Innovation in Ecological Restoration techniques: Enhancing *Portulacaria afra* survivorship in degraded arid Thicket

Yondela Masande Norman

Submitted in fulfilment of the requirements for the degree of

Magister Scientiae

in the Faculty of Science at
Rhodes University

April 2022

Supervisor: Mike Powell
Co-supervisors: Prof. James Gambiza & Dr. Rebecca Powell

Abstract

The Albany Thicket Biome has undergone extensive degradation over the past century, particularly from overstocking of livestock in the arid types of thicket. The degradation of the biome, coupled with little to no natural recovery, prompted the South African Government to implement the Subtropical Thicket Restoration Programme (STRP) in the early 2000s to rehabilitate degraded thicket using unrooted *Portulacaria afra* truncheons. The STRP also sought to create job opportunities, bring about social upliftment, promote biodiversity conservation and incentivise farmers and landowners in the region to promote carbon trading and farming of *P. afra* using carbon credits.

However, the survivorship of *P. afra* planted using the STRP planting protocol has been less than ~30% because they are planted in degraded areas, where the truncheons are subjected to harsh biophysical conditions such as hard, capped soil, high soil temperatures, drought and herbivory. The aim of this study, therefore, was to improve the survivorship of *P. afra* planted in degraded thicket to at least 30%, using modified planting methods.

In this study, four treatments were applied in 210 pondings (or micro-dams), each with *P. afra* planted in them (the first three treatments used rooted cuttings). The first treatment involved watering regimes where pondings were watered at varying frequencies. The second treatment involved planting companion species alongside *P. afra* while the third treatment involved planting *P. afra* underneath a nurse canopy. In the fourth treatment, unrooted *P. afra* truncheons were planted inside pondings.

The mean survivorship of the *P. afra* cuttings was assessed 18 months after planting. The watering regime treatments, including the control, all yielded a survivorship of over 75%, with the significantly highest levels of survivorship being displayed in the weekly watering treatment (90.9 ± 6.8%) ($p < 0.01$). The companion plant treatment also produced a high mean survivorship of *P. afra* (94.9 ± 3.6%), significantly higher than that of the nurse plants (87.5 ± 6.3%) ($p < 0.01$).

Among the unrooted truncheons it was found that untreated truncheons had the highest survivorship (76.2 ± 17.6%), with the lowest mean being found in truncheons that were both pruned and scarified (70.8 ± 20.8%), suggesting that this treatment, out of all of them, is the least successful under the prevailing environmental conditions in the study area. However, there was no significant differences among the unrooted truncheon treatments and their respective survivorship values to further substantiate this assertion ($p = 0.26$).

Findings in this study also suggest that planting *P. afra* cuttings under a nurse plant was ideal for *P. afra* survivorship due to the nurse plant's ability to ameliorate the microclimate under which the cuttings can establish and grow. Despite the soil under the nurse canopy having a significantly lower mean soil water potential (-160.9 ± 200.5 kPa), compared to the open areas (-73.4 ± 55.7 kPa) ($p = 0.04$), the pondings under the nurse canopy still had a lower mean soil temperature (31.4 ± 5.25°C) than the treatments in the open areas (38.5 ± 2.7°C), during the harsh midday sun. These favourable characteristics are reflected in the *P. afra* cuttings under a nurse canopy having a higher chlorophyll fluorescence (0.76 ± 0.06). compared to those planted in open areas (0.73 ± 0.13), suggesting that the *P. afra* cuttings in the former treatment had ahigher photosynthetic efficiency compared to those in the latter, despite there being no statistically significant difference in chlorophyll fluorescence between the treatments ($p = 0.14$).

Although these findings suggest that applying these modifications to planting *P. afra*, having achieved its objective of increasing survivorship to well over 50%, is likely to produce favourable results not only in overall survivorship, but also in fast-tracking arid thicket rehabilitation and restoration, further research on these rehabilitation techniques and their effectiveness is required. Furthermore, the downside to these treatments is that they are costly and time consuming, which puts the feasibility of large-scale programmes using these applications into question.Further investigation is required to determine ways in which the cost-effectiveness of these applications can be enhanced.

