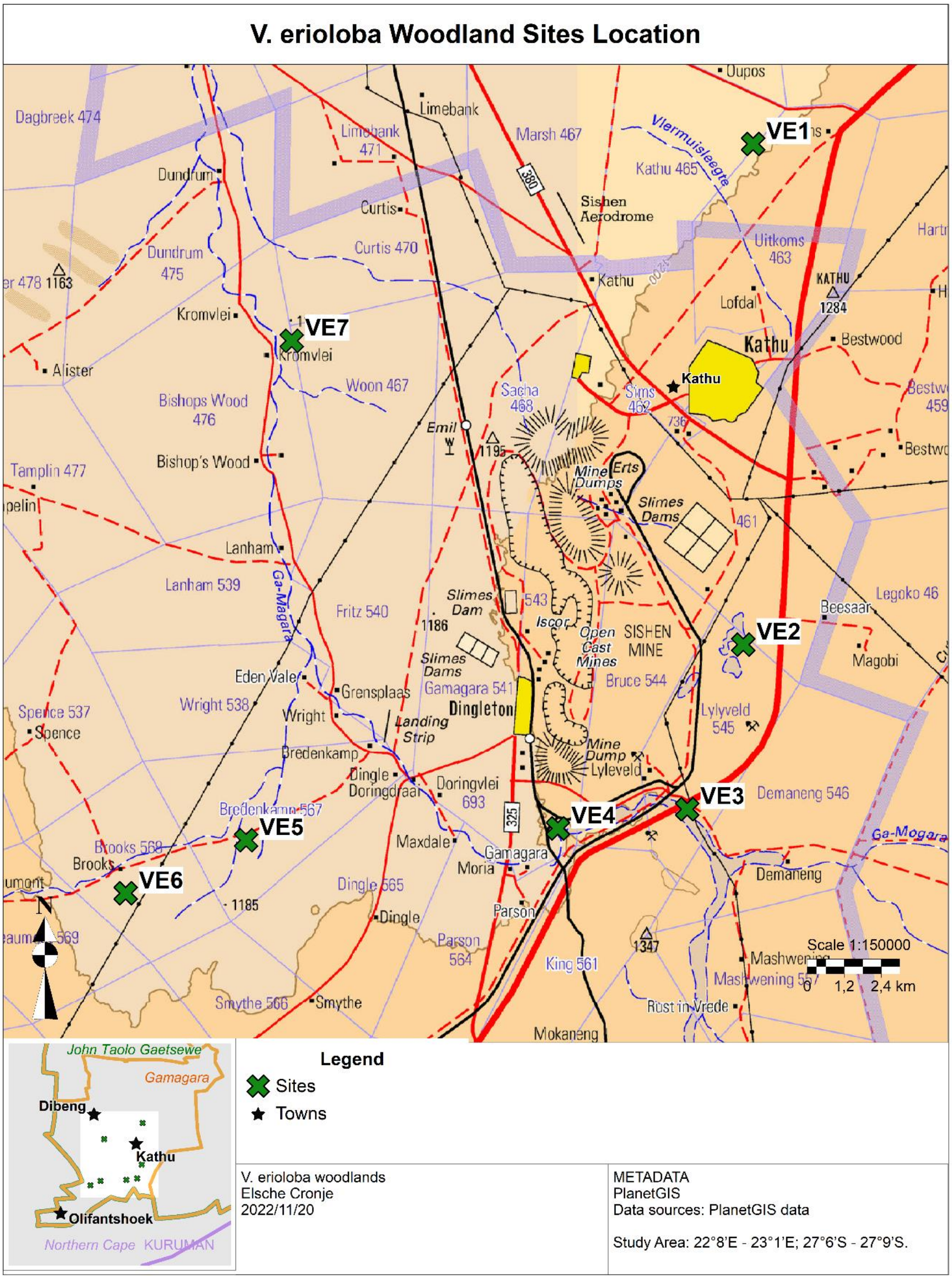

Figure 3-1: Locality of Kathu within the Northern Cape Province indicating the locality of town and study sites

The study was carried out at selected sites near the town of Kathu in the Northern Cape Province of South Africa where populations of *Vachellia erioloba* woodland occur (Figure 3.1) (VE1 – VE7). Kathu lies within the Gamagara local municipality and John Taolo Gaetsewe district municipality.

3.2 Vegetation

The study area is situated within the Savanna Biome, representing 32.8% of South Africa and, more specifically, within the Eastern Kalahari Bushveld Bioregion (Mucina & Rutherford, 2006) (Figure 3.2). Werger and Coetzee (1978) broadly described the vegetation unit of this bioregion as “Open *Acacia* savanna of the southern Kalahari”. According to Mucina and Rutherford (2006), the study area resort under the arid/eutrophic savanna or fine-leaved savanna, representing the entire Eastern Kalahari Bushveld, Lowveld, Sub-escarpment Savanna, Kalahari Duneveld and Mopane Bioregions as well as the lower-lying region of the Central Bushveld Bioregion.

3.2.1 Vegetation types

Four (4) of the sites are located within the Kathu Bushveld vegetation type two (2) within the Olifantshoek Plains Thornveld vegetation type and one (1) within the Kuruman Mountain Bushveld (Figure 3.2).

3.2.1.1 Kathu Bushveld

The Kathu Bushveld lies at an altitude of 960-1300 m extending from Kathu to Botswana’s border between Van Zylsrus and McCarthysrus within the Northern Cape Province (Mucina & Rutherford, 2006). This vegetation type was originally described as Kalahari Thornveld and Shrub Bushveld (Acocks, 1953) and later as part of the Kalahari Plains Thorn Bushveld (Low & Rebelo, 1996). The landscape is dominated by a shrub layer, frequented with a medium to tall tree layer and a variable grass layer (Mucina & Rutherford, 2006; SANBI, 2006). *V. erioloba* and *Boscia albitrunca* are the main tree species present within this landscape (Mucina & Rutherford, 2006; SANBI, 2006). Even though Mucina and Rutherford reported in 2006 that the conservation target of 16%, is not being adhered to since no areas are conserved statutorily, in 2016, Van Rooyen *et al*. categorised this vegetation type as least threatened with only approximately 2.5% of the total area transformed.

3.2.1.2 Olifantshoek Plains Thornveld

The Olifantshoek Plains Thornveld lies at an altitude of 1 000-1 500 m and extends from Olifantshoek to Griekwastad, across the plains of the Korannaberg, Langeberg and Asbestos Mountains within the Northern Cape Province (Mucina & Rutherford, 2006; SANBI, 2006). This vegetation type was previously described as Kalahari Thornveld invaded by Karoo and Kalahari Thornveld and Shrub Bushveld (Acocks, 1953) and later as part of the Kalahari Mountain Bushveld (Low & Rebelo, 1996). Mucina and Rutherford (2006) described the landscape as wide, diverse plains with a sparse grass layer and open tree and shrub layers. *V. erioloba, B. albitrunca, V. luederitzii* and *Searsia tenuinervis* being the dominant tree species (Mucina & Rutherford, 2006; SANBI, 2006). As mentioned, with a conservation target of 16%, Mucina and Rutherford (2006) categorised this vegetation type as least threatened since only 0.3% is conserved within the Witsand Nature Reserve and only 1% of the area has been transformed (Mucina & Rutherford, 2006).

3.2.1.3 Kuruman Mountain Bushveld

The Kuruman Mountain Bushveld lies at an altitude of 1 100-1 800 m and extends across the Northern Cape and North West Provinces, from Griekwastad and the Asbestos Mountains towards the Kuruman Hills (Mucina & Rutherford, 2006). This vegetation type was previously described as Kalahari Thornveld and Shrub Bushveld (Acocks, 1953) and later as Kalahari Plains Thorn Bushveld (Low & Rebelo, 1996). Slight to moderate sloping hills with open shrubland and a well-established grass layer are characteristic of this landscape (Mucina & Rutherford, 2006) but, rather than *V. erioloba,* tree species such as *Searsia lancea* are more prevalent here (Mucina & Rutherford, 2006; SANBI, 2006). This vegetation type is categorised as least threatened with a conservation target of 16%, with currently no conserved areas and only a few transformed areas (Mucina & Rutherford, 2006).

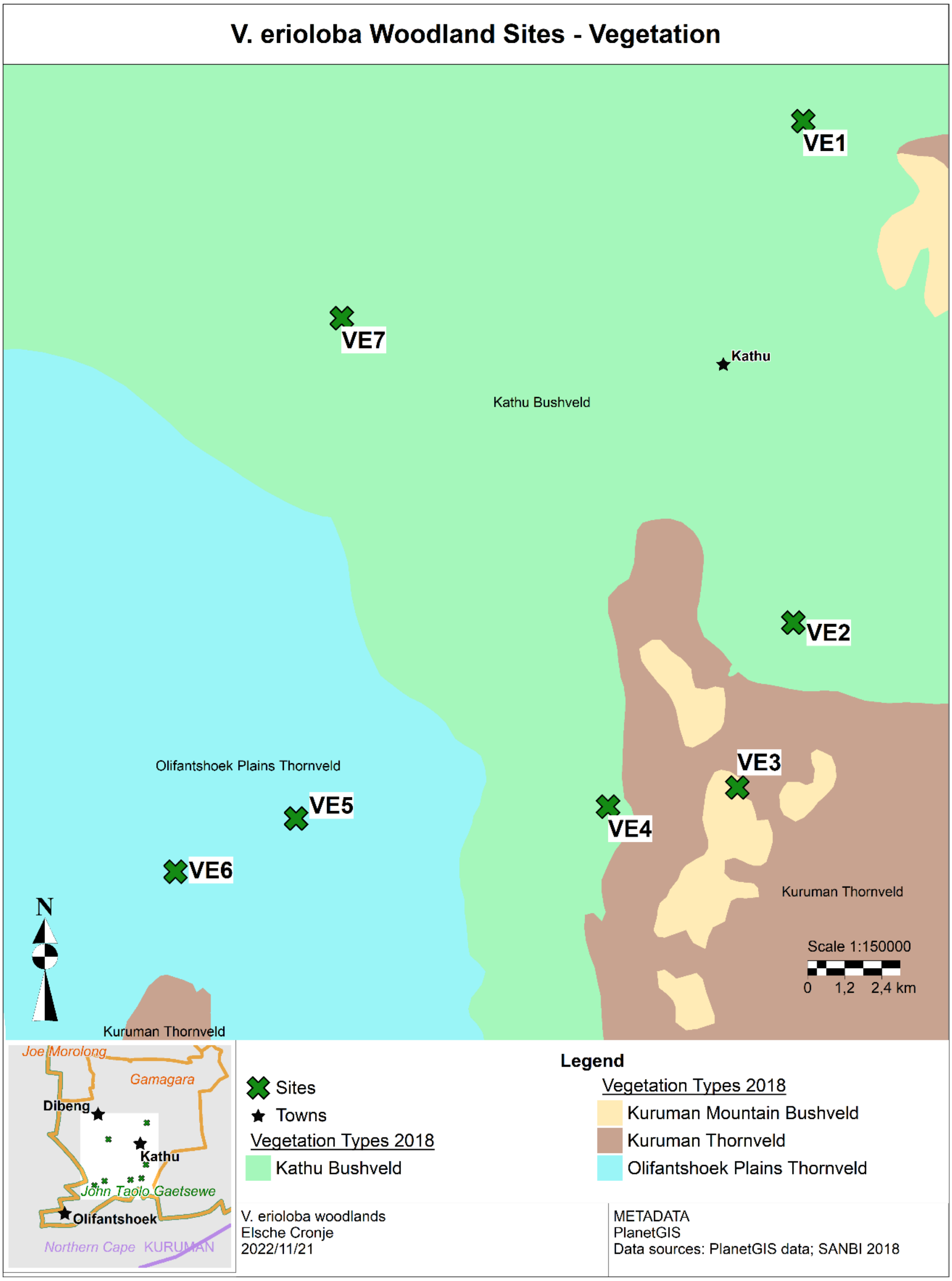

Figure 3-2: Vegetation map indicating the locality of the selected sites in the different vegetation types

3.2.2 Vegetation associations

Van Rooyen *et al.,* (2016) conducted a revision of the Kathu Bushveld vegetation type (Mucina & Rutherford, 2006) and a 10 km buffer area as part of a research offset agreement. Accordingly, various plant associations were defined and described, and it is based on this newly proposed delineation that four (4) sites are located within the Kathu Bushveld (VE1, VE5, VE6 and VE7) and three (3) within the Kathu Shrubland (VE2, VE3 and VE4) vegetation types. These vegetation associations as described by Van Rooyen *et al.* (2016) are discussed below.

3.2.2.1 Kathu Bushveld (new delineation) (Van Rooyen *et al.,* 2016)

The Kathu Bushveld consists mainly of bushveld, with some areas being woodland. The dominant tree species is *V. erioloba* and the dominant shrub is *S. mellifera*. Five (5) plant associations, as described by Van Rooyen *et a*l. (2016), form part of this vegetation type (A1-A5). Three (3) sites are located within the A4 association (VE5, VE6 and VE7) and one (1) site within the A5 association (VE1) as explained below.

- A4: *Acacia haematoxylon – Pogonarthria squarrosa – Tarchonantus camphoratus* Bushveld

Situated west of the Kuruman Hills and Gamagara River and extending to Olifantshoek, this association is largely located within the Kathu Bushveld with a section located within the Kuruman Thornveld and Olifantshoek Plains Thornveld as described by Mucina and Rutherford (2006). This association can be described as bushveld with *V. erioloba* as the dominant tree species and *S. mellifera* and *T. camphoratus* as the dominant shrubs. Apart from a dwarf shrub layer, an herbaceous layer, and a moderately developed grass layer, only a few geophytes and no succulents are present in this association.

- A5: *Eragrostic pallens – Plinthus sericeus – Schmidtia pappophoroides – Acacia erioloba* Woodland

This association surrounds the declared Kathu Forest near Kathu and is situated entirely within the Kathu Bushveld vegetation type. It consists of open woodland and bushveld with the dominant tree species being *V. erioloba* and dominant shrub species being *S. mellifera* and *T. camphoratus*. As in the case with the A4 association described above, apart from a dwarf shrub layer, an herbaceous layer, a moderately developed grass layer, only a few geophytes and no succulents are present in this association.

3.2.2.2 Kathu Shrubland (Van Rooyen *et al.,* 2016)

The Kathu Shrubland is described as a dense shrubveld dominated by *T. camphoratus* and *S. mellifera*. It extends from the plains, on the eastern side of the Gamagara River, situated between Kathu and Deben, to Hotazel. All three (3) sites within this vegetation type display the A2 vegetation association as described below.

- A2: *Digitaria eriantha – Pentzia calcarean – Enneapogon desvauxii – Coelachyrum yemenicum* Shrubland

This association extends from Dingleton northwards to Hotazel and is situated within the Kathu Bushveld vegetation type as mapped by Mucina and Rutherford (2006). The A2 association can be described as a dense shrubveld dominated by *S. mellifera* and *T. camphoratus.* A very sparse tree layer can be found in most of this association, with *Ziziphus mucronata* as the dominant species and only scattered *V. erioloba*. Even though the vegetation association found in the A2 association is somewhat similar to that of A4 and A5 associations (dwarf shrub layer, herbaceous layer), the grass layer is poorly developed and only some succulent species and hardly any geophytes are present.

3.3 Climate

In South Africa, the Savanna Biome is situated at low altitudes ranging from 1 500 m to 1 800 m above sea level, whereas the surrounding grasslands are situated at higher altitudes resulting in lower temperatures (Mucina & Rutherford, 2006).

The average annual maximum and minimum temperatures for Sishen Mine, which is near Kathu, is 27°C and 12°C respectively (Shangoni, 2017). January is the hottest month of the year with an average maximum temperature measuring 32.9°C, but as Shangoni (2017) pointed out, extremely high temperatures of 45°C and above are often recorded in the months of December and January. July is the coldest month of the year with an average minimum temperature measuring 3.1°C (Shangoni, 2017). The winter months extend from June to August, but frost - a regular occurrence during the winter months – has been recorded as early as mid-May and as late as September (Van Rooyen *et al.,* 2016; Mucina & Rutherford, 2006).

Falling in South Africa’s summer rainfall region (December to February interspersed with very dry winter months), Mucina and Rutherford (2006) estimated the Mean Annual Precipitation (MAP) of the western part of the study area (i.e. the Olifantshoek Plains Thornveld) to be between 200 mm and 350 mm, that of the Kathu Bushveld between 220 mm and 380 mm and between 250 mm and 500 mm for the Kuruman Mountain Bushveld. The actual average rainfall recorded at Sishen

Mine, within the study area for the period of the study (2014 to 2019) was 392 mm, with 2014, 2016 and 2017 presenting with above average rainfall and 2015, 2018 and 2019 with below average rainfall. The highest amount of rainfall was recorded for 2014 (665 mm), whereas 2019 received the least rain (196 mm).

Based on UNEP's Global Humidity Index (2002) which uses a ratio between annual precipitation and potential evapotranspiration (described as P/PET) to indicate mean annual potential moisture availability, the study area would be classified as a semi-arid zone P/PET ranging between 0.2 and 0.5. In contrast, the Agricultural Geographic Information System (AGIS, 1997) classifies the study area as an arid zone with the eastern part of the study area bordering between an arid and semi-arid zone. Due to these differences, the study area falls within a semi-arid and an arid region.

The prevailing wind direction during daytime is from the northwest with wind speeds regularly measuring more than 5 m/s (Shangoni, 2017; AGES, 2014;). During the night, the wind blows predominately from the southeast with calmer conditions and decreased velocity than daytime winds. In this regard, note that sites VE2, VE3 and VE4 are located within the prevailing wind direction that creates dust pollution originating from the mining activities within the area (see Chapter 3 for site descriptions).

3.4 Geology / soils

The basic lithology of the study area is classified as Unconsolidated Eolian (aeolian) (UE) of the Gordonia formation, in other words sediment not compacted or hardened consisting of fine to medium sand and coarse silt particles (Van Engelen & Dijkshoorn, 2013; Partridge *et al*., 2006). The Gordonia formation is the top layer of the seven Kalahari Group sediment formations and is commonly known as "Kalahari sands" (Partridge *et al*., 2006). These sands can be up to 30 m deep and can rest directly on pre-Kalahari bedrock or lay on underlying calcrete surfaces.

According to the World Reference Base (WRB) for soil resources (IUSS WG, 2007 as cited by Van Engelen & Dijkshoorn, 2013), three soil units are found within the study area, Rhodic Cambisols (CM-ro), Calcic Solonchaks (SC-cc) and Rubic Arenosols (AR-ru) (Table 3.1). Further, the generalised soil patterns in the study area indicate the presence of three (3) soil types, namely CM, LP2 and AR2 (Table 3.1).

The soil unit for the western part of the study area is Rhodic Cambisols (CM-ro). Rhodic Cambisols are characterised as brown soils with a Munsell hue of 2.5 YR or redder according to the Munsell colour system (Van Engelen & Dijkshoorn, 2013). Rhodic Cambisols have a subsurface layer of 30 cm or thicker within 150 cm of the soil surface, with a moist value of less than 3.5 and a dry value of less than one (1) unit higher than the moist value (Van Engelen & Dijkshoorn, 2013). The generalised soil pattern for this part of the study area is CM, and can be described as red soils with a high base status (CM) (AGIS, 2004).

The eastern part of the study area consists of Rubic Arenosols (AR-ru), commonly called sandy soils, where Rubic (ru) can be described as soils with a subsurface layer of 30 cm or thicker within 100 cm of the soil surface, a moist value of five (5) or more and a Munsell hue of 10 YR or redder (Van Engelen & Dijkshoorn, 2013; ISRIC, 2004). The Agricultural Geographic Information System (AGIS, 2004) classifies the generalised soil pattern of this part of the study area as AR2 soils, i.e. well-drained, red/yellow, sandy soils with a high base status.

The soil unit for the central region of the study area is Solonchacks, also called saline soils, with Calcic (cc) as the second-level classification. The International Soil Reference and Information Centre describes calcic soils as soils having a calcic horizon or concentrations of secondary carbonates within 100 cm off the soil surface (ISRIC, 2004). AGIS (2004) classifies these soils as LP2, i.e. shallow soils with minimal development, found on hard or weathering rock, with or without intermittent diverse soils, where lime is present in most of the landscape.

In terms of the soil's water-holding capacity, in other words the amount of water available within the soil at root level (Van Rooyen *et al.,* 2016), AGIS (2004) estimated that this capacity ranges from 61 - 80 mm in the western part of the study area to less than 20 mm in the eastern part of the study area (Table 3.1).

The land types found within the study area are Ae to the south and Ag to the north. Both land types comprise red, high-base soils, where Ae is >300 mm deep and Ag <300 mm deep (LTSS, 2021) (Table 3.1). In this regard note that, according to Barnes *et al.,* 1997, *V. erioloba* is mostly found in deep sand over limestone within the Kalahari region.

According to the study conducted by Van Rooyen *et al.* (2016), the soil present at site VE2, VE3 and VE4 has free lime in the A horizon with 9% clay, 7.8% silt and 83.2% sand content. The soil at site VE1 has the highest sand percentage (92.7%) with 8% clay and no silt, whereas the soil at site VE5, VE6 and VE7 has 85.8% sand, a high clay component of 11.2% and only 3% silt.

No detailed soil analysis was carried out at the different sites for this study.

Table 3-1: Summary of soil type, land type and water holding capacity at each study site with their corresponding vegetation types (LTSS, 2021; Van Engelen & Dijkshoorn, 2013; AGIS, 2004; ISRIC, 2004)

Site ID	WRB Soil Unit	Generalised soil pattern	Land type	Water holding capacity (mm)	Vegetation type
VE1	SC-cc	LP2	Ag	<20	Kathu Bushveld
VE2	SC-cc	LP2	Ag	<20	Kathu Bushveld
VE3	AR-ru	AR2	Ae	21-40	Kuruman Mountain Bushveld
VE4	SC-cc	LP2	Ag	<20	Kathu Bushveld
VE5	CM-ro	CM	Ae	61-80	Olifantshoek Plains Thornveld
VE6	CM-ro	CM	Ae	61-80	Olifantshoek Plains Thornveld
VE7	SC-cc	LP2	Ag	<20	Kathu Bushveld

3.5 Topography / hydrology

With an altitude ranging between 1 189 m and 1 200 m above sea level, the topography of the study area is indicated as plains with slopes of 0-10% and relief intensities of less than 100 m/km (DRDLR, 2016; Van Engelen & Dijkshoorn, 2013; ISRIC 2003).

In terms of hydrology, the study area is situated within the D41J quaternary catchment of the lower Orange River primary catchment (D), (DWS, 2011). The ecological importance and sensitivity category (EISC) of this catchment is categorised as low, indicating a catchment that is not unique with rivers that are generally not very sensitive to flow modifications (Kleynhans, 2000). The present ecological status class (PESC) for this catchment is rated as B (largely natural).

The Gamagara River is the only main drainage system within the study area, flowing from the north to the south-east toward the Kuruman River. The Gamagara River is an ephemeral river and is dry for most part of the year, except after heavy rainfalls when flow is present for a few hours or days (Pietersen and Gaffoor, 2008). Flood events are unpredictable and can occur several times in one year or only once in a decade (Schachtschneider, 2010). Shallow aquifers are present within the vicinity of the Gamagara River within the Kalahari sediments on the calcrete beds (Pietersen & Gaffoor, 2008). Many ephemeral pans are also present within the study area with the majority located to the east of the Gamagara River (CSIR, 2011).

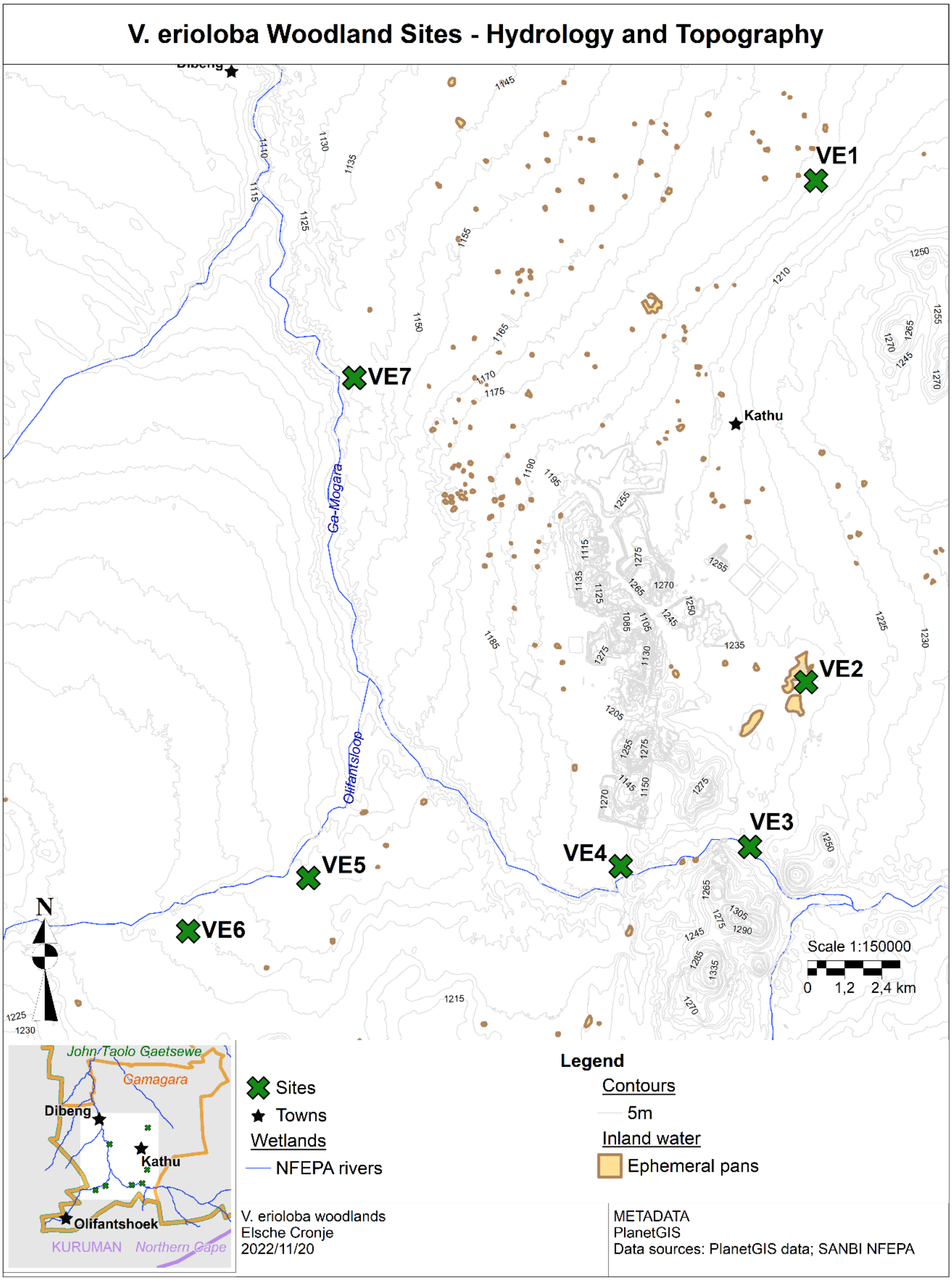

Figure 3-3: **Topography with contours (5 m interval), ephemeral pans and National Freshwater Ecosystem Priority Areas (NFEPA) rivers (CSIR, 2011) showing the hydrology of the study area and surrounding villages in the *V. erioloba* woodland**

3.6 Site selection

A total of seven (7) sites where *V. erioloba* populations occur were chosen (VE1 – VE7), representing different areas at various distances from the possible impacts of developing agencies, such as dust pollution and de-watering. Note that various mining activities are present within the study area, which are situated close to VE2, VE3 and VE4. Further, a recently built solar park is situated close to VE1. A number of variables such as the occurrence of fire, differences in woodland population age and structure and available water resources (e.g. distances to riverbeds or topographical units), were taken into account for site selection. The de-watering (Shangoni, 2017) and dust pollution zone of influence (ZoI) of the Sishen Mine were also taken into account during site selection, with due consideration of the prevailing wind direction for each site. The ZoI was calculated by a 2 km radius from all mining activity (pits and dumps).

Initial site selection was based on forest maps compiled by the Council for Scientific and Industrial Research (CSIR, 1988) and ground truthing was done to select the seven (7) sites.

The seven (7) sites selected with descriptions of location and other important aspects (such as current and past disturbances, land use and distance from wetlands) are mentioned in Table 3.2.

Table 3-2: Description of site location and disturbances for all sites assessed (VE1-VE7)

Site number	Description of site location
VE1	Site VE1 is located on the Kathu Farm 465, furthest north-east of all the sites, located 5,6 km from the town and furthest from any mining activities (9,9 km). The closest development to this site is the Kathu Solar Park, located 1,8 km to the north-west of the site. A gravel road, frequently used by those in the employ of the solar park, is located 1,5 km to the north of the site. The site is situated on a portion of the farm set out as an offset for the Kathu Solar Park development and do not have any livestock or game, except for a few ostriches utilising the area. This site is far from a river (7,2 km) but close to some ephemeral pans, with the closest one being 0,5 km. This is further the only site located within the declared "Kathu Forest". (-27,6275535 S; 23,0579292 E)
	

VE2	VE2 is located 5,8 km south of Kathu, only 1,7 km east of the Sishen Mine. This site is located on Lylyveld Farm 545, situated within the de-watering and dust ZoI of the Sishen Mine and well within the prevailing wind direction. A national route (N14) is situated 1,3 km to the east and a railway 1,1 km to the west. This site was also subject to a fire that occurred within the area in 2011. Both livestock and game were farmed on this farm in the past. Site VE2 is located far from a river (5,5 km) but buffers on a large ephemeral pan. (-27,7734306 S; 23,0546859 E)
	

VE3	VE3 is located 11 km to the south of the town Kathu, close to the N14 (0,3 km) to the north-west of the site, a gravel road used for mine traffic to the north-east of the site (0,2 km) and mining activities 0,5 km to the south-west of the site. Furthermore, this site is located on Lylyveld Farm 545 where a fire occurred in the course of 2011. Note, too, that VE3 is situated on the floodplain of the Gamagara River, within the de-watering and dust ZoI of the Sishen Mine and directly within the prevailing wind direction. No access control is in place at this site. The land use of this farm was previously livestock farming. (-27,8213509 S; 23,0363712 E)
	

VE4	VE4 is situated on Sishen Farm 543 with mining activities to the north (1,1 km) and to the south (1,5 km) of the site. Further the N14 is situated 1,2 km south-east of the site, a main road (325) 1,4 km to the west, a secondary road 1,2 km to the north and railway lines 0,2 km to the north and west of the site. The site is located within a highly fragmented area with various gravel access roads in close proximity. VE4 is also located within the de-watering zone and dust ZoI of the Sishen Mine and is situated on the floodplain of the Gamagara River. The land use of this farm was previously livestock farming. (-27,8269844 S; 22,9940988 E)
	

VE5	VE5 is located on the newly proclaimed Sishen Nature Reserve on Bredenkamp Farm 567. This site has few disturbances with the closest mining activities 7 km to the south-east and the closest road 4,8 km to the north-east. The nearest town is Olifantshoek which is located 17 km to the south-west of the site. The stream Olifantsloop is located 0,2 km north-west of the site. Game has recently been introduced on this farm, where it was previously occupied by livestock. (-27,8304317 S; 22,8920406 E)

VE6	VE6 is the second site located on the newly proclaimed Sishen Nature Reserve. This site is located on Brooks Farm 568 and is the site located furthest from any mining activities or developments (10 km and 14 km respectively). The stream Olifantsloop is located 0,7 km north of the site. Game has recently been introduced on this farm, where it was previously occupied by livestock. (-27,8458899 S; 22,8526542 E)

VE7	VE7 is located on Bishops Wood Farm 476 north-west of the Sishen Mine, where the mine is currently expanding. The site is located close to a railway (1,1 km) and a secondary road (0,7 km) but relatively far from mining activities (3,6 km). Various access roads surround the site with a farm house located 0,7 km to the south-west. The Gamagara River runs to the west of the site. The land use of this farm is livestock farming. (-27,6848952 S; 22,9060879 E)

CHAPTER 4 MATERIALS AND METHODS

4.1 Site layout

All sites were 50 m wide with corner A and corner D permanently marked (Figure 4.1). Each site differs in size, since the length of the site was extended up to a point where 50 adults *Vachellia erioloba* trees could be assessed. Lines B and C were at a right angle to the permanent back line (corner A to D) (Figure 4.1). For the purpose of this study a specimen was regarded as an adult tree with a height of 1.5 m and above. All specimens were assessed, whether dead or alive. A tree was recorded as dead when no foliage was present on the tree, whether on the crown or on any resprouts.

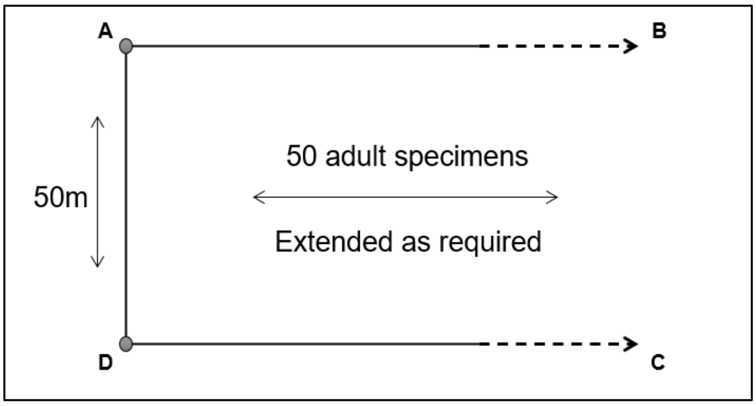

Figure 4-1: Site Layout. The distance between points A and D was 50 m. The distance from A to B and D to C differed and was extended up to a point where 50 adult specimens could be recorded in the site.

4.2 Desktop analysis

A desktop analysis, which included a spatial analysis was carried out for each of the sites described in Chapter 3.6 to determine the following (also see Tables 3.1 and 3.2 for site descriptions):

- The vegetation type for each site was determined by overlaying the VEGMAP, version 2018 (SANBI, 2006) on the sampling sites;
- The distance of each site from a wetland was determined by overlaying the NFEPA rivers geospatial dataset (CSIR, 2011) on the sampling sites;
- The soil and land type for each site was determined through the use of the World Reference Base for soil resources (Van Engelen & Dijkshoorn, 2013);

- The distance of each site from roads and other disturbances were determined by the use of Google Earth Pro (2022);
- Rainfall data was obtained from Sishen Mine for the study period; and
- Management strategies applied at each site were noted.

4.3 Assessment schedule

Three (3) assessments were conducted over a period of five (5) years. To include the main pod bearing season for *V. erioloba* trees (Palgrave, 2002), the first assessment was conducted in January 2015, but since very few pods were observed during the first assessment, the second assessment was moved to September 2016. During the second assessment, still very little pod production was observed, which indicated that all populations produce pods at different periods. The third assessment was therefore carried out in January 2019.

4.4 Tree profile

By using a handheld Global Positioning System (GPS), the coordinates and a photo of each adult tree was taken for record purposes. Direct measurements of the tree's height, height of first leaves and canopy diameter at the broadest part were taken by using a calibrated measurement rod. To arrive at a suitable evaluation methodology that would serve as a baseline for the assessment of the *V. erioloba* trees, the Department of Forestry, Fisheries and the Environment (DFFE) was consulted (DAFF, 2015a). Accordingly, only the main tree canopy was measured, ignoring small protruding side branches. The circumference of trunks was taken at a height of 50cm above ground level. The tree height was divided into the following six (6) height classes, i.e. saplings, class one (1) and two (2) and adult plants represented by classes three (3) to six (6) (Table 4.1).

Table 4-1: Height class distribution indicating the six height classes used to measure the tree heights at each site

Class	Height (m)	
1	0.0 to < 0.75	Saplings
2	0.75 to < 1.5	
3	1.5 to < 2.5	Adults
4	2.5 to < 3.5	
5	3.5 to < 5.5	
6	≥ 5.5	

4.5 Quantitative assessment

The following quantitative data were recorded for each tree in a *V. erioloba* population per site:

- The number of saplings per site.
- The number of bird nests were counted and recorded.
- The number of pods present on each tree were counted and recorded, as well as the number of newly dropped pods underneath the tree canopy.
- The number of *Gonometa. postica* cocoons, moths and caterpillars found within each tree were counted and recorded.

4.6 Qualitative assessment

Qualitative data was recorded in addition to the quantitative data described above. The qualitative data included the estimation of resprouting, percentage canopy flowering, bark damage, presence of resin on the tree trunk, and presence and appearance of lichens.

Apart from paying particular attention to the presence or absence of Bruchidae beetle predation within pods and the presence of large or small wood-boring beetles and the latter’s location, all types of living insects occurring on the tree trunk were noted. In this regard, particular attention was paid to the presence of insect nests, as well as the absence or presence of spines, swollen at the base, that could serve as a habitat for a variety of insects.

The percentage of canopy die-back of each tree, was recorded according to a graded scale ranging from low to high (Table 4.2).

Table 4-2: Scale to determine canopy die-back, ranging from low to high given in percentages.

Percentage canopy die-back	Scale
0% to 25%	Low
>25% to 50%	Moderate
>50% to 75%	Relatively high
>75% to 100%	High

**0% canopy die-back is considered as very healthy (no die-back), whereas, 100% die-back equates to a dead tree.*

4.7 Data analysis

All quantitative and qualitative data recorded was used to determine the structure and health of the trees of the *V. erioloba* populations per site and to compare the populations between sites. Three years' monitored data (every second year over a period of five years) was used in the data analysis.

All living adult specimens were taken into account to determine the production rate of a population. Further, the health of the potential seed bank per productive adult tree was determined by the Bruchid predation present on dropped pods. The number of saplings recorded was used to determine the recruitment and persistence of a population. The establishment of the population can be determined by the number of class 1 (<0.75 m) saplings compared to the number of class 2 (<0.75 m to <1.5 m) saplings per population present per year.

The health of the population was determined by the die-back per tree height class per year between assessments. The health of each population is then compared in terms of past and present impacts that could have contributed to the survival of the population per site.

4.8 Statistical analysis

R version 3.6.2 (2019-12-12) and RStudio V 1.2.5033 software were used for all statistical analysis run for this study (R Core Team, 2022). The following packages were used:

- Base packages: stats; graphics, grDevices; utils; datasets; methods; base
- Other packages: ggplot2_3.2.1; janitor_2.1.0; emmeans_1.7.2

Boxplots were generated to visually compare the different sites (VE1 – VE7), regarding the height structures, average circumference, die-back per tree and mortality of each population.

One-way ANOVA tests were run on the model to test the difference in means between the sites and each variable mentioned above. Tukey contrast for multiple comparisons between groups, with a confidence level of 0.95 were run on each variable.

The correlation between die-back or mortality and the presence of small/large wood-boring beetles were also tested by way of pairwise comparisons.

CHAPTER 5 RESULTS & DISCUSSION

In order to draw a comparison between the sites as described in Chapter 3, the results of all data as reported in Chapter 4 will be discussed in this chapter, with specific reference to production, seed banks, recruitment, establishment, size class distribution, die-back and mortality. Based on those results, this chapter will conclude by discussing past and present environmental stressors that could impact the structure and health of the respective *Vachellia erioloba* populations.

5.1 Production of flowers and pods

Barnes *et al.* (1997) found that the flowering period of *V. erioloba* trees is usually relatively short (as little as six weeks) and mainly occurs in spring, however it varies from population to population and can start during the winter. Late frost can thus deter the development of flowering buds (Seymour & Milton, 2003).