Acknowledgements

I would like to thank-

My supervisor, Mike Powell and my co-supervisors Prof. James Gambiza and Dr. Rebecca Powell for their continued support, guidance and patience through the exciting highs and frustrating lows that make up Masters' research.

Dr. Kathleen Smart, Dr Mahomed Desai and Dr Matthew Mamera for providing valuable guidance and insights on statistical analyses and the ecophysiological aspects of my research.'

Karen Milne, Margaret Wolff and Nolusindiso Kepe, Kathy Cassidy, Monde Ntshudu and everyone in RRRG and the Rhodes Department of Environmental Science for all their help when it came down to planning around the budget and logistics of my research.

The Tchnuganoo Community, for the permission to conduct this study on their farm.

Otto & Maya Beukes, Melloson Allen, Justin Gird, Luyanda Luthuli and all the personnel at Living Lands for providing the study site, the fencing, the spekboom as well as the team that helped get everything started. I also want to thank them all for making my stay in the Baviaanskloof a fond and memorable one.

Yolande Vermaak and the Gamtoos Irrigation Board for providing me with the companion plants used in the study.

Lois Stahl, for accommodating me in the field station, where I spent much of my time in the Baviaanskloof.

Barend Lamprecht, for providing accommodation at the student house on his farm.
The Baviaanskloof community for welcoming me with open arms and for providing help, advice and good company during my years in the valley.

The Department of Forestry, Fisheries and the Environment (DFFE) (Chief Directorate Natural Resource Management) and the Global Environment Facility (GEF) for providing the funding to make this research happen.

My friends and peers for providing perspective, advice, support and generally good banter.

My family, in particular my mother Judge Thandi Victoria Norman, for always encouraging me, pushing me and speaking life into my passions.

Last but not least, the Most High and my ancestors for continuing to bless, guide and protect me through this journey called Life.

Ndiyabulela

Table of Contents

List of Figures

List of Tables

List of Plates

List of Acronyms

AENP	Addo Elephant National Park
AFR100	African Forest Landscape Restoration Initiative
thicket	Albany Thicket Biome
BMR	Baviaanskloof Mega-Reserve
BNR	Baviaanskloof Nature Reserve
CBD	Convention on Biological Diversity
CNP	Camdeboo National Park
DFFE	Department of Forestry, Fisheries and the Environment
ESF	Ecosystem Structure and Function
EPWP	Expanded Public Works Programme
Fv/0	Initial Fluorescence
Fv/Fm	Maximum Fluorescence
GEF	Global Ecosystem Facility
GFRNR	Great Fish River Nature Reserve
IPCC	Intergovernmental Panel on Climate Change
LDN	Land Degradation Neutrality
LSF	Linear Structure and Function
PAR	Photosynthetically Active Radiation
SANBI	South African National Biodiversity Institute
SANParks	South African National Parks Board
SBEM	Soil Bioengineering Measures
SER	Society for Ecological Restoration
SLM	Sustainable Land Management
SDG	Sustainable Development Goals
STEP	Subtropical Thicket Ecosystem Planning
STM	State and Transition Model
STRP	Subtropical Thicket Restoration Programme
SWC	Soil and Water Conservation
UNCCD	United Nations Convention to Combat Degradation

VEGMAP	National Vegetation Map Project
WfE	Working for Ecosystems
WfW	Working for Water
WOCAT	World Overview of Conservation Approaches and Technologies

Chapter 1. GENERAL INTRODUCTION

1.1. Overview of land degradation

Land degradation, a pervasive and systemic phenomenon that can occur anywhere in the terrestrial world, is the reduction in biological or economic productivity in a terrestrial ecosystem due to anthropogenic disturbance (Ma & Zhao, 1994; Scholes *et al*. 2018). It is characterized by the decline in ecosystem structure and function. Ecosystem structure refers to the biotic (vegetation types and species) and abiotic (soil resources and nutrients) components of a system. Ecosystem function on the other hand, refers to the processes which retain ecosystem structure such as abiotic nutrient and water cycling; biotic growth, establishment and mortality of species (Turnbull *et al.* 2008). According to Cortina *et al.,* (2006), degradation occurs along an ecosystem structure and function gradient whereby a loss in structure leads to a loss in function and *vice versa* (Cortina *et al*. 2006).