This was a point-in-time assessment of each *V. erioloba* population and not a seasonal production assessment, thus no repeat assessments were conducted during one season to determine total production for each year. Although flowering individuals were noted in this study, no assessments were carried out during the main flowering season (spring) in 2015 and 2019. Consequently, no flowering individuals were recorded during 2015 with only one individual noted at site VE3 during 2019 (see Chapter 3 for description of sites).

The assessment in 2016 was, however, carried out during September, which coincided with the flowering period. Only two individual trees were recorded with flowers (one within site VE3 and one within site VE4). Both these sites are situated on the banks of the Gamagara River (see Figure 3.3 for site locations in reference to the river system). This corresponds with the study conducted by Joubert in 2002 who reported that the highest number of flowering and pod-bearing specimens were found in the river channel or on the river banks.

The pod-bearing season generally extends from December to March and the pods will drop from April (Palgrave, 2002). Since pod production is important for the future sustainability of a population, assessments in this study were initially scheduled for January each year so as to include the pod-bearing period. It was, however, still only a point-in-time assessment and very few pods were recorded during the first assessment in January of year 2015. Consequently, the assessment period was moved to September in 2016 but since no change was observed, the third assessment (i.e. 2019) was moved back to January.

By the age of 20 years, *V. erioloba* trees should be able to produce large quantities of pods that can weigh up to 200 kg in total per year per adult tree (Schachtschneider, 2010; Barnes *et al.,* 1997). However, several studies found that fluctuations in groundwater and rainfall events can affect pod production (Schachtschneider, 2010; Seely *et al.*, 1979 as cited by Seymour and Milton, 2003; Nel *et al.,* 1985 as cited by Barnes *et al*., 1997), while Barnes *et al.* (1997) found that severe thunderstorms and strong winds can cause young pods to break off before they are fully developed, further reducing pod production.

The percentage productive adult *V. erioloba* were determined per site per survey period (2015, 2016 and 2019) for the three assessments (Figure 5.1). To determine the number of productive trees per site, all trees with flowers, pods or newly dropped pods were counted per site during the assessment periods. Only living adult trees were considered when determining the percentage productive adults for each population.

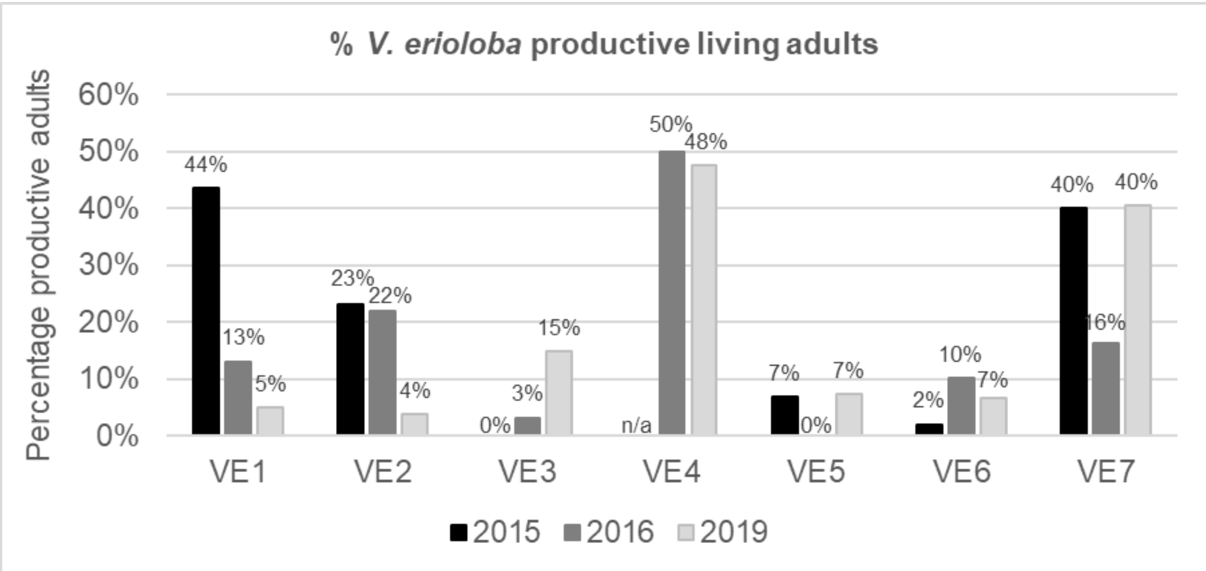

Figure 5-1: Percentage productive *V. erioloba* adults per site (VE1 – VE7) for the three assessment periods (2015, 2016 and 2019) (see Chapter 3 for site description)

When looking at the overall production capability of a *V. erioloba* population, the average of the three assessment periods can be taken into account. The highest average % productive adults were observed at site VE4, although no assessment was done at this site during 2015 (Average for 2016 and 2019 = 49%). Overall, the highest percentage productive adults (50% of the living adult population) were observed at site VE4 in 2016, as well as the second highest percentage productive adults (48% of the living adult population) observed during the 2019 assessment. On average, the *V. erioloba* population at site VE7 had the second highest percentage productive living adults (32%), with the highest percentage (40%) for this site recorded during 2015 and 2019 and the lowest (16%) during 2016.

The population at site VE5 had the lowest average percentage productive living adults (5%) for the three years of assessment, followed by sites VE3 and VE6 (both with an average of 6% productive living adults). The populations at sites VE5 and VE3 are also the only populations where 0% productive living adults were recorded during the three assessment periods (0% at VE3 in 2015 and 0% at VE5 in 2016). Since this was only a point-in-time assessment, it should be kept in mind that populations bear flowers and pods at different times, thus the assessments of each individual population did not necessarily take place at the optimal pod-bearing period for that specific population.

The results from the assessment contradict the study conducted by Joubert in 2002 (Joubert, 2022), where the populations at sites VE1 and VE2 (not located close to any river system) had a higher average percentage of productive adults over the three assessment periods than at sites VE3, VE5 or VE6 (all located within one kilometre from a river). This could be due to this species' extensive root system and the possible availability of ground water further away from river systems (Jennings, 1974). It should however be noted that both, site VE1 and VE2, had a lower percentage productive adults recorded during the 2019 assessment period than sites VE3, VE5 and VE6.

According to Powell (2005), dust pollution can also inhibit pod development in *V. erioloba* adults. This was not seen during this assessment, since VE2, VE3 and VE4 are the three sites located within the Sishen Mine de-watering and dust ZoI, of which VE3 was the only site with an exceptionally low average production. Site VE4, however, had the highest average production of all sites assessed.

Pod removal for fodder is known to be a problem in Kathu and the surrounding area (Van der Merwe, 2001), but due to strict access control in the mining activity area, this does not seem to be an issue in the assessed sites. VE3 is the only site where access control is lacking and where locals might have been able to collect pods.

Other impacts that could have an effect on the availability of ripe pods include high browsing pressure (Van der Merwe, 2001) and will manifest as a lack of dropped pods near productive trees. Since no individual trees presented high numbers of carried pods with a lack of dropped pods during the three assessment years (2015, 2016 and 2019) in any of the sites where *V. erioloba* populations occur (sites VE1 – VE7), browsing pressure is unlikely to be significant.

According to Joubert (2002), the number of pods produced depends on the canopy size of a tree and not on its age or height. Nevertheless, Van Wyk *et al*. (1985 as cited by Barnes *et al.*, 1997) found that the *V. erioloba* trees in the Kuiseb River valley did not produce any pods until they reached a height of 3 m. The smallest tree with any production found during this study was 3.5 m in height (recorded in 2016).

Regarding pod production per assessment year, no clear trend was observed (Figure 5.1). The population at site VE1 had the largest decline in percentage productive living adults from the 2015 assessment (44% of all living adults) to the 2019 assessment (5% of all living adults). The only other decline throughout the three assessment periods was observed at site VE2 (23% in 2015 to 4% in 2019). At site VE3, however, an increase in the percentage productive living adults could be seen from 2015 (0%) to 2019 (15%) as well as an increase at site VE6 (2% in 2015 to 7% in 2019). The fluctuation in pod production per year could be as a result of rainfall events.

5.2 Predation of pods and seed bank

Sabiiti and Wein (1987) found that the seed bank of *V. erioloba* can be used to determine the sustainability of a population. Several factors can however lead to the depletion of the seed banks, such as the removal of pods by humans or animals (Eriksson & Ehrlen, 1992), the consumption of unripe seed by animals such as primates (Barnes, 2001), the underdevelopment of seed (Van der Merwe, 2001) or seed predation by insects (Tybirk *et al*., 1994 as cited by Jeltsch *et al*., 1999).

Although the seed bank was not determined for the woodlands during this study, predation by bruchid beetles was assessed, as these beetles damage or destroy the seed embryos and render unviable seed (Van der Merwe, 2001; Halevy, 1974 as cited by Hoffman *et al*., 1989).

The percentage bruchid beetle predation at each site was calculated by determining the number of trees where bruchid predation was present as a percentage of the number of productive living adult trees. The percentage predation was calculated for both dropped and carried pods. Dropped pods had much higher infestation rates than carried pods.

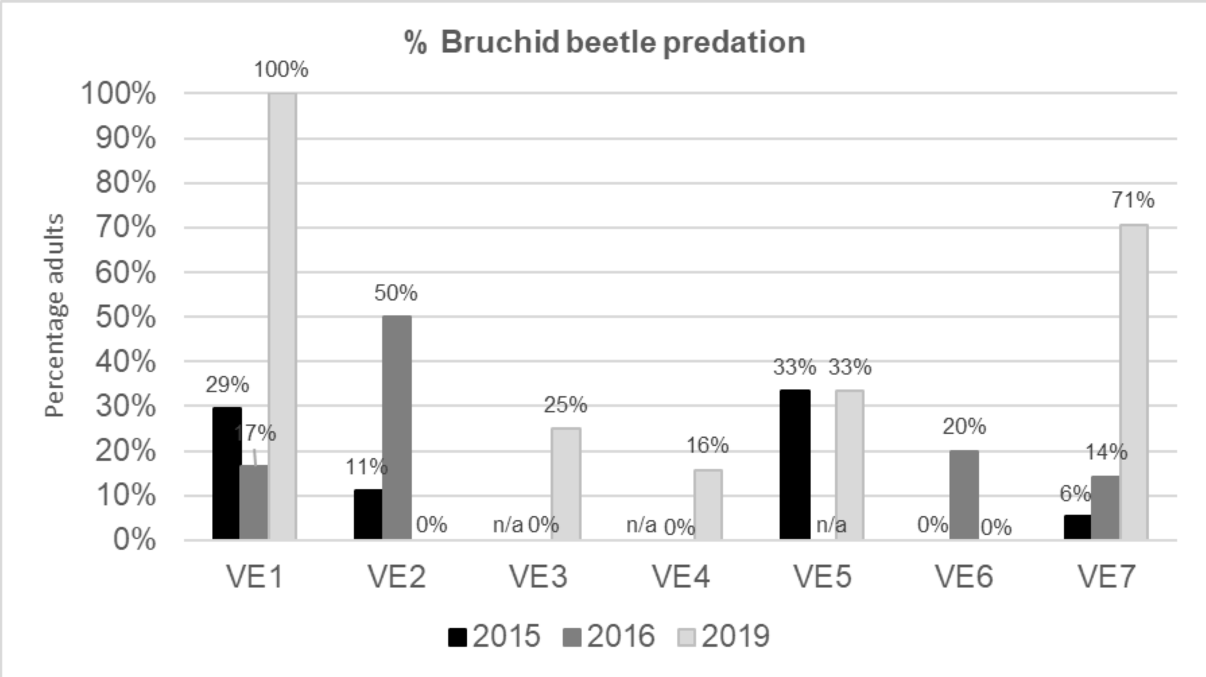

Figure 5-2: **Percentage productive adult *V. erioloba* trees presenting signs of bruchid beetle predation per site (VE1 – VE7) as recorded during the three assessment periods (2015, 2016 and 2019) (see Chapter 3 for site description)**

The percentage of trees affected by bruchid predation was only determined for living and productive adults within the population. Site VE3 had no productive adults recorded during 2015, thus the percentage bruchid beetle predation cannot be described as 0%, but rather as not applicable, as well as site VE5 which had no productive adults recorded during 2016. No assessment was conducted at site VE4 during 2015 and thus no data for bruchid infestation or productivity is available for this year assessment for this site.

Although the rate of infestation differed from site to site and from year to year, the predation by bruchid beetles was present in pods at all the sites. The highest average percentage of all three assessments of bruchid predation was found at site VE1 where 49% of all productive trees were infected (Figure 5.2). The second highest average predation was found at site VE7 for the three assessment years (30%). During the 2019 assessment, bruchid beetle predation was observed at 100% of all productive living adults at site VE1 (Figure 5.2) (5% of the total living adult population). The second highest infestation rate for a single assessment year was observed at site VE7 with 71% of all productive living adults infested during the 2019 assessment. At this site (VE7) 40% of all living adults were recorded as productive adults during 2019 (Figure 5.1), thus accounting for 28% of the total living adult population being infested by bruchid beetles.

Even though pods in the population occurring at sites VE3 and VE6 showed hardly any signs of bruchid predation (Figure 5.2), it ought to be noted that in 2015, 0% productive living adults were recorded at site VE3, in 2016 only 3% productive living adults and in 2019 only 15% (Figure 5.1). Likewise, at site VE6 a mere 2% productive living adults were recorded in 2015, 10% in 2016 and only 7% in 2019 (Figure 5.1). Data gathered from sites where too few pods were found might not be reliable since true bruchid infestation cannot be determined if the number of pods are too low.

The population at site VE4, is the only population where a very low percentage bruchid predation (0% in 2016 and 16% in 2019) was recorded (Figure 5.2) where a high percentage of productive living adults were present (50% in 2016 and 48% in 2019) (Figure 5.1). There are various possible reasons as to why the bruchid predation rate are higher at sites VE1 and VE7 than at VE4 and the specific reasons was not investigated.

Except for sites VE2 and VE6, the highest bruchid predation percentages were found in all sites during the assessment carried out in 2019 (Figure 5.2). The differing rates of infestation per assessment year were not investigated but, according to Traveset (1990), this might be ascribed to predation by ants or the destruction of bruchid eggs as a result of natural occurrences (i.e. a change in the ferocity of winds and fluctuations in temperature).

5.3 Recruitment

The less seeds predated or destroyed; the better recruitment is expected within a population. In order to determine the recruitment and persistence of a population, the number of saplings were recorded. As found in several studies (Van der Merwe *et al*., 2020; Van der Merwe *et al*., 2019; Seymour, 2008; Barnes, 2001; Jeltsch *et al*., 1999), *V. erioloba* saplings commonly emerge in large numbers at once, following on a good rainy period. However, as Jeltsch *et al.* (1999) pointed out, the availability of a viable seed bank is another determining factor in a population's ability to recruit sufficient saplings.

The percentage recruitment of each population was determined by calculating the number of saplings from the smallest height class (height class 1) as a percentage of the total population (saplings + adults) (Figure 5.3). See Table 4.1 for an explanation regarding the height class distribution across the respective sites.

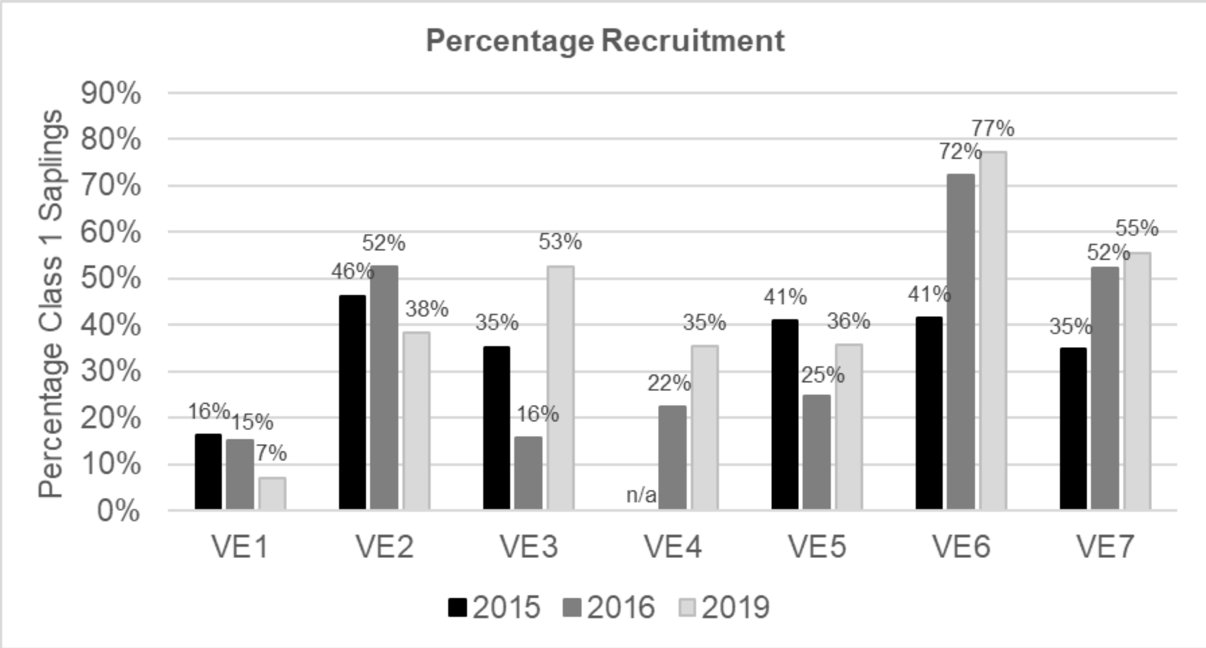

Figure 5-3: **Class 1 saplings as a percentage of the population (saplings + adults) at each site (VE1 – VE7) recorded during the three assessment periods (2015, 2016 and 2019) (see Chapter 3 for site description)**

On average, the population at site VE1 had the least recruitment during the three assessments with a decline in percentage class 1 saplings from 2015 to 2019 (16% of the population to 7% of the population). The population at site VE6 had the highest average percentage recruitment during the three assessment years, with a maximum of 77% of the population recorded as class 1 saplings during 2019. The second highest average recruitment rates were recorded at site VE7, these two sites (VE6 and VE7) were also the only two sites with an increasing percentage recruitment during the three assessment periods. These same trends can be seen when looking at the population density data.

The density of trees per hectare at each site was calculated separately for class 1 saplings and adults (height class 3-6) (Figure 5.4).