According to the UNCCD (2016), approximately 25%, 36% and 10% of land globally is severely degraded, moderately degraded and improving, respectively. This equates to an annual degradation rate of 12 million ha, or an annual economic lossof US$ 490 billion (up to 3% of the global agriculture gross domestic product) (UNCCD, 2016).

Dryland degradation is usually caused by overstocking of livestock which graze or browse the vegetation in those drylands, thereby causing negative consequences on biodiversity and the livelihood of millions of people (Hanke *et al.* 2011). Browsing refers to eating woody and non-woody dicotyledonous plants and grazing refers to eating grass (Gordon & Prins, 2008). Dryland degradation is often marked by several factors or syndromes. These include perennial plant cover loss, bush and shrub encroachment, alien plant invasion, a reduction in palatable plant species, an increase in amount, size and connectivity of bare ground patches and ultimately increased soil crusting, compaction and erosion as well as decreased water infiltration (and thus a reduced water table) and increased surface run-off (Mills & Fey, 2004; Scholes, 2009).

Land degradation in South Africa is caused by several drivers, namely biophysical, socio-economic and socio-political (Hoffman & Todd, 2000). Regarding the

biophysical factors, much of rural South Africa's communally owned land is situated in areas of higher rainfall and steeper slopes. The land is therefore more susceptible to land degradation from high densities of both people and livestock; a consequence of poverty as well as colonial and apartheid socio-political planning (Meadows & Hoffman, 2003).

The key socio-economic drivers include food security and unsustainable profitability issues which have driven private farmers to focus more on intensive farming to maximize income (commercial and subsistence farming). These issues often result in unsustainable agricultural practices such as overstocking, which results in overgrazing (Du Plessis, 2016) and overbrowsing (excessive defoliation of trees/shrubs) (Trollope *et al*. 1990). Other socio-economic forces include urbanization, which results in pristine natural areas being cleared and converted for urban use (Parnell & Walawege, 2011), as well as the rising demands for food and declining profits from agriculture across the country. These factors vary spatially, and the severity of the impacts thereof vary between different vegetation types. The xeric vegetation types experienced more overgrazing/overbrowsing-induced land degradation compared to the coastal types, where impacts from urban and commercial agriculture development were more prevalent (Lloyd *et al*. 2002).

The socio-political drivers of land degradation in South Africa are evident in the land dispossession policies of the previous apartheid regime. The oppressive laws saw black Africans being forcibly removed from their land and relocated to homelands or Bantustans. This meant that relatively small expanses of land were supporting high densities of people and livestock and since these pressures were unsustainable, land degradation followed soon after (Shackleton & Gambiza, 2008).

The state of an ecosystem depends on the state of the biophysical domains that make up a landscape (Stringham *et al*. 2003:7). Human-induced alterations of said domains can sometimes cause ecosystems to change from one state to another due to loss in ecosystem integrity (Stringham *et al.* 2003). The resilience of an ecosystem, therefore, is the magnitude of disturbance an ecological state can resist before being changed into another state that is maintained by a different set of structural and functional processes (Peterson *et al*. 1998). Suding *et al*., (2004), however refer to the definition as one of ecological resistance. Suding *et al*., (2004) further state that

resilience is the rate at which a system can recover to its original state after having experienced a major perturbation.