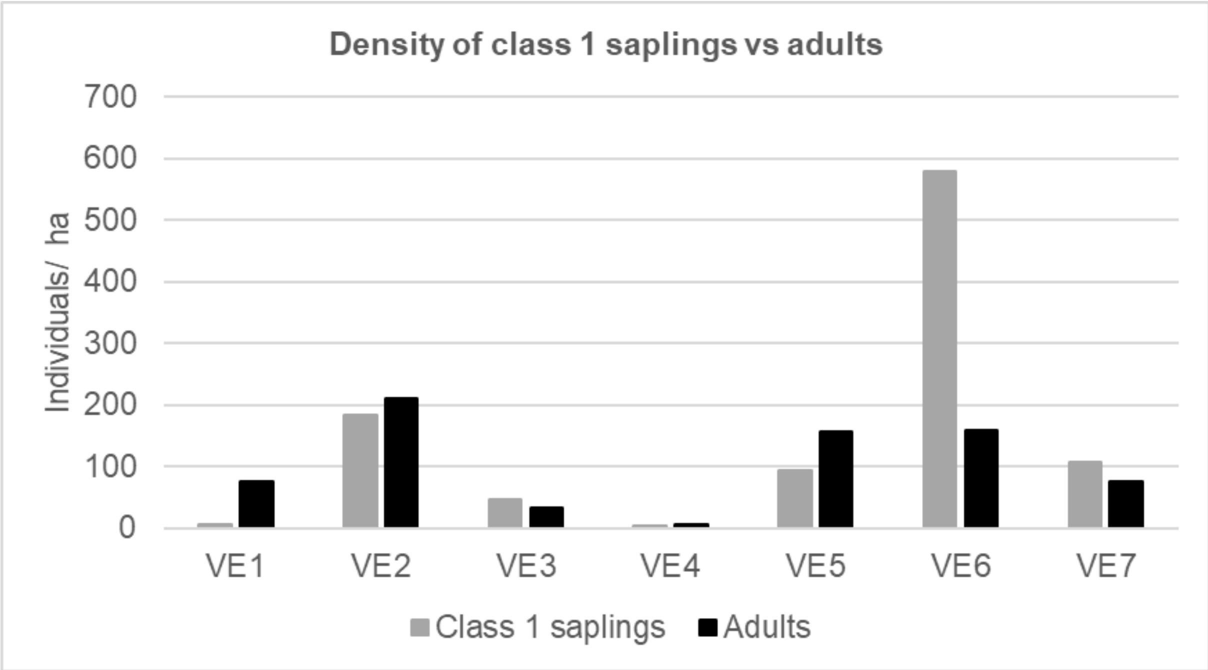

Figure 5-4: **Number of saplings for height class 1 compared to number of adult trees (height class 3-6) per ha for each site (VE1 – VE7) as recorded during the 2019 assessment (see Chapter 3 for site description)**

Even though only two class 1 saplings per hectare were recorded for site VE4, the *V. erioloba* population at his site cannot be described as dense, since only five adult trees per hectare were recorded. Site VE1, however, had a much denser population (75 adults per hectare); yet only six class 1 saplings were found, indicating a low recruitment rate at this site. The class 1 sapling density was higher than the adult density at the populations in sites VE3, VE6 and VE7, indicating that these populations have a much higher recruitment rate. This phenomenon is especially true for the population at site VE6 where the class 1 sapling density was 580/ha and the adult density 158/ha.

As mentioned above, research conducted by Farmer in 1993 (as cited by Van der Merwe, 2001) pointed out that dust pollution deposition will hinder germination of *V. erioloba* seed. During this study the lowest germination rate was however recorded for the population at site VE1, which is not situated within the de-watering or dust ZoI of the Sishen Mine. The populations at sites VE2, VE3 and VE4, all with moderate recruitment rates, are however, situated within the de-watering and dust ZoI.

Seed grazed by herbivores germinate quickly since the digestive system destructs the permeability of the seed coat (Barnes, 2001), thereby increasing the recruitment rate. Apart from a couple of ostriches, no livestock or game were present at site VE1, thus the low recruitment rate reported for this site could be ascribed to a lack of herbivore activity.

Differences in rainfall patterns at the respective sites assessed (VE1-VE7) might also account for variances in their recruitment rates.

However, given that rainfall data was only available for the larger area and not for each individual site, this aspect could not be investigated thoroughly.

Naturally, recruitment rates can also be influenced by a population's production and its available seed bank. The comparison between production (Figure 5.1), predation (Figure 5.2) and recruitment (Figure 5.3) rates of the populations at each site are represented in the figure below (Figure 5.5).

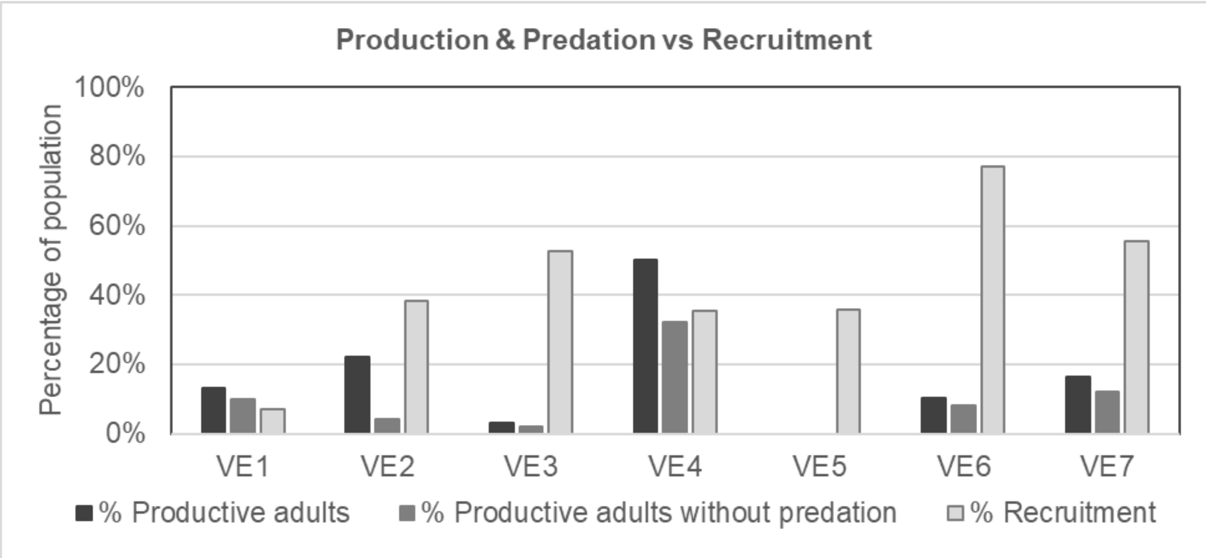

Figure 5-5: Percentage productive living adults recorded for each site (VE1 – VE7) in 2016; percentage productive living adults without bruchid predation present recorded for each site in 2016 and recruitment rate percentages for those sites as assessed in 2019 (see Chapter 3 for site description)

The assumption that populations with more viable pods present will have higher recruitment rates can be assessed qualitatively by comparing the production rates recorded in 2016 with the recruitment rates assessed in 2019 (Figure 5.5). This assumption is not true for the populations at sites VE3, VE5, VE6 and VE7 that had only a few productive adults in 2016 yet high recruitment rates in 2019. This is also not true for the population at site VE4 that, despite its high production in 2016, had less recruitment in 2019.

In this study, it was further assumed that the presence of the bruchid beetle predation will affect the recruitment rate of a population negatively, which could only be assessed qualitatively (Figure 5.5). Populations assessed in 2016 that had a higher percentage of productive adults with no predation present ought to have a higher recruitment rate in 2019. However, the findings for almost all sites contradict this assumption. Within most sites (VE2, VE3, VE5, VE6 and VE7) a very low percentage of productive living adult trees without any bruchid predation was recorded, compared to the percentage recruitment at these sites.

Note that in this study, a significant correlation between production and recruitment or predation and recruitment could not be determined statistically given that the data for recruitment per site only reflected a single figure (number of saplings present per population) and the data recorded for production rates was too limited to reflect the true production rate of each population. Since predation data is dependent on the number of productive adults, such an analysis was also not possible due to data limitations.

5.4 Establishment

Good recruitment does not guarantee the sustainability of a population, since the determining factor is whether the class 1 saplings will establish and grow into a higher height class. As Barnes (2001) pointed out, the rate of establishment of *V. erioloba* saplings depends on above-average rainfall events but, as found by Steenkamp *et al.* (2007), the low survival of saplings can also be ascribed to factors such as plant species competing for soil moisture (especially in arid, sandy soils); the presence of predatory invertebrates, rodents and herbivores; and the availability of suitable microhabitats (so-called “safe sites”) that are conducive to the germination of seeds and the survival of saplings so that they can develop into adult plants.

In this study, the establishment of saplings for each site (VE1 – VE7) was calculated based on the number of height class 2 saplings (< 0.75 m to < 1.5 m) present within each population for the respective assessment years. Height class 2 indicates the number of saplings which survived into the next height class (from class 1). In Figure 5.6 below, class 2 saplings are presented as a percentage of the total population (consisting of all adults and saplings) for the years 2015, 2016 and 2019.

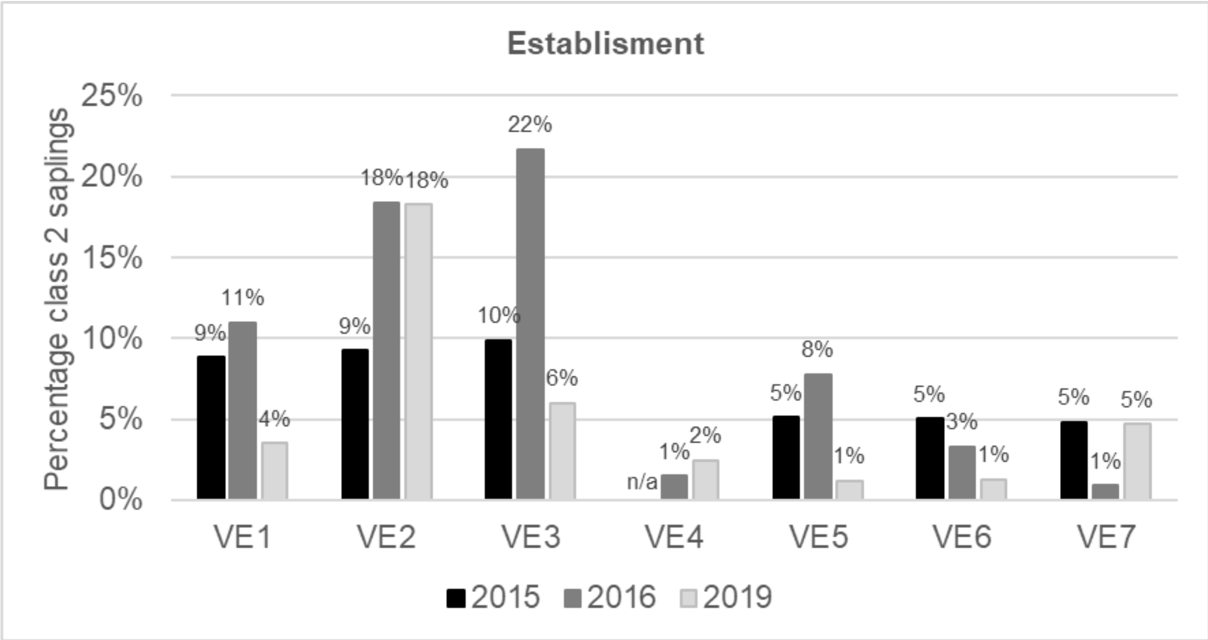

Figure 5-6: **Class 2 saplings (%) of the population (saplings + adults) for each assessment period (2015, 2016 and 2019) (see Chapter 3 for site description)**

The population at site VE2 had the highest average percentage class 2 saplings, followed by the populations at sites VE3 and VE1. The population at site VE2 had average production (Figure 5.1) and average recruitment (Figure 5.3). The population at site VE4 had the lowest average percentage class 2 saplings followed by the populations at sites VE5, VE6 and VE7. The population at site VE4 had the highest percentage productive adults for all sites (Figure 5.1), yet this population had below-average recruitment (Figure 5.3) and the lowest establishment for all sites (Figure 5.6).

For the 2019 assessment, the density of *V. erioloba* populations per site was calculated for the number of class 1 saplings (indicating recruitment), class 2 saplings (indicating establishment) and adults per hectare respectively (Figure 5.7).

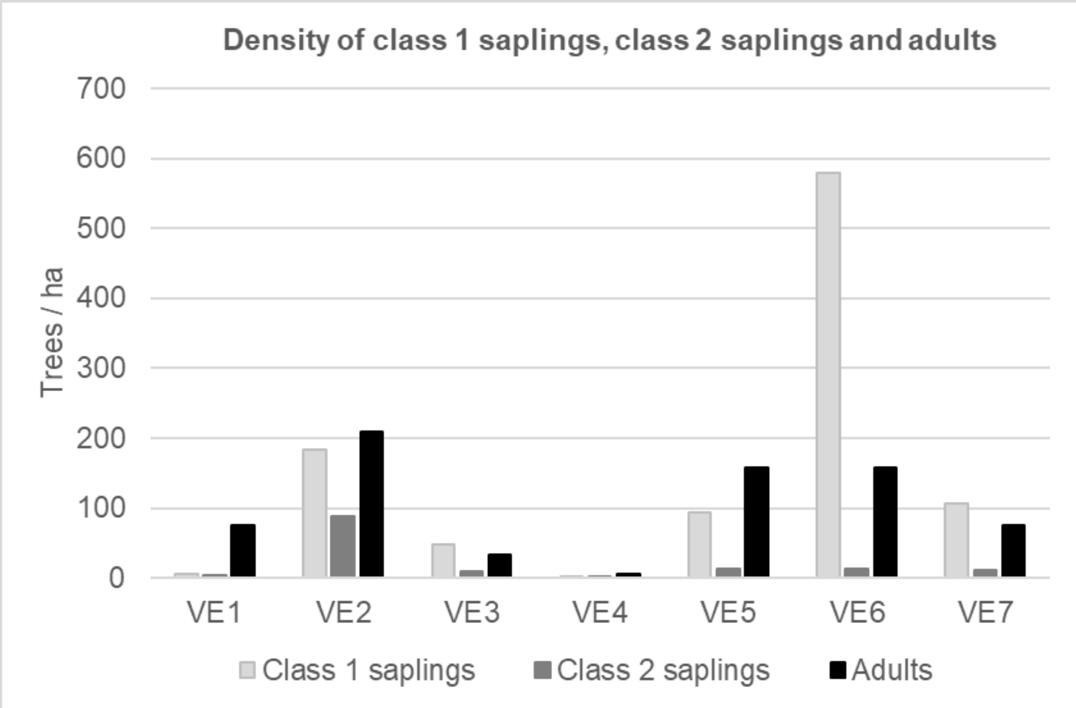

Figure 5-7: Number of class 1 saplings, class 2 saplings and adults (class 3 – 6) correlated with the number of individual trees per hectare for each site (VE1 – VE7) during the 2019 assessment (see Chapter 3 for site description)

The density of class 2 saplings was lower than the number of class 1 sapling density or adults for all sites (Figure 5.7). This is normal since the general survival of *V. erioloba* saplings that eventually reach larger size classes is very low and can be influenced by numerous biotic and abiotic factors as identified by previous research (Van der Merwe *et al*., 2020; Van der Merwe *et al.,* 2019; Steenkamp *et al.,* 2007; Barnes, 2001).

The large difference between class 1 sapling density (580/ha) and class 2 sapling density (12/ha) at site VE6 (Figure 5.7) and the high recruitment rate of class 1 saplings for all three assessment years (Figure 5.3) at this site indicate very low establishment for this population. The reason for this is not known but possible explanations could include natural loss of recruitment, drought, vegetative competition, trampling and grazing by high concentrations of game and or livestock in the area (Jeltsch *et al*., 1999).

5.5 Size class distribution

Different populations of *V. erioloba* and even different individual trees might grow at different rates due to micro-site conditions as well as differences in soil type, moisture availability, grazing pressure, fire events and competition by surrounding vegetation (Stahle *et al*., 1996 as cited by

Van der Merwe *et al*., 2019; Seymour, 2006; Midgley & Bond, 2001; Barnes *et al.,* 1997; Scholes & Archer, 1997; Belsky, 1984).

Nevertheless, a population's size class distribution can still be used as a measure to determine the sustainability of a specific population (Clark, 1991 as cited by Van der Merwe, 2001). In a well-distributed population, all size classes will be represented with most of the population resorting under smaller size classes (Van Rooyen *et al*., 2016; Morkel, 2013).

The size class distribution for each population at all sites was measured in the 2019 assessment. All individuals (living and dead) were considered as presented in the figure below (Figure 5.8). The percentage height class within a population is depicted for each site, where height class 1 and 2 represent saplings and class 3 to 6 represent adult trees (See Table 4.1 for height class descriptions).

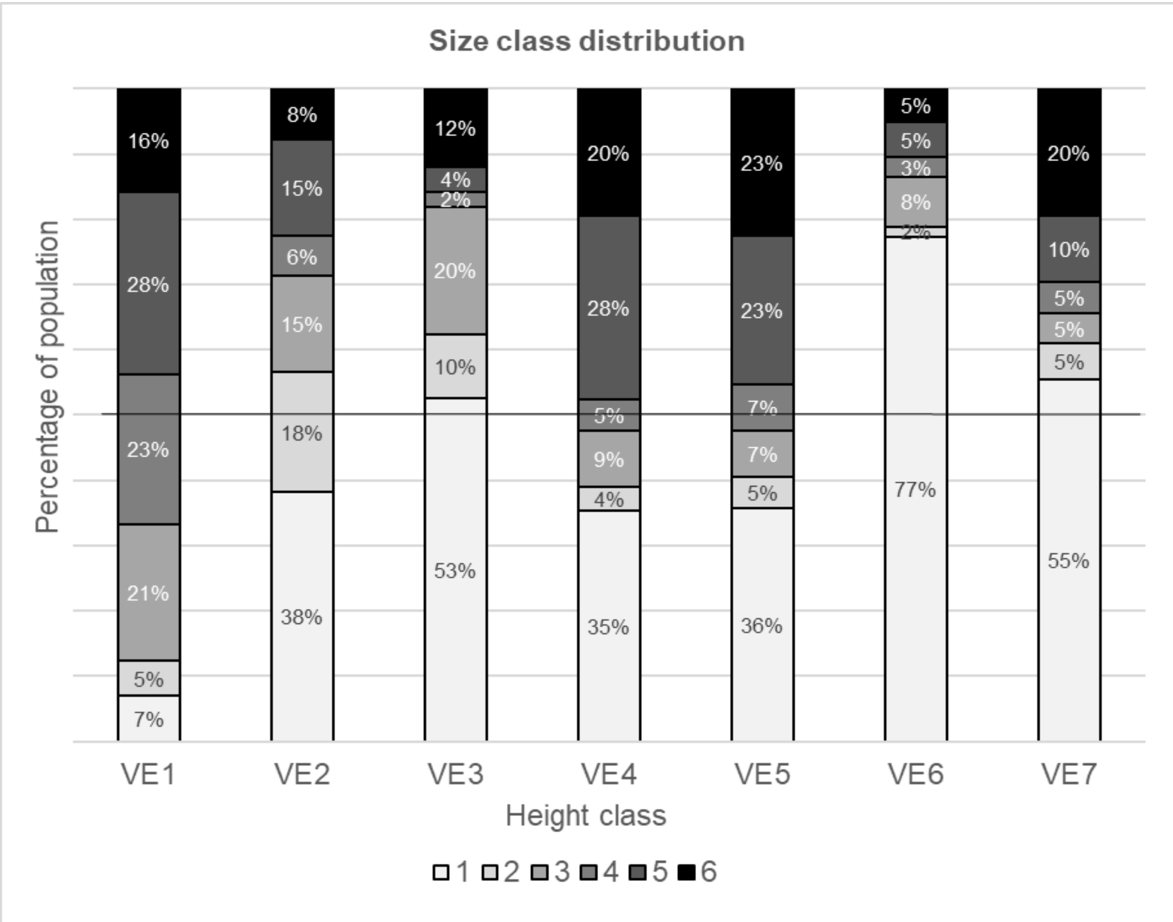

Figure 5-8: Representation (%) of height classes 1 to 6 for sites VE1 – VE7. The horizontal line indicates 50% of the population with saplings resorting under class 1 and 2 and adult trees resorting under classes 3 to 6 (see Chapter 3 for site description)

From the results reflected in Figure 5.8, it is evident that most of the populations are well distributed, where the majority (more than 50%) of individuals at sites VE2, VE3, VE6 and VE7 were saplings (class 1 and 2). At sites VE4 and VE5, saplings represented at least 40% of the population, where saplings only represented 12% of the population at site VE1.

In most populations, all size classes were well represented with at least 5% of each height class present. However, at site VE3, height class 4 only represented 2% of the population and 3% of the population at site VE6, with height class 2 representing only 2% of the population at site VE6.

The size class distribution for only the 50 adult trees within each population are given below (Figure 5.9).

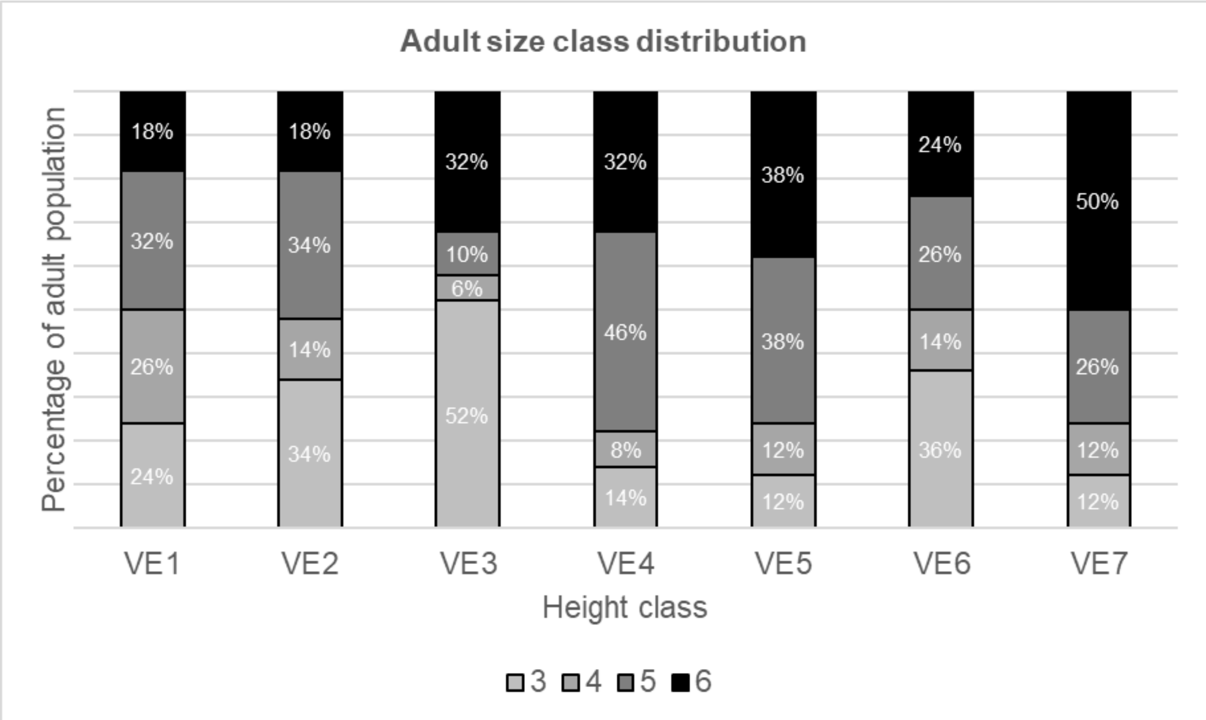

Figure 5-9: Representation (%) of height classes 3 to 6 (adults) for sites VE1 – VE7. (see Chapter 3 for site description)

The highest percentages for individuals in height class 5 and 6 (indicative of a more mature population) were recorded at sites VE4, VE5 and VE7 with 50% of the adult population in the highest height class (class 6) within population VE7. At sites VE1, VE2, VE3 VE6, though, fewer individuals in height classes 5 and 6 were recorded (52% or less for each population), which is indicative of younger, developing populations.

The height of all adult individuals was compared between sites and reflected in the boxplot below (Figure 5.10). A confidence level of 0.95 was used in this Tukey's pairwise comparisons (PennState, 2022) to determine significant differences between sites.

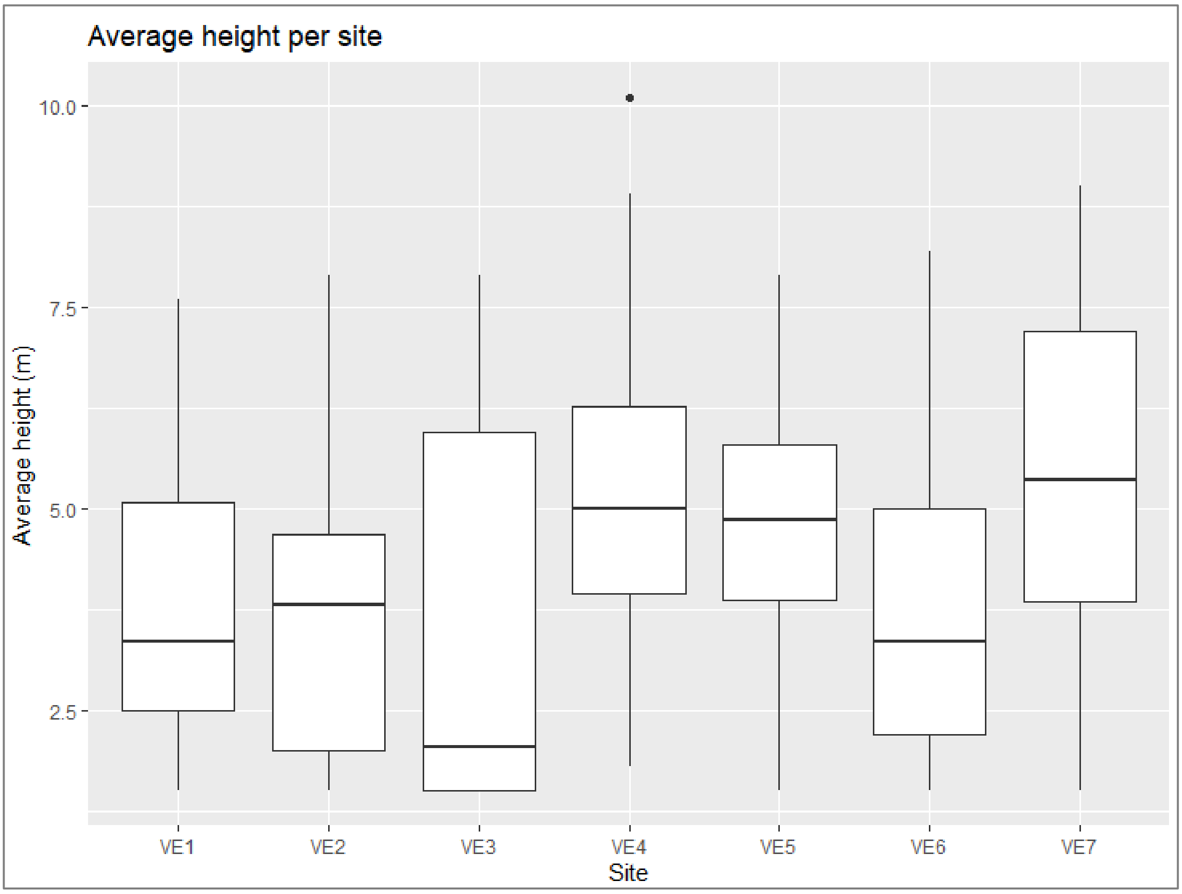

Figure 5-10: Comparison between average height (m) for sites assessed (VE1 – VE7). The boxes represent the interquartile range (IQR) or middle 50% of all data. The horizontal line represents the median height. Whiskers represent 1.5 times the IQR, and outliers are represented as dots separately above or below the whiskers (see Chapter 3 for site description)

Overall, the population at site VE3 had the widest variation in height, also indicative of a young developing population, whereas the population at sites VE4 and VE5 (mature populations) had little variation with the smallest interquartile range.

When comparing the mean height of all sites, the height of trees within the younger populations (i.e. at sites VE1, VE2, VE3 and VE6) were all similar, with the shortest of these being 3.58 m (VE3) and the tallest 3.70 m (VE1 and VE6). The mean height within the populations at sites VE4 and VE7 was significantly higher compared to the rest of the populations, with a mean height of 5.02 and 5.49 m, respectively ($p<0.05$).

The populations at sites VE1, VE2 and VE3 are not normally distributed, whereas the populations at all other sites are normally distributed (Figure 5.10). The only outlier observed was within site VE4 with one height recorded above 10 m, whereas the maximum height was normally indicated as 9 m.

The circumference of all adult individuals was compared between sites and reflected in the boxplot below (Figure 5.11). A confidence level of 0.95 was used in this Tukey's pairwise comparisons (PennState, 2022) to determine significant differences between sites.

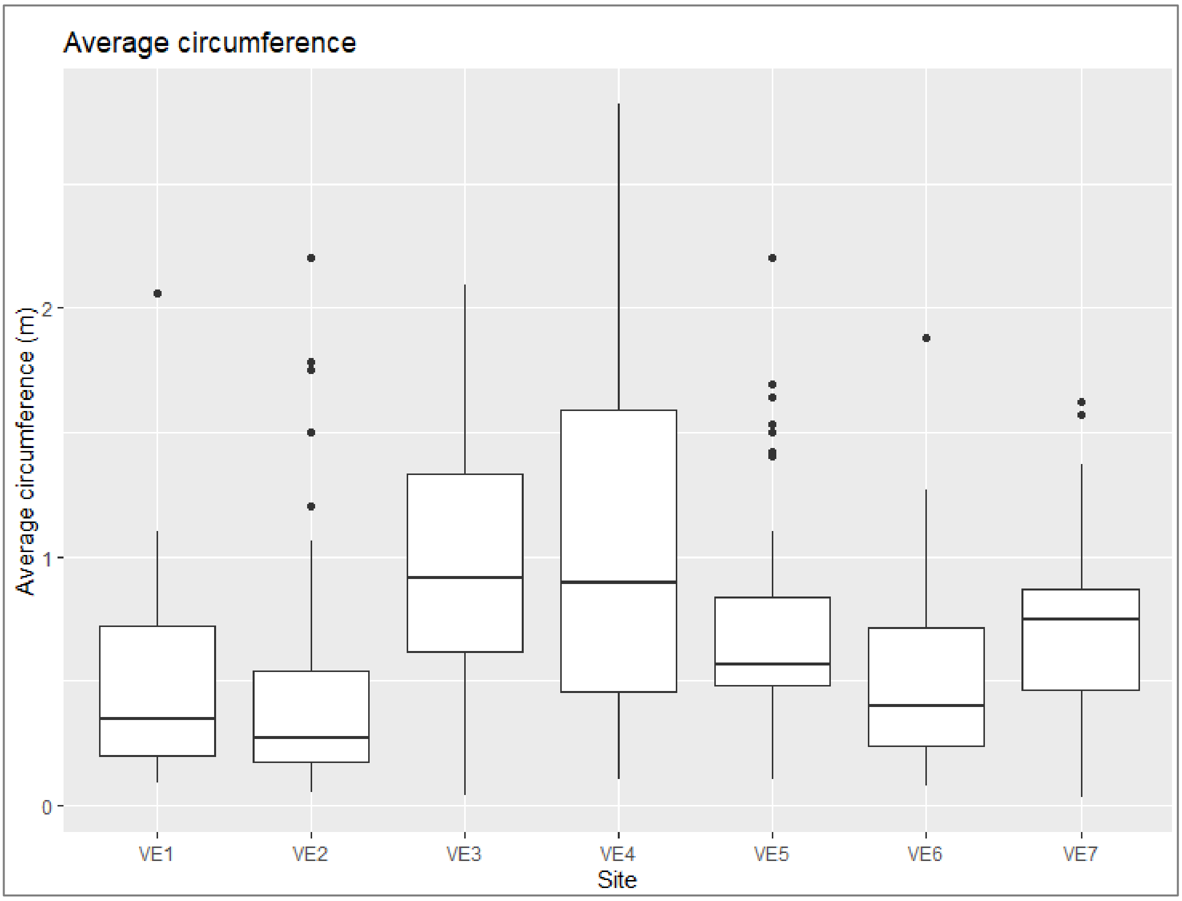

Figure 5-11: **Comparison between average stem circumference (m) of adult trees in the populations of sites assessed (VE1 – VE7). Boxes represent the interquartile range (IQR) or the middle 50% of all data. The horizontal line represents the median circumference. Whiskers represent 1.5 times the IQR, and outliers are represented as dots separately above or below the whiskers (see Chapter 3 for site description)**

Even though all sites differed in terms of distribution of tree stem circumference of the adult population, most sites showed little variation in circumference between individual adult trees within that population. The exception is site VE4, reflecting a large IQR as well as a large range between minimum and maximum circumference (0.1 m – 2.8 m) (Figure 5.11).

All sites have been compared by using Tukey contrast for multiple comparisons between groups. Site VE3 and VE4 showed the greatest similarity in terms of the tree stem circumference of adult populations ($p = 1.0000$), with sites VE1, VE2 and VE6 also displaying a degree of similarity. Populations at sites VE1, VE2 and VE6 had significantly smaller mean circumference than the populations at site VE3 and VE4, all with a significant p value of $<.0001$.

5.6 Canopy die-back

Canopy die-back is more common in older trees but can be an indication of environmental stress when present in younger individuals, especially in saplings (Milton & Dean, 1995 as cited by Barnes *et al.,* 1997).

Various factors, such as predation by *Gonometa postica* caterpillars, the presence of alien invasive species such as *Prosopis glandulosa,* overutilisation of available water sources and a drop in the water table can cause stress in individual adult trees, ultimately increasing canopy die-back (Schachtschneider & February, 2013). Since only a few individual cocoons of *G. postica* caterpillars were observed on some of the trees during the time of assessment, it seems that the predation of these caterpillars did not cause canopy die-back in *V. erioloba* trees. Since no invasive species were found within or near the assessed sites, this phenomenon was not tested. Die-back within saplings could be indicative of various environmental and vegetative factors such as competition between plant species, soil moisture or predation (Steenkamp et al., 2007).

In figures 5.12, 5.13 and 5.14, canopy die-back is indicated per die-back class (i.e. high, moderately high, moderate or low) for all individuals, including saplings and adults respectively (see Table 4.2 for a description of each die-back class).

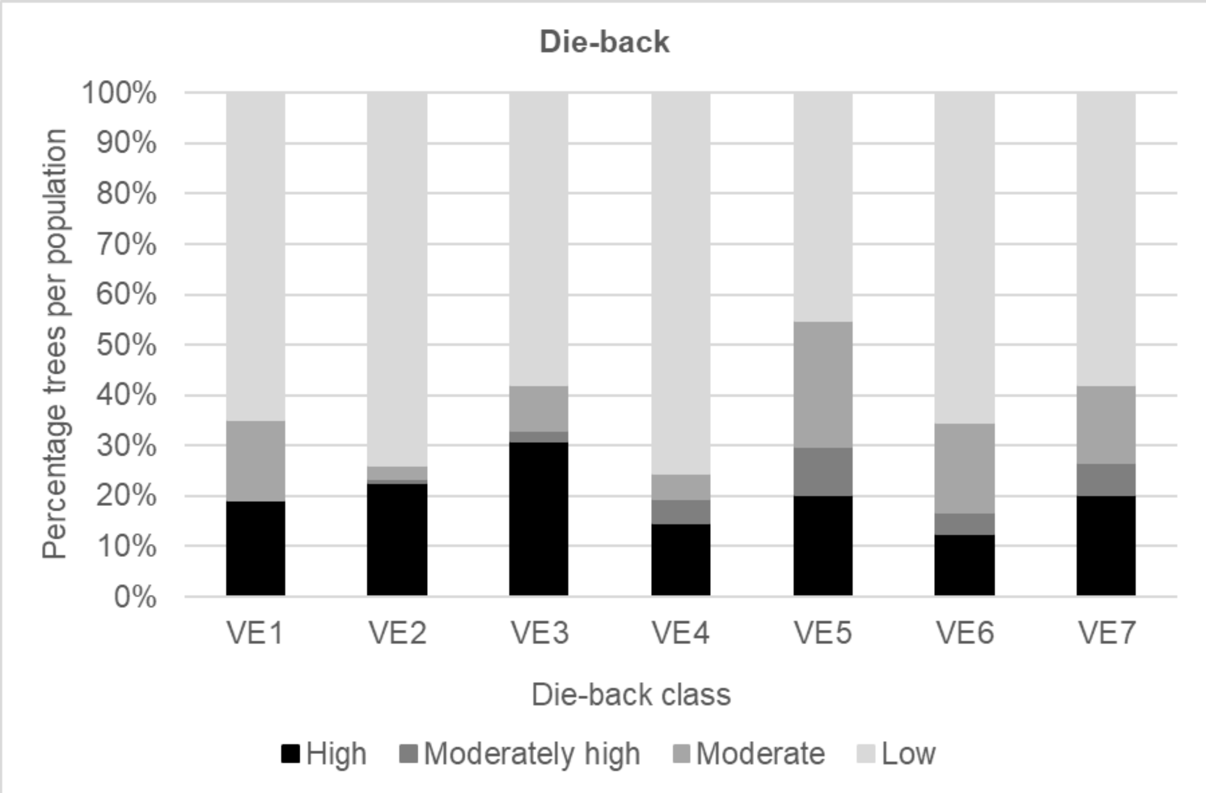

Figure 5-12: Total percentage canopy die-back per die-back class for all individuals (saplings and adults) of each population at study sites VE1 – VE7 (see Chapter 3 for site description)

From Figure 5.12, it is evident that canopy die-back occurred within all populations at several rates. Of all populations assessed, the canopy die-back was the highest at site VE3, with 31% of all trees in this population falling in the highest die-back class (75% -100% canopy die-back). The population at VE6 had the least individual trees falling in the high canopy die-back class (13% of the population). Overall, the least canopy die-back was observed at site VE2 and VE4, with 74% and 76% of the respective populations falling in the lowest canopy die-back class (0%-25% canopy die-back), whereas the population at site VE5 only had 45% of individuals within this class (Figure 5.12).

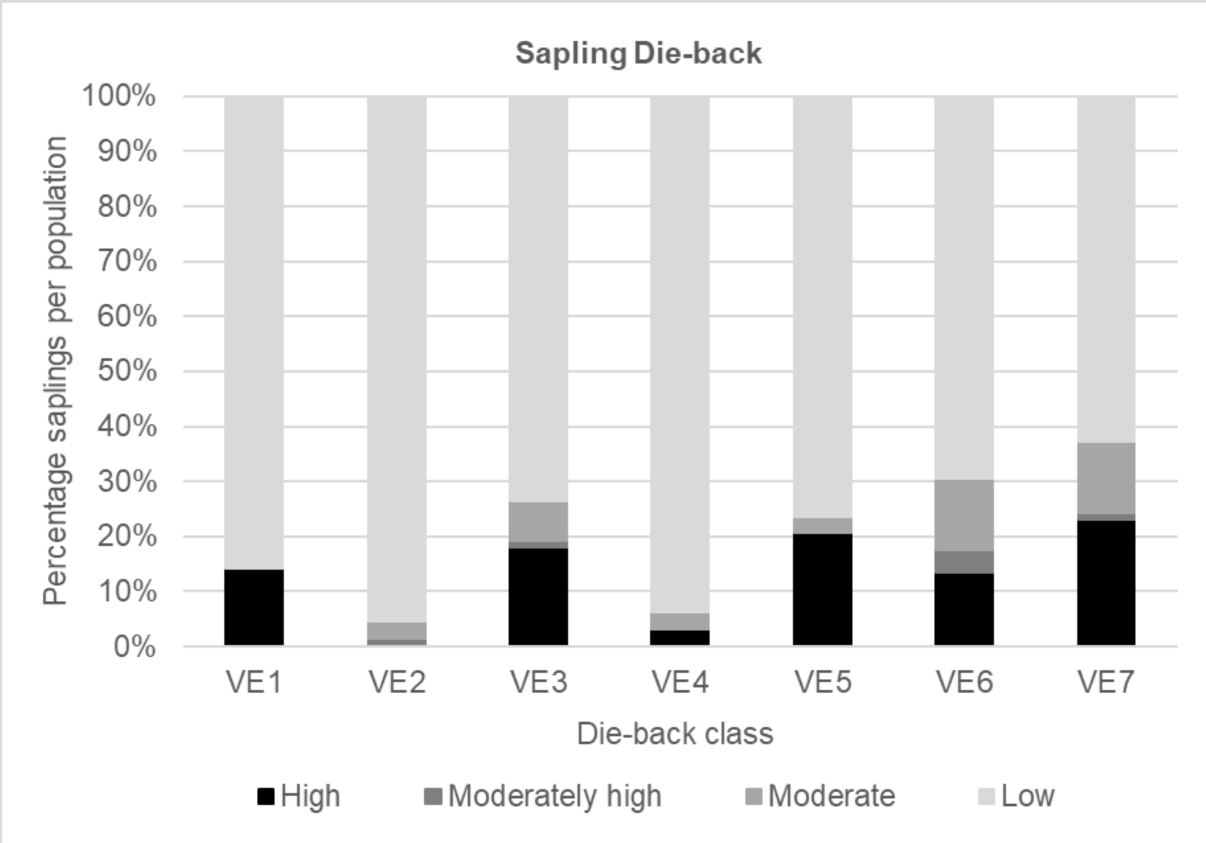

Figure 5-13: Percentage die-back per canopy die-back class for only saplings at study sites VE1 – VE7 (see Chapter 3 for site description)

Die-back was observed in all populations' saplings. The population at site VE2 had the least sapling die-back followed by the population at site VE4, with the remainder of the populations all consisting of at least 10% the recorded saplings within the high die-back class (75%-100% canopy die-back) (Figure 5.13). Possible reasons for the difference in sapling die-back at the different populations could be a difference in rainfall amounts and periods at each site.