With resilience to disturbances there also exists a point at which a system can no longer withstand said disturbances and thus switches into another state. This is where degradation occurs (Scholes, 2009).

State-and-transition models

The causal and resultant processes of land degradation are often best illustrated using what is known as a state-and-transition model (Figure 1.1; Figure 1.2). A state- and -transition model is a model used to describe rangeland dynamics in a set of states and the transitions between said states (Westoby *et al.* 1989). The "states" refer to the states of vegetation currently existing on a landscape and the "transitions" refer to the changes in said vegetation which then leads to a change from one state to another. These transitions are usually facilitated by natural events (climate change, fire, weather events) and management actions that include changes in stocking rate, clearing of existing vegetation and the introduction of new plant species (Bestelmeyer *et al*. 2009).

The application of this model is aimed at the sustainable management of semi-arid drylands through the development and applications of management strategies. These strategies help identify constraints and opportunities in taking proactive measures that will promote desirable vegetation states and avoid the risk of degradation (Bestelmeyer *et al*. 2010). It shows that multiple vegetation states can occur within a single rangeland, whether naturally or anthropogenically. Therefore, the state-and -transition model is structured to monitor transitions from grassland to shrubland or changes in plant communities that can result from changes in management or environmental conditions (Ash *et al*. 1994; Rietkerk & van de Koppel, 1997; Stringham *et al*. 2003; Suding *et al*. 2004; Quétier *et al*. 2007).

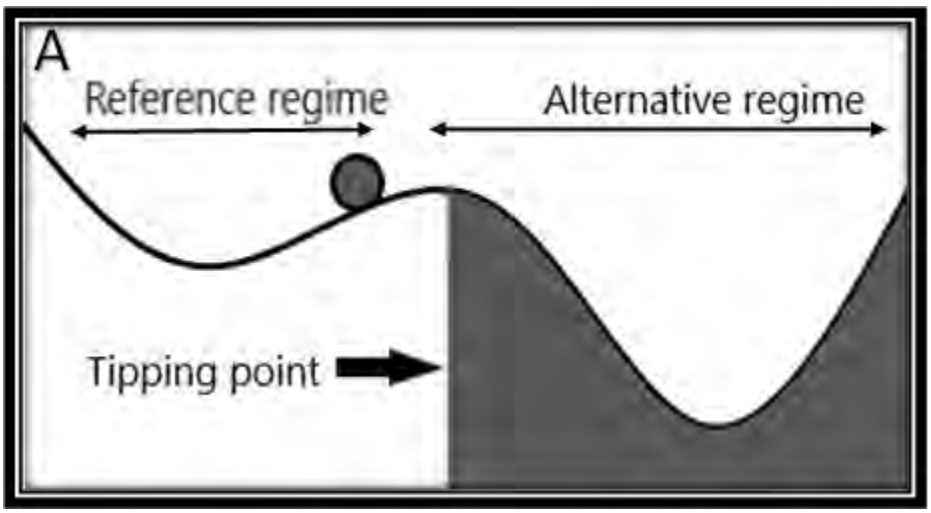

Figure 1.1. State-and-transition model between two states separated by an ecological threshold (source: Bestelmeyer, 2015).