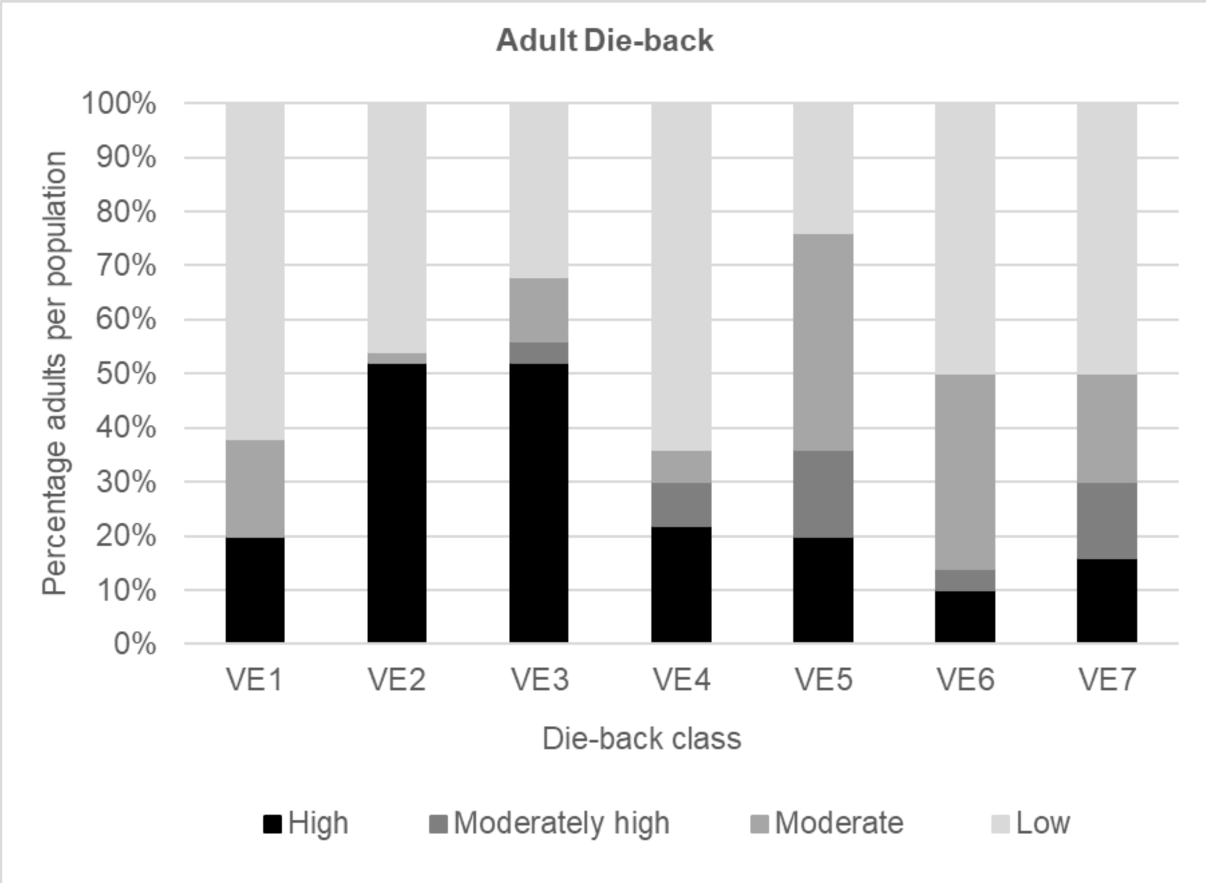

Figure 5-14: Percentage die-back per canopy die-back class for only adult trees at study sites VE1 – VE7 (see Chapter 3 for site description)

Considering the distribution across the respective canopy die-back classes for adult trees only, the populations at sites VE2 and VE3 consisted of the highest percentage of adult trees (52%) in the high die-back class (75%-100% canopy die-back) (Figure 5.14). This high percentage canopy die-back, especially amongst larger individuals, could have been caused by the fire events that happened at these two sites.

The populations at sites VE6 and VE7 were the only ones where a rate of less than 20% adult trees in the high canopy die-back class (75%-100%) was recorded (10% and 16% respectively). The reason for this lower die-back, could be due to the fact that these two sites are located far from any disturbance, no past fires recorded and not located within the Sishen Mine de-watering and dust ZoI (see Chapter 3 for site description).

All assessed sites were compared according to canopy die-back for adult trees within a population, and those findings are reflected in the boxplots to follow in the figures below. The confidence level was set at 0.95, and significant differences between sites were determined with the aid of Tukey's pairwise comparisons.

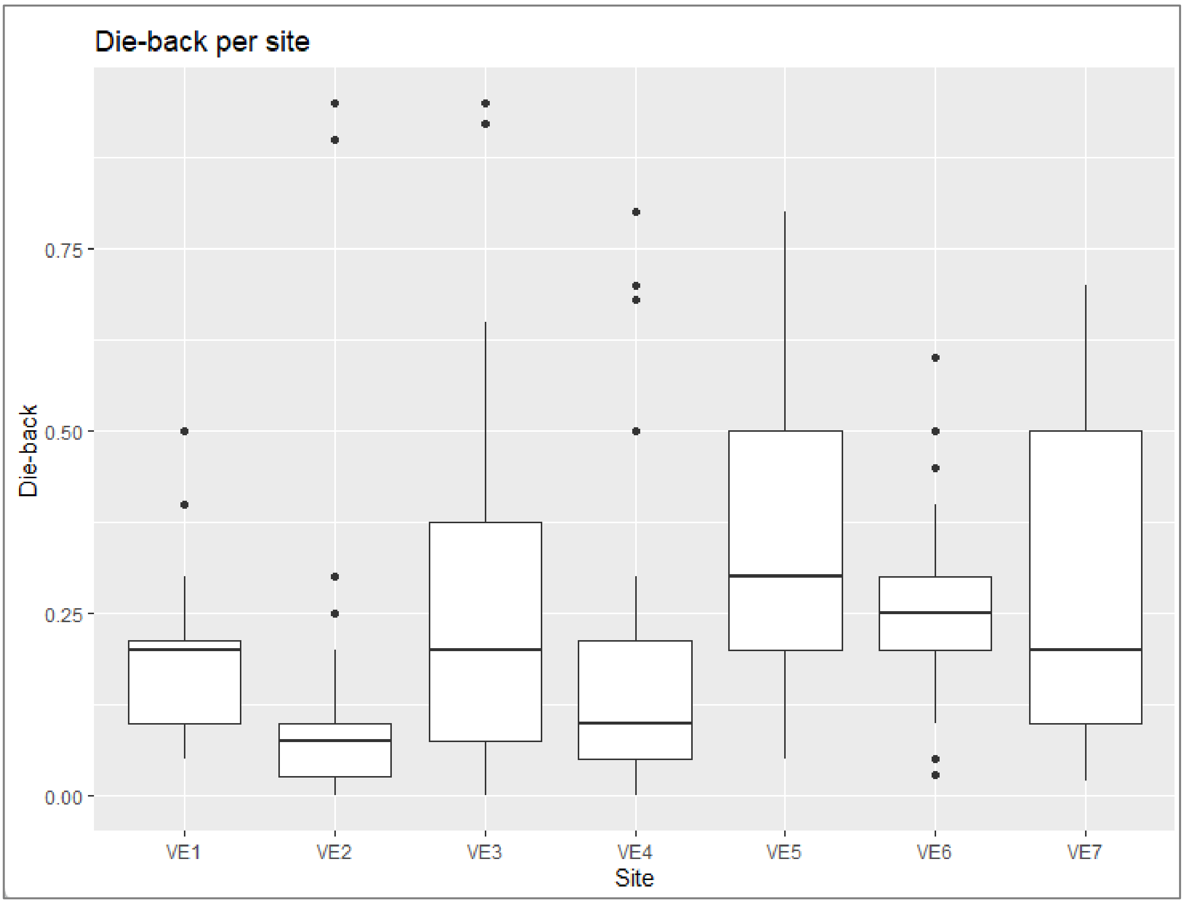

Figure 5-15: Comparison between canopy die-back of adult individuals at sites assessed (VE1 – VE7). Boxes represent the interquartile range (IQR) or the middle 50% of all data. The horizontal line represents the median die-back percentage. Whiskers represent 1.5 times the IQR, and outliers are represented as dots separately above or below the whiskers (see Chapter 3 for site description)

The populations at sites VE3, VE5 and VE7 show a lot of variation in terms of canopy die-back, with much less variation at sites VE1, VE2 and VE6 (Figure 5.15). Only the population at site VE6 is normally distributed regarding canopy die-back. The only site where no outliers were observed is VE7. All other sites had outliers above the 4th quartile, and VE6 had additional outliers below the minimum (Figure 5.15).

All sites have been compared by using Tukey contrast for multiple comparisons between groups. The population at site VE2 had significantly less die-back than the populations at sites VE5, VE6 or VE7 ($p<0.05$).

Similar to the findings of Barnes *et al*. in 1997, damage caused by wood-boring beetles (Buprestids and Cerambycid beetles) was common in the dead branches of *V. erioloba* trees in this study, too. Pairwise comparisons were carried out to establish the effect the presence or absence of small and large wood-boring beetles had on the die-back structure of all assessed populations.

The canopy die-back structure where small wood-boring beetles were present differs significantly from instances where small wood-boring beetles were absent ($p = 0.0016$). The same holds true for instances where large wood-boring beetles were present/absent ($p = <0.0001$). These findings indicate wood-boring beetles (small and large) are more common in tree branches with high die-back, confirming that wood-boring beetles seek out weak or dying parts of individual trees.

5.7 Tree mortality

As pointed out by Zens and Peart (2003), the process of mortality impacts the dynamics of a population. One of the biggest concerns regarding the survival of *V. erioloba* woodlands in the Northern Cape Province is deforestation (Van Rooyen *et al*., 2016; Mans, 2011; Anderson & Anderson, 2007), while fire events, overutilisation and the impact of arboricides are also known to cause mortality in these populations (Van Rooyen *et al*., 2016; Seymour, 2006; Anderson & Anderson, 2001; Liversidge, 2001; Van der Merwe, 2001; Jeltsch *et al*., 1999). In addition, several researchers are of the opinion that factors such as excessive dust pollution, de-watering, overgrazing, trampling, insect damage and invasive species also contribute to mortality in *V. erioloba* woodlands (Barnes *et al.,* 1999 as cited by Morkel, 2013; Schachtschneider & February, 2013; Anderson, 2009; Anderson & Anderson, 2007; Evans *et al*., 2007; Van der Merwe, 2001).

The percentage dead trees (including both saplings and adults) found in the total population during the 2019 assessment is presented in Figure 5.16 below. The mortality of saplings (height classes 1 and 2) and adult trees (height classes 3 to 6) is depicted in Figure 5.17.

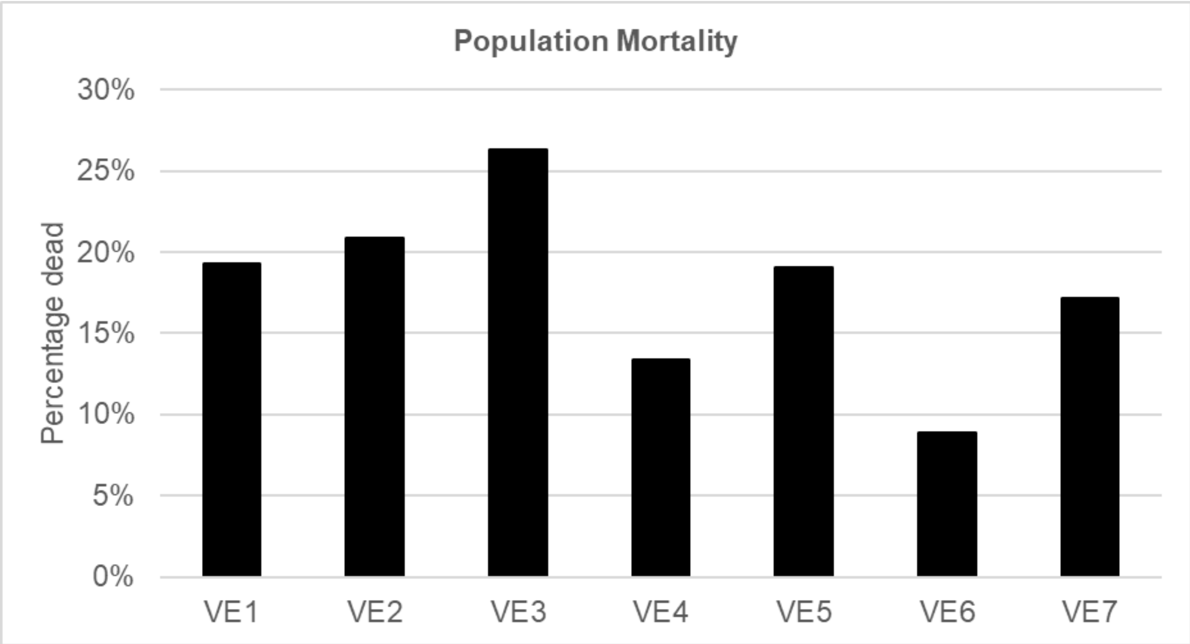

Figure 5-16: **Mortality of total population i.e all *V. erioloba* individuals (saplings + adults) in all populations at sites VE1 – VE7 as assessed in 2019 (see Chapter 3 for site description)**

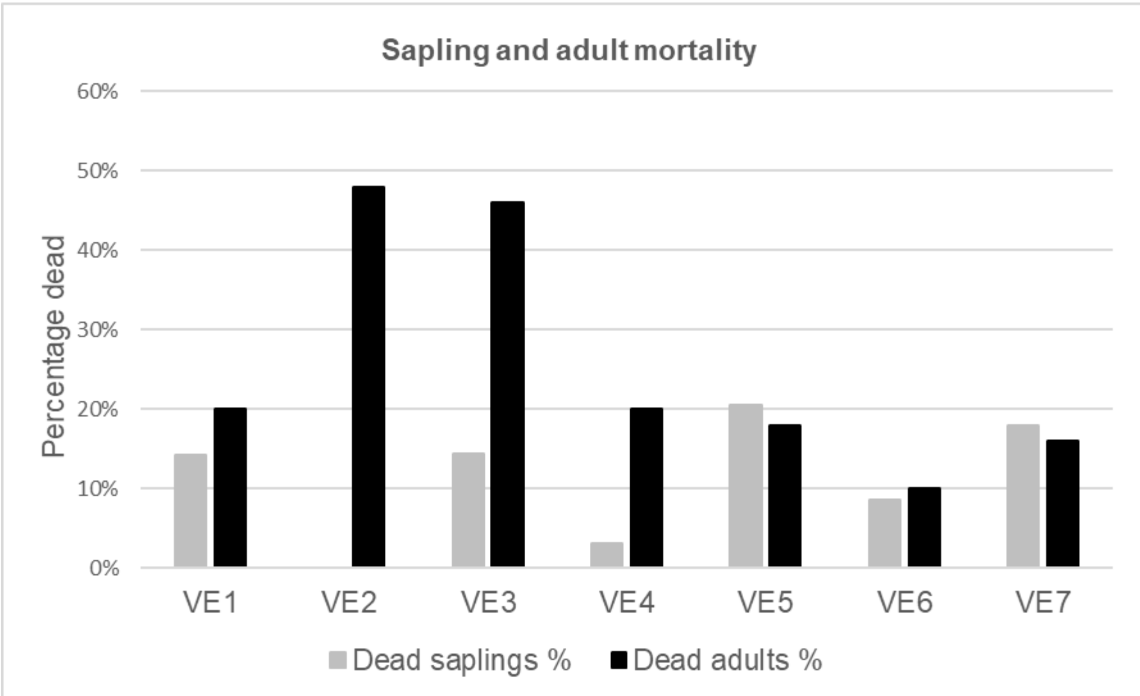

Figure 5-17: **Percentage dead saplings (height class 1 and 2) and percentage dead adults (height class 3-6) for each population at sites VE1 – VE7 as assessed in 2019 (see Chapter 3 for site description)**

The highest percentage of dead individuals were found at study site VE3 (26% of the population), of which the majority were adults (46%) and only 14% saplings. Note that VE3 is located very close to the active mine activity area and railway line, as well as the N14 and a secondary road, which implies that this site falls well within the Sishen Mine's de-watering and dust ZoI. Furthermore, access to this site is not strictly controlled, which could have led to overutilisation with larger trees being cut down and removed for firewood. In fact, 18 of the 23 dead adult trees found at this site were cut down. Note, though, that at the time of the first assessment in 2015, 16 of these were already recorded as dead. Fire damage was evident on all dead and some living individuals, which could have led to the high mortality rate at this site.

Site VE2 (also falling within Sishen Mine's de-watering and dust ZoI) had the highest percentage of dead adults (48%), with the lowest percentage of dead saplings (0%) (Figure 5.17). Besides the possible stressors caused by the relatively close proximity of mining activities (1.7 km) and a railway line (1.1 km), fire damage was also present within this population. It is important to note that although 64% of adults were recorded as dead in 2016, that percentage dropped to 48% in 2019, all of which had fire damage. This indicates that *V. erioloba* trees can resist extensive fire damage, as not all trees found to be without any leaves or new growth are, in fact, dead. This observation is supported by earlier studies conducted by Morkel (2013), Steenkamp *et al*. (2007) and Seymour (2006) who found that most trees, except for large mature individuals and saplings, can resprout after fire damage. If the sustainability of the population at site VE3 is to be ensured, the question arises whether the cutting down of seemingly dead trees that suffered fire damage should be allowed, since some of them might resprout over time. Resprouting of damaged trees should be further investigated over the long-term.

The population at site VE6 had the lowest percentage of deaths for both saplings and adults (9% and 10% respectively) (Figure 5.17). The only known possible stressor at this site is the presence of livestock and game that could have browsed the trees.

All sites have been compared by using Tukey contrast for multiple comparisons between groups. Sites VE2 and VE3 differ significantly in terms of mortality from sites VE6 and VE7 ($p<0.05$). There is also a significant difference between the mortality of populations at sites VE2 and VE5 ($p = 0.0325$).

Populations at sites VE2 and VE3 have both been impacted by past fires, whereas sites VE5, VE6 and VE7 were not impacted in this manner. The latter three sites are situated outside Sishen Mine's de-watering and dust ZoI and are quite a distance from any other developments or mining activities. Of these, site VE7 is situated closest to any mining activity (3.6 km), whereas sites VE5 and VE6 are situated within a declared nature reserve where impacts are minimal and management regulated. All three of these sites (VE5, VE6 & VE7) are also situated close to rivers (0.3 km to 0.7 km).

In this study, neither the effect of arboricides nor the presence of invasive species was assessed given that both stressors seemed to be absent. Likewise, it was not possible to arrive at a conclusion as to the impact overgrazing or trampling has on mortality rates. These aspects will require further investigation.

Even though Barnes *et al.* (1997) found that dead branches of *V. erioloba* trees commonly show signs of damage caused by wood-boring beetles, a thorough examination of the extent of this damage fell outside the purview of this study. Nevertheless, given that wood-boring beetles was a common occurrence at all woodland sites, the correlation between mortality and the presence of small/large wood-boring beetles were tested by way of pairwise comparisons.

The difference in the mortality of individual trees where small or large wood-boring beetles were present or absent is insignificant ($p = 0.5377$; $p = 0.201$). Therefore, the presence of wood-boring beetles cannot be regarded as a major cause of mortality in the assessed woodlands.

CHAPTER 6 CONCLUSION & RECOMMENDATIONS

The main aim of this study was to determine certain characteristics of selected *Vachellia erioloba* woodland populations around Kathu. The study focussed on the production, recruitment, establishment and die-back of *V. erioloba* in the different populations at seven selected sites (i.e. VE1 – VE7). The sites represent different areas at various distances from possible impacts caused by developing agencies. *V. erioloba* is regarded as a keystone species in the area and given that this species is protected, it is important to understand all dynamics that might impact the future management of these woodlands.

A review of the literature revealed that pod production and development can be impacted by various factors such as dust pollution; pod removal for fodder; high browsing pressure; water availability; and climatic factors such as frost, strong winds and hail (Schachtschneider, 2010; Powell, 2005; Seely *et al.*, 1979 as cited by Seymour & Milton, 2003; Van der Merwe, 2001; Nel *et al.,* 1985 as cited by Barnes *et al*., 1997). Based on this study's findings, though, dust pollution seemingly did not have an impact on pod production at the assessed sites given that the population with the highest average pod production assessed was located within the dust zone of influence (ZoI) of Sishen Mine. Pod removal could, however, have had an impact on one of the woodlands where public access is not currently controlled, as pod production was low at these populations, hence the recommendation for stricter access control to this site.

Since this study was only appoint-in-time assessment and not a seasonal production assessment, determining each woodland’s pod production rate proved to be more challenging than anticipated given that the flowering and pod-bearing periods of the individual *V. erioloba* trees varied from one population to the next, as well as between assessment years. The assumption that populations with more viable pods present will have higher recruitment rates was found to be true for most of the populations assessed, but due to limited production data, no significant correlations could be derived.

It was further assumed that Bruchidae beetle predation will affect the recruitment rate of a population negatively. This hypothesis was based on a study conducted by Tybirk *et al*. in 1994 (as cited by Jeltsch *et al*., 1999) in which it was found that predation will deplete the available viable seed bank of a population. Even though bruchid predation was present at all monitored sites, due to the limited pods found on site, true infestation rates could not be determined and statistical correlations between predation and recruitment could not be derived. With the data at hand, this assumption could neither be proved nor disproved since even though the woodland population at site VE1 (where the highest infestation rate was observed) had the lowest recruitment rate, the population at site VE4 (where the lowest infestation rate was observed) conversely had the second lowest recruitment rate.

With reference to site VE1, apart from the high infestation rate of Bruchidae beetles, the lack of herbivory activities at this site could also have accounted for the low recruitment rate. Several earlier studies (Barnes *et al.,* 1997; Hoffman *et al.,* 1989; Coe & Coe, 1987) found that the scarification of the seed coat through the digestive track of animals as well as the resultant increase in moisture can aid the germination process and could thus increase the recruitment rate, hence the recommendation that livestock or game be introduced to this site in keeping with the recommended grazing capacity for the area. Note that exceeding the stocking rate of an area will have the exact opposite effect on the desired outcome since excessive herbivory activity will reduce the available seed bank and, consequently, affect the recruitment rate negatively. Overgrazing and increased trampling was found to further hinder the establishment of these saplings.

Disturbances, whether natural or anthropogenic, not only have an impact on the production and recruitment of a population but also on the health of the population. In their 1995 study, Milton and Dean (as cited by Barnes *et al.,* 1997) postulated that increased die-back can be an indication of the severity of stress on individual trees within a woodland. In the current study, die-back occurred within all populations assessed but differed significantly from one population to the next. Nevertheless, the number of dead trees within each population within each height class, was further investigated, and it was found that populations at sites VE2 and VE3 had significantly higher mortality than the populations at sites VE6 and VE7.

Site VE3 had the highest percentage of individuals within the high die-back class (75%-100% canopy die-back) as well as the highest percentage dead individuals of all populations assessed. Site VE2 had the highest percentage adults within the high die-back class (75%-100% canopy die-back) as well as the highest percentage of dead adult trees. These two sites were the only sites assessed where the impact of past fires was evident, which could have led to the high die-back and deaths observed within the respective populations.

In addition to fire, further impacts on the populations at these two sites (VE2 and VE3) include close proximity to mining activity, railway lines and roads. Nevertheless, site VE4 is also located close to mining activities, site VE4 and VE7 close to railways, and all seven sites are located close to roads or gravel roads. Sites VE2 and VE3 are notably the only two sites with fire impacts. Given that this study found that fire can have a major impact on assessed sites over the short and long term, better management of this aspect is recommended.

Site VE6 and VE7 also showed significant similarities in terms of dead trees. Site VE6 had the lowest percentage of adult deaths recorded, followed by site VE7. The same trend can be seen for die-back, where site VE6 had the lowest number of individuals in the high die-back class (75%-100% canopy die-back), followed by site VE7. These two sites showed few signs of known impacts, are situated far from any disturbances, and have no past fire damage. Wood-boring beetles was found to have no significant impact on the mortality of these trees, although present within all populations.

In all the *V. erioloba* woodlands surrounding Kathu (not just the selected sites), the main impact on these woodlands is anthropogenic. This includes primarily the unsustainable harvesting of wood and pods and the destruction of woodland habitat due to developments (Mans, 2011; Van Rooyen *et al*., 2016). These concerns should be addressed by making the public aware of the importance of these woodlands through education and environmental campaigns. Involving the local community and improving their respect for the environment will not only minimise the impacts on the *V. erioloba* woodlands but on the surrounding natural veld as a whole and will improve the sustainability of the local natural veld.

Some limitations of the study include the following:

- One of the main limitations of this study is the variances between the flowering periods of the *V. erioloba* trees at the respective sites, especially between survey years (i.e. 2015, 2016 and 2019). Trees bear pods and flowers at different times during the productive season and no repeat evaluations of each population were conducted to determine the total production per season.
- Site VE4 was not assessed in 2015; consequently, data for this site is limited to two assessments (i.e. 2016-2019).
- The study was limited to a period of five years. When studying long living trees, such as *V. erioloba*, five years of research only provides short-term information and cannot fully describe the ecological processes within the system (Jeltsch *et al*., 1999). According to O'Connor (1985), monitoring should be conducted for more than 20 years when looking at woody components that are long-lived. Short-term monitoring does not take climatic cycles into account (Jeltsch *et al.,* 1999), nor does it allow enough time to witness the impacts long-term stresses have on trees, such as the lowering of water tables (Meintjies & Van der Westhuizen, 2017).
- Rainfall data was only available for the larger area and no individual data per site was available. This limited information on recruitment and establishment trends, given that the amount of precipitation has a huge impact on these aspects.

- No historic borehole data is available for the selected sites, thus no groundwater variations could be assessed and correlated with the health of the populations.
- Due to data limitations, statistical analysis of the correlation between seed predation and recruitment was not possible.
- No soil analysis was done to determine the content of arboricides used to control the encroachment of surrounding woody species that could possibly have an impact on *V. erioloba.* Soil types could also have differed between the sites which could have had a major impact on the populations.

The following recommendations are proposed for future studies of this kind:

- It is highly recommended that monitoring continues at these study sites over a longer period (at least 20 years) so as to be able to observe long-term impacts such as de-watering. Nevertheless, the data gathered over a period of five years (as presented in this study) can be used to inform future studies aimed at gaining a long-term understanding of these woodlands as an ecosystem and what effects certain impacts will have on these systems.
- The effect de-watering caused by industries and land users have on these woodlands should be studied over longer time periods. Long-term studies are important to determine woodlands' ability to recuperate from water stress (Meintjies & Van der Westhuizen, 2017). It is, therefore, recommended that boreholes be placed strategically near each site to monitor the water table and to derive possible correlations between de-watering and the health of the trees at each specific site.
- Data on rainfall events over the long term should be obtained per site since rainfall events can affect the production recruitment and establishment of a population. This implies that a rain gauge be set up at each site that can be monitored regularly for the duration of future studies.
- Seasonal pod production per site should be monitored through repeat evaluations during the productions season.
- The correlation between seed predation and recruitment should further be investigated.
- The effect of arboricides on *V. erioloba* woodlands should be investigated. It is recommended that future studies include soil sampling and analysis at the different sites, especially in areas where arboricides have been applied.
- The impact surrounding local communities has on the *V. erioloba* woodlands in the Kathu area should be studied in detail, given that most impacts on these woodlands are anthropogenic (e.g. pod collection and wood harvesting; total destruction to make room for developments).

- The impact livestock and game introduced to the sites in keeping with recommended stocking rates should be recorded properly.
- A fire management plan for the area should be in place so as to minimise damages caused to the existing woodlands. As a part of this management plan, long-term monitoring of sites affected by fire is recommended to determine how the fire effects the rate of mortality, recruitment and the resprouting of damaged individuals.

BIBLIOGRAPHY

Acocks, J.P.H. 1953. *Veld types of South Africa*. Pretoria: Government Printer.

AGES (Africa Geo-Environmental Services Gauteng). 2014. *Final scoping report as part of the environmental impact assessment for the proposed new HEF plant on Sishen Iron Ore Mine at Kathu, Northern Cape Province.* https://sahris.sahra.org.za/sites/default/files/additionaldocs/2014-06-25%20Sishen%20HEF_Scoping%20Report%20FINALpdf.pdf Date of access: 05 Mar. 2020.

AGIS (Agricultural Geographic Information System). 1997. *Aridity zones* [Map]. http://daffarcgis.nda.agric.za/arcgis/rest/services/AGIS/Climate/MapServer/generateKml Date of access: 10 Jan. 2022.

AGIS (Agricultural Geographic Information System. 2004. *Generalized soil patterns* [Map]. http://daffarcgis.nda.agric.za/arcgis/rest/services/AGIS/Soil/MapServer/generateKml Date of access: 10 Jan. 2022.

Anderson, M.D. & Anderson, T.A. 2001. Too much, too quickly? Doubts about the sustainability of the camelthorn wood harvest. *African Wildlife*, 55(3):21-26.

Anderson, T.A. 2009. The 'forest' in the Kalahari. *PlantLife*, 37&38:32-37.

Anderson, T.A. & Anderson, M.D. 2007. Will we save the Kathu Forest ecosystem? *African Wildlife*, 61(1):32-33.

Barnes, M.E. 2001. Seed predation, germination and seedling establishment of *Acacia erioloba* in northern Botswana. *Journal of Arid Environments*, 49(3):541-554.

Barnes, R.D., Fagg, C.W. & Milton, S.J. 1997. *Acacia erioloba: monograph and annotated bibliography*. Oxford: Oxford Forestry Institute.

Becking, D. 2020. *Vachellia erioloba*. https://treesa.org/vachellia-erioloba/ Date of access: 1 Jun. 2021.

Belsky, A.J. 1984. Role of small browsing mammals in preventing woodland regeneration in the Serengeti National Park, Tanzania. *African Journal of Ecology*, 22(4):271-279.

Belsky, A.J. & Canham, C.D. 1994. Forest gaps and isolated forest trees: an application of patch dynamics in two ecosystems. *BioScience*, 44(2):77-84.

Cambridge University Press & Assessment, 2022. *Cambridge Advanced Learner's Dictionary & Thesaurus*. https://dictionary.cambridge.org/dictionary/english/deforestation Date of access: 19 Nov. 2022.

Campbell, H., Fellowes, M.D.E. & Cook, J.M. 2013. Arboreal thorn-dwelling ants coexisting on the savannah ant-plant, *Vachellia erioloba*, use domatia morphology to select nest sites. *Insectes Sociaux*, 60:373-382.

Campbell, H., Fellowes, M.D.E. & Cook, J.M. 2015. The curious case of the Camelthorn: competition, coexistence, and nest-site limitation in a multispecies mutualism. *The American Naturalist,* 186(6):172-182.

Coe, M. & Coe, C. 1987. Large herbivores, acacia trees and bruchid beetles. *South African Journal of Science*, 83:624-635.

CSIR (Council for Scientific and Industrial Research). 1988. A forest map of southern Africa with the aid of LANDSAT imagery. *South African National Scientific Programmes Report*, 154.

CSIR (Council for Scientific and Industrial Research). 2002. *Baseline study on woodlands in South Africa.* https://www.environment.gov.za/sites/default/files/reports/baselinestudyonwoodlands.pdf Date of access: 05 Mar. 2020.

CSIR (Council for Scientific and Industrial Research). 2011. *NFEPA rivers* [Vector geospatial dataset]. Biodiversity GIS. http://bgis.sanbi.org Date of access: 31 Aug. 2020.

DAFF (Department of Agriculture, Forestry and Fisheries). s.a. *Policy principles and guidelines for control of development affecting natural forests.* https://www.environment.gov.za/sites/default/files/legislations/policyandguidelines_controlofdevelopmentaffecting_naturalforests.pdf Date of access: 05 Mar. 2020.

DAFF (Department of Agriculture, Forestry and Fisheries). 2015a. Evaluation methodology for *Vachellia erioloba* trees [e-mail correspondence].

DAFF (Department of Agriculture, Forestry and Fisheries). 2015b. *State of the forests report 2010-2012.* Pretoria: Department of Agriculture, Forestry and Fisheries.

Davic, R.D. 2003. Linking keystone species and functional groups: a new operational definition of the keystone species concept. *Conservation Ecology*, 7(1). doi:10.5751/ES-00502-0701r11

Dean, W.R.J., Milton, S.J. & Jeltsch, F. 1998. Large trees, fertile islands, and birds in arid savanna. *Arid Environment*, 41:61-78.

DRDLR (Department of Rural Development and Land Reform). 2016. *SA contours* (5 m) [Vector geospatial dataset]. National geospatial information.

Dreber, N., Van Rooyen, S.E. & Kellner, K. 2019. One savanna, many shapes: How bush control affects the woody layer in the southern Kalahari. *South African Journal of Botany*, 125:511-520.

DWAF (Department of Water Affairs and Forestry). 2007. *Protected trees: Tree species protected in terms of the National Forests Act of 1998 (Act 84 of 1998) of South Africa* [Poster]. https://www.environment.gov.za/sites/default/files/docs/publications/protectedtreesposters.pdf Date of access: 05 Mar. 2020.

DWS (Department of Water and Sanitation). 2011*. Catchments of South Africa* – quaternary [Vector geospatial dataset]. doi: 10.15493/SARVA.DWS.10000003

Eriksson, O. & Ehrlén, J. 1992. Seed and microsite limitation of recruitment in plant populations. *Oecologia,* 91:360-364.

Esterhuyse, N. 2017*. Remarkable trees of South Africa*. 2nd ed. Pretoria: Briza.

Evans, H.F., Moraal, L.G. & Pajares, J.A. 2007. Biology, ecology and economic importance of Buprestidae and Cerambycidae. In: Lieutier. F., Day, K.R., Battisti, A., Grégoire, J.C. & Evans H.F., eds. *Bark and wood boring insects in living trees in Europe, a synthesis*. Dordrecht: Springer. pp. 447-474.

Farmer, A.M. 1993. The effects of dust on vegetation – a review. *Environmental Pollution,* 79(1):63-75.

Fenner, M.W. 1985. *Seed ecology*. Dordrecht: Springer.

Geo Pollution Technologies. 2013. *Hydrocensus Survey and the Interpretation of Data in the Kathu Area*. Unpublished internal report prepared for Khumani Iron Ore Mine. https://pdf4pro.com/cdn/khumani-iron-ore-mine-kathu-gcs-sa-biz-516202.pdf Date of access: 20 November 2022.

Golder Associates. 2014. *Bestwood Estate: Geohydrological Report*. Unpublished internal report prepared for Kathu Property Developers (Pty) Ltd.

https://sahris.sahra.org.za/sites/default/files/additionaldocs/AnnD3_Geohydrological%20report.pdf Date of access: 20 Nov. 2022.

Google Earth Pro V 7.3.4.8642. 2011. *Kathu, South Africa. 27°42'30.61"S, 23°03'53.18"E, Eye alt 7.49 km*. Maxar Technologies 2022. [Computer software]

Google Earth Pro V 7.3.4.8642. 2022. *Kathu, South Africa. 27°42'30.61"S, 23°03'53.18"E, Eye alt 11.33 km*. Maxar Technologies 2022. [Computer software]

Harmse, C.J., Kellner, K. & Dreber, N. 2016. Restoring productive rangelands: A comparative assessment of selective and non-selective chemical bush control in a semi-arid Kalahari savanna. *Journal of Arid Environments*, 135:39-49.

Harper, J.L. 1977. *Population biology of plants*. New York: Academic Press.

Hartland-Rowe, R. 1992. The biology of the wild silkmoth Gonometa rufobrunnea Aurivillius (Lasiocampidae) in northeastern Botswana, with some comments on its potential as a source of wild silk. *Botswana Notes and Records*, 24(1):123-133.

Hoffman, M.T., Cowling, R.M., Douie, C. & Pierce, S.M. 1989. Seed predation and germination of *Acacia erioloba* in the Kuiseb River Valley, Namib Desert. *South African Journal of Botany*, 55(1):103-106.

ISRIC (International Soil Reference and Information Centre). 2003. *SOTER database for southern Africa, SOTERSAF ver. 1.0* [Vector geospatial dataset]. https://data.isric.org/geonetwork/srv/api/records/3571c1f3-159d-442c-b324-0af53d03f12e Date of access: 15 Aug. 2020.

ISRIC (International Soil Reference and Information Centre). 2004. *Major soils of the world* [Map]. https://www.isric.org/sites/default/files/major_soils_of_the_world/set3/ar/arenosol.pdf Date of access: 10 Jan. 2022.

IUCN (International Union for Conservation of Nature). 2021. *The IUCN red list of threatened species* [Version 2021-1]. https://www.iucnredlist.org Date of access: 10 Jul. 2020.

Jeltsch, F., Milton, S.J., Dean, W.R.J. & Van Rooyen, N. 1996. Tree spacing and coexistence in semiarid savannas. *Journal of Ecology*, 84(4):583-595.

Jeltsch, F., Moloney, K. & Milton, J. 1999. Detecting process from snapshot pattern: lessons from tree spacing in the southern Kalahari. *Oikos*, 85(3):451-466.

Jennings, C.M.H. 1974. *The hydrology of Botswana*. Durban: University of KwaZulu-Natal. (Thesis – PhD).

Joubert, D. 2002. *Productivity of camel thorn (Acacia erioloba) and ana tree (Acacia albida) species and the factors that may limit maximum production: an index. Gobabeb*, Namibia: Polytechnic of Namibia.

Kleynhans, C.J. 2000. *Desktop estimates of the ecological importance and sensitivity categories (EISC); default ecological management classes (DEMC), present ecological status categories (PESC), present attainable ecological management classes (present AEMC) and best attainable ecological management class (Best AEMC).* Pretoria: Department of Water Affairs and Forestry.

Li, Y., Hui, G., Zhao, Z., Hu, Y. & Ye, S. 2014. Spatial structural characteristics of three hardwood species in Korean pine broad-leaved forest – validating the bivariate distribution of structural parameters from the point of tree population. *Forest Ecology and Management*, 314:17-25.

Lindenmayer, D.B. & Likens, G.E. 2009. Adaptive monitoring: a new paradigm for long-term research and monitoring. *Trends in Ecology & Evolution*, 24(9):482-486.