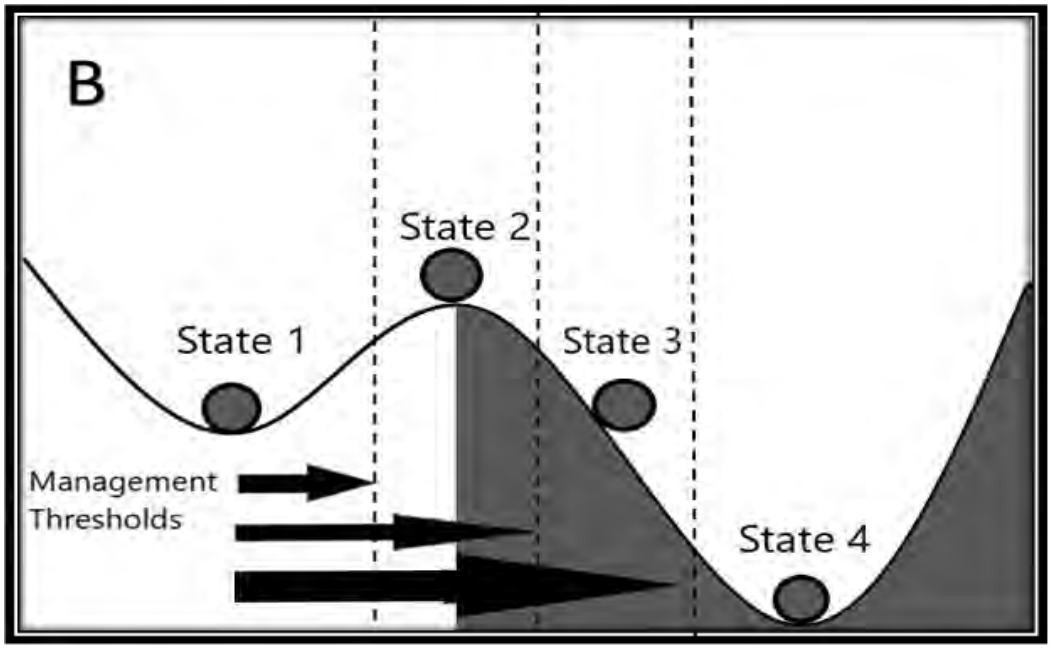

Figure 1.2. Multiple states existing in response to management of reversing/preventing a shift from one state to another in relation to multiple ecological thresholds (source: Bestelmeyer, 2015).

The ball in both Figure 1.1 and Figure 1.2 symbolizes the ecosystem and its movement. When the ball is at the reference state (Figure 1.1) or Stage 2 (Figure 1.2), it can still be reversed to its pre-disturbance condition. However, if the ball is at the

bottom of the alternative state (Figure 1.1) or Stage 4 (Figure 1.2), then returning the system back to its previous ecological state becomes much more difficult and degradation occurs (Bestelmeyer, 2015).

To illustrate this, Bestelmeyer (2015) used the example of juniper (*Juniperus ashei*) encroachment on grassland, whereby the juniper encroachment represents the tipping point between two states. In this example, the grassland is the reference regime that represents a historical or pristine state (Bestelmeyer *et al*. 2009). In this regime which juniper recruitment and encroachment are limited by the existing fire regime. Natural or anthropogenic disturbance would trigger a shift, via the tipping point (i.e., juniper recruitment), from the reference state to an alternative stable state, which often persists under characteristics different to those of the reference state (Beisner *et al*. 2003). In the alternative stable state, fuel loads are reduced, thus increasing juniper recruitment and eventually encroachment (Figure 1.1). On the other hand, these states can be defined according to the management practices required to prevent or reverse these transitions over four states (Figure 1.2).

This example of a state-and -transition model is like the linear structure and function model whereby degradation and recovery paths follow a similar trajectory (Figure 1.1; Figure 1.2). However, this is not that simple in field studies (Maestre & Cortina, 2004) because the steps along this trajectory vary in magnitude and some can hardly be reversed without external inputs (Whisenant, 1999). Furthermore, a linear trajectory model can lead to very narrow definitions of pristine ecosystems and the inputs required to restore degraded landscapes can be mistaken (Cortina *et al*. 2006). Regarding this, Cortina *et al.,* (2006) therefore recommended that the linear structure and function model be integrated with the state-and-transition model. This way, there is greater flexibility in identifying reference ecosystems (the definitions of which depend on the target ranges of composition and function) and appropriate measures for managing landscapes can be better formulated (Cortina *et al*. 2006).

1.2. Land rehabilitation and restoration

Since approximately 25% and 36% of land globally is severely degraded and moderately degraded respectively (UNCCD, 2016), this translates to 25% and