Liversidge, R. 2001. A unique habitat threatened: the Kathu camelthorn tree forest in the Northern Cape. *African Wildlife*, 55(3):24-26.

Low, A.B. & Rebelo, T. 1996. *Vegetation of South Africa, Lesotho and Swaziland: a companion to the vegetation map of South Africa, Lesotho and Swaziland*. Pretoria: Department of Environmental Affairs and Tourism.

LTSS (Land Type Survey Staff). 2021. Land types of South Africa: Digital Map (1:250 000 scale) and soil inventory databases. Pretoria: Institute for Soil, Climate and Water, Agricultural Research Council. https://www.arc.agric.za/ /arc-iscw/Pages/ARCISCW-Homepage.aspx

Maclean, G.L. 1967. *A contribution to the biology of the sociable weaver.* Makhanda: Rhodes University. (Thesis – PhD).

Mannheimer, C.A. & Curtis, B.A., eds. 2009. *Le Roux and Müller's field guide to the trees & shrubs of Namibia*. Windhoek: Macmillan Education Namibia.

Mans, J. 2011. Protecting a rare woodland in Kathu, Northern Cape Province. *SA Forestry*. https://saforestryonline.co.za/articles/natural_forests/protecting_a_rare_woodland_in_kathu/ Date of access: 05 Mar. 2020.

Marquart, A., Van Coller, H., Van Staden, N. & Kellner, K. 2023?. Impacts of selective bush control on herbaceous diversity in wildlife and cattle land use areas in a semi-arid Kalahari savanna. *Journal of Arid Environments*, 208 (Provisionally accepted).

Meintjies, M. & Van der Westhuizen, E. 2017. *Determining the presence of Vachellia erioloba trees and the effect ground water abstraction for mining related activities will have on V. erioloba aquifer (ground water) dependant ecosystems, within the Northern Cape Province.* https://sahris.sahra.org.za/sites/default/files/additionaldocs/STS%20SLR%20Vachellia%20erioloba%20aquifer%20dependant%20ecossytems%20Report%20Final%20310817.pdf Date of access: 12 Mar. 2020.

Menges, E.S. 2000. Population viability analyses in plants: challenges and opportunities. *Trends in Ecology & Evolution*, 15(2):51-56.

Midgley, J.J. & Bond, W.J. 2001. A synthesis of the demography of African acacias. *Journal of Tropical Ecology*, 17(6):871-886.

Morkel, M. 2013. *Diversity, structure and dynamics of an Acacia erioloba woodland in the Windhoek area: insights for the management of urban habitats.* Windhoek: University of Namibia. (Dissertation – MSc).

Moustakas, A., Guenther, M., Wiegand, K., Mueller, K., Ward, D., Meyer, K.M. & Jeltsch, F. 2006. Long-term mortality patterns of the deep-rooted *Acacia erioloba*: the middle class shall die! *Journal of Vegetation Science*, 17(4):473-480.

Mucina, L. & Rutherford, M.C. 2006. *The vegetation of South Africa, Lesotho and Swaziland.* Pretoria: South African National Biodiversity Institute.

Murovhi, R.N. 2003. *Concentration of soil nutrients beneath canopies of Acacia erioloba trees in a semi-arid savanna environment of the North-West province*, South Africa. Mahikeng: North-West University. (Dissertation – MSc).

National Forests Act 84 of 1998.

Naudé, P., Leeuw, L., Ncube, P., Setzin, D., Tlou, B., Suping, L. ... Pasha, A. 1987. *Operation camelthorn. Southern African Journal of Environmental Education,* 6:19-20.

Ncedana, C., Pavlic-Zupanc, D. & Roux, J. 2005. *What is killing Acacia erioloba trees in the Kathu region of the Northern Cape Province?* South African Journal of Botany, 98. doi:10.1016/j.sajb.2015.03.174

O'Connor, T.G. 1985. *A synthesis of field experiments concerning the grass layer in the savanna regions of southern Africa*. Pretoria: Foundation for Research Development, Council for Scientific and Industrial Research.

Pakenham, T. 2007. *In search of remarkable trees: on safari in Southern Africa*. Johannesburg/Cape Town: Jonathan Ball.

Palgrave, M.C. 2002. *Keith Coates Palgrave trees of Southern Africa*. 3rd ed. Cape Town: Struik.

Partridge, T.C., Botha, G.A. & Haddon, I.G. 2006. Cenozoic deposits of the interior. In: Johnson, M.R., Anhaeusser, C.R. & Thomas, R.J., eds. *The geology of South Africa.* Pretoria: Council for Geoscience. pp. 585-604.

PennState (Pennsylvania State University). 2022. *Tukey test for pairwise mean comparisons*. https://online.stat.psu.edu/stat502_fa21/lesson/2/2.3 Date of access: 16 Mar. 2022.

Pietersen, K. & Gaffoor, Z. 2018*. Review of mine water reports in the Hotazel, Kathu and Postmasburg areas, Northern Cape*. [Final report for Agri Northern Cape] https://docplayer.net/160629615-Review-of-mine-water-reports-in-the-hotazel-kathu-and-postmasburg-areas-northern-cape.html Date of access: 20 Jun. 2020.

Powell, E. 2005. What is happening to the camel thorns of Kathu? *Veld & Flora*, March:34-35.

Prayag, K.D., Du Toit, C.J., Cramer, M.D. & Thomson, R.L. 2020. Faunal input at host plants: can camel thorn trees use nutrients imported by resident sociable weavers? *Ecology and Evolution,* 10(20):11643-11656.

R Core Team, 2022. R: A language and environment for statistical computing. R Foundation for Statistical Computing. https://www.R-project.org Date of access: 12 Mar 2022.

Raath, M.J., Le Roux, P.C., Veldtman, R., Campbell, H., Buyens, I.P.R. & Greve, M. 2017. Empty *Gonometa postica* cocoons function as nest sites and shelters for arboreal ants. *Journal of Arid Environments*, 144:42-47.

Raliselo, M.A. 2003. *Camelthorn (Acacia erioloba) firewood industry in Western Cape, South Africa.* Stellenbosch: University of Stellenbosch. (Dissertation – MSc).

Richards, J.H. & Caldwell, M.M. 1987. Hydraulic lift: substantial nocturnal water transport between soil layers by *Artemisia tridentata* roots. *Oecologia*, 73:486-489.

Sabiiti, E.N. & Wein, R.W. 1987. Fire and *Acacia* seeds: a hypothesis of colonization success. *Journal of Ecology*, 75(4):937-946.

SANBI (South African National Biodiversity Institute). 2006. *The vegetation map of South Africa, Lesotho and Swaziland,* Mucina, L., Rutherford, M.C. & Powrie, L.W., eds. [Dataset]. Version 2018. Biodiversity GIS. http://bgis.sanbi.org/SpatialDataset/Detail/18 Date of access: 20 Nov. 2019.

Schachtschneider, K. 2010. *Water sourcing by riparian tree species in ephemeral rivers*. Cape Town: University of Cape Town. (Thesis – PhD).

Schachtschneider, K. & February, E.C. 2013. Impact of *Prosopis* invasion on a keystone tree species in the Kalahari Desert. *Plant Ecology*, 214(4):597-605.

Scholes, R.J. & Archer, S.R. 1997. Tree-grass interactions in savannas. *Annual Review of Ecology and Systematics*, 28:517-544.

Seymour, C. 2006. *The influence of size and density of the Camelthorn (Acacia erioloba Meyer) on its keystone role in the Xeric Kalahari*. Cape Town: University of Cape Town. (Thesis – PhD).

Seymour, C. & FitzPatrick, P. 2007. *Slowly does it: Acacia erioloba growing large in Southern Kalahari savannas.* http://the-eis.com/elibrary/sites/default/files/downloads/literature/Slowly%20does%20it_Acacia%20erioloba%20growing%20large%20in%20southern%20Kalahari%20savannas.pdf Date of access: 05 Mar. 2020.

Seymour, C.L. 2008. Grass, rainfall and herbivores as determinants of *Acacia erioloba* (Meyer) recruitment in an African savanna. *Plant Ecology*, 197:131-138.

Seymour, C.L. & Huyser, O. 2008. Fire and the demography of camelthorn (*Acacia erioloba Meyer*) in the southern Kalahari – evidence for a bonfire effect? *African Journal of Ecology*, 46(4):594-601.

Seymour, C.L. & Milton, S.J. 2003. *A collation and overview of research information on Acacia erioloba (Camelthorn) and identification of relevant research gaps to inform protection of the species.* Pretoria: Department of Water Affairs and Forestry.

Shadwell, E. & February, E. 2017. Effects of groundwater abstraction on two keystone tree species in an arid savanna national park. *PeerJ*. doi:10.7717/peerj.2923

Shangoni Management Services (Pty.) Ltd. 2017. *Preliminary closure plan*. Unpublished internal report prepared for Anglo America Kumba Iron Ore. https://minedocs.com/20/Anglo_American_Kumba_Iron_Sishen_Mine_02282017.pdf Date of access: 23 Apr. 2021.

Skarpe, C. 1991. Spatial patterns and dynamics of woody vegetation in an arid savanna. *Journal of Vegetation Science*, 2(4):565-572.

Smith, T.M. & Shackleton, S.E. 1988. The effects of shading on the establishment and growth of *Acacia tortillis* seedlings. *South African Journal of Botany*, 54(4):375-379.

Steenkamp, C.J., Vogel, J.C., Fuls, A., Van Rooyen, N. & Van Rooyen, M.W. 2007. *Age determination of Acacia erioloba trees in the Kalahari. Journal of Arid Environments,* 72(4):302-313.

Tessema, Z., Ejigu, B. & Nigatu, L. 2017. Tree species determine soil seed bank composition and its similarity with understory vegetation in a semi-arid African savanna*. Ecological Processes.* doi:10.1186/s13717-017-0075-7

Thomas, V., Moll, E. & Grant, R. 2009. *Sappi tree spotting, Cape: from coast to Kalahari*. Johannesburg: Jacana Media.

Trager, M.D., Bhotika, S., Hostetler, J.A., Andrade, G.V., Rodriquez-Cabal, M.A., Mckeon, C.S., Osenberg, C.W. & Bolker, B.M. 2010. Benefits for plants in ant-plant protective mutualisms: a meta-analysis. *PLoS ONE*. doi:10.1371/journal.pone.0014308

Traveset, A. 1990. Bruchid egg mortality on *Acacia farnesiana* caused by ants and abiotic factors. *Ecological Entomology*, 15(4):463-467.

Tybirk, K., Schmidt, L.H. & Hauser, T. 1992. Notes on soil seed banks of African acacias. *African Journal of Ecology*, 32:327-330.

UNEP (United Nations Environment Programme). 2002. *Global humidity index* [Geodata]. Environmental data explorer. http://geodata.grid.unep.ch/results.php Date of access: 10 Nov. 2019.

Van der Merwe, H., Van Rooyen, N., Bezuidenhout, H., Bothma, J.du P. & Van Rooyen, M.W. 2019. *Vachellia erioloba* dynamics over 38 years in the Kalahari Gemsbok National Park, South Africa. *Koedoe,* 61(1). doi:10.4102/koedoe.v61i1.1534

Van der Merwe, H., Van Rooyen, N., Bezuidenhout, H., Bothma, J.du P. & Van Rooyen, M.W. 2020. Woody vegetation change over more than 30 years in the interior duneveld of the Kalahari Gemsbok National Park. *Bothalia*, 50(1). doi: 10.38201/btha.abc.v50.i1.2

Van der Merwe, I., Steyn, H.J.H., Hugo, C.J. & Schall, R. 2017. The physical fibre properties of *Gonometa postica* after degumming the cocoons with different methods. *Journal of Consumer Sciences*, 45.

Van der Merwe, K. 2001. *An investigation into the possible causes of decline in the Acacia erioloba population of the Kathu area*. Bloemfontein: University of the Free State. (Dissertation – MSc).

Van der Walt, P.T. & Le Riche, E.A.N. 1984. The influence of veld fire on an *Acacia erioloba* community in the Kalahari Gemsbok National Park. *Koedoe*. doi:10.4102/koedoe.v27i2.571

Van Engelen, V.W.P. & Dijkshoorn, J.A. 2013. *Global and national soils and terrain digital databases (SOTER) procedures manual.* Wageningen, The Netherlands: ISRIC.

Van Rooyen, G., Van Rooyen, N. & Van den Berg, H. 2016. Kathu bushveld study: research offset for first development phase of Adams solar energy facility. Department of Environment and Nature Conservation Northern Cape (DENC) and Department of Agriculture, Forestry and Fisheries (DAFF). (Unpublished internal report).

Veldtman, R., McGeoch, M.A. & Scholtz, C.H. 2002. Variability in cocoon size in southern African wild silk moths: implications for sustainable harvesting. African Entomology, 10(1):127-136.

Venter, F. & Venter, J. 2012. *Making the most of indigenous trees*. Pretoria: Briza.

Werger, M.J.A. & Coetzee, B.J. 1978. The Sudano-Zambezian region. In: Werger, M.J.A., ed. *Biogeography and ecology of Southern Africa.* pp. 301-462. doi: 10.1007/978-94-009-9951-0_10

Woodborne, S. 2004. *A water stress history of an Acacia erioloba (Kameeldoring) tree from the Koichab*. http://the-eis.com/elibrary/sites/default/files/downloads/literature/A%20water%20stress%20history%20of%20an%20Acacia%20erioloba_Koichab_2004.pdf Date of access: 06 Mar. 2020.

Zens, M.S. & Peart, D.R. 2003. Dealing with death data: individual hazards, mortality and bias. *Trends in Ecology & Evolution*, 18(7):366-373.

ANNEXURES

Annexure A: R script for data analysis

```
library(janitor)
library(ggplot2)
library(emmeans)
rm(list=(ls()))
df <- read.csv("C:/Users/ec/Desktop/PDP/M/DATa/elsche.csv",sep=",")
df
str(df)
df<- clean_names(df)
cols <- names(df)[c(-7,-9,-10,-12:-16,-23,-24)]
df[cols] <- lapply(df[cols], factor)
df1 <- df[!df$nr_adults==0,]
df15 <- df1[df1$year==2015,]
df16 <- df1[df1$year==2016,]
df19 <- df1[df1$year==2019,]
plot(sort(df19$dieback))

### DIEBACK 19 ####
dfdb19 <- df19[df19$dead==0,]
ggplot(data=dfdb19, aes(x=plot_number,y=dieback))+
  geom_boxplot()
hist(sqrt(dfdb19$dieback),breaks = 20 )
modeldieback <- lm(data=dfdb19,sqrt(dieback)~plot_number)
plot(modeldieback)
summary(modeldieback)
modeldieback1 <- lm(data=dfdb19,sqrt(dieback)~1)
anova(modeldieback,modeldieback1)
pw <- emmeans(modeldieback, list(pairwise ~ plot_number), adjust = "tukey")
pw
plot(pw, comparisons = TRUE)

### MORTALITY 19 ####
df19test <- df19
```

```
df19test$dead <- sample(0:1, 350, replace=T)
modelmort <- glm(dead~plot_number, family = binomial,data = df19)
plot(modelmort)
summary(modelmort)
pROC::roc(dead~modelmort$fitted.values, data = df19, plot = TRUE, main = "ROC CURVE", col=
"blue")
modelmort1 <- glm(dead~1, family = binomial,data = df19)
anova(modelmort,modelmort1,test = 'Chisq')
anova(modelmort,test = 'Chisq')
MASS::stepAIC(modelmort)
pw <- emmeans(modelmort, list(pairwise ~ plot_number), adjust = "tukey")
pw
plot(pw, comparisons = TRUE)
glm.probs <- predict(modelmort,type = "response")
glm.probs[1:5]

### HEIGHT 19 ####
ggplot(data=df19, aes(x=plot_number,y=height_m))+
  geom_boxplot()
hist((df19$height_m),breaks = 15 )
modelHeight <- lm(data=df19,(height_m)~plot_number)
plot(modelHeight)
summary(modelHeight)
modelHeight1 <- lm(data=df19,(height_m)~1)
anova(modelHeight,modelHeight1)
pw <- emmeans(modelHeight, list(pairwise ~ plot_number), adjust = "tukey")
pw
plot(pw, comparisons = TRUE)

### CIRCUMFERENCE 19 ####
ggplot(data=df19, aes(x=plot_number,y=circumference_combined_trunks_m))+
  geom_boxplot()
hist(sqrt(df19$circumference_combined_trunks_m),breaks = 10 )
modelCirc <- lm(data=df19,sqrt(circumference_combined_trunks_m)~plot_number)
plot(modelCirc)
summary(modelCirc)
```

```
modelCirc1 <- lm(data=df19,sqrt(circumference_combined_trunks_m)~1)
anova(modelCirc,modelCirc1)
pw <- emmeans(modelCirc, list(pairwise ~ plot_number), adjust = "tukey")
pw
plot(pw, comparisons = TRUE)

### SMALL BORER DIEBACK 19 ####
ggplot(data=dfdb19, aes(x=tetradia_lophoptera_sml_borer,y=dieback))+
  geom_boxplot()
hist(sqrt(dfdb19$dieback),breaks = 20 )
modeldieback <- lm(data=dfdb19,sqrt(dieback)~tetradia_lophoptera_sml_borer)
plot(modeldieback)
summary(modeldieback)
modeldieback1 <- lm(data=dfdb19,sqrt(dieback)~1)
anova(modeldieback,modeldieback1)
pw <- emmeans(modeldieback, list(pairwise ~ tetradia_lophoptera_sml_borer), adjust = "tukey")
pw
plot(pw, comparisons = TRUE)

### SMALL BORER MORTALITY 19 ####
df19test <- df19
df19test$dead <- sample(0:1, 350, replace=T)
modelmort <- glm(dead~tetradia_lophoptera_sml_borer, family = binomial,data = df19)
plot(modelmort)
summary(modelmort)
pROC::roc(dead~modelmort$fitted.values, data = df19, plot = TRUE, main = "ROC CURVE", col=
"blue")
modelmort1 <- glm(dead~1, family = binomial,data = df19)
anova(modelmort,modelmort1,test = 'Chisq')
anova(modelmort,test = 'Chisq')
MASS::stepAIC(modelmort)
pw <- emmeans(modelmort, list(pairwise ~ tetradia_lophoptera_sml_borer), adjust = "tukey")
pw
plot(pw, comparisons = TRUE)
glm.probs <- predict(modelmort,type = "response")
glm.probs[1:5]
```

```
### Large BORER DIEBACK 19 ####
ggplot(data=dfdb19, aes(x=macrotoma_palmata_lrg_borer,y=dieback))+
  geom_boxplot()
hist(sqrt(dfdb19$dieback),breaks = 20 )
modeldieback <- lm(data=dfdb19,sqrt(dieback)~macrotoma_palmata_lrg_borer)
plot(modeldieback)
summary(modeldieback)
modeldieback1 <- lm(data=dfdb19,sqrt(dieback)~1)
anova(modeldieback,modeldieback1)
pw <- emmeans(modeldieback, list(pairwise ~ macrotoma_palmata_lrg_borer), adjust = "tukey")
pw
plot(pw, comparisons = TRUE)

### Large BORER MORTALITY 19 ####
df19test <- df19
df19test$dead <- sample(0:1, 350, replace=T)
modelmort <- glm(dead~macrotoma_palmata_lrg_borer, family = binomial,data = df19)
plot(modelmort)
summary(modelmort)
pROC::roc(dead~modelmort$fitted.values, data = df19, plot = TRUE, main = "ROC CURVE", col=
"blue")
modelmort1 <- glm(dead~1, family = binomial,data = df19)
anova(modelmort,modelmort1,test = 'Chisq')
anova(modelmort,test = 'Chisq')
MASS::stepAIC(modelmort)
pw <- emmeans(modelmort, list(pairwise ~ macrotoma_palmata_lrg_borer), adjust = "tukey")
pw
plot(pw, comparisons = TRUE)
glm.probs <- predict(modelmort,type = "response")
glm.probs[1:5]
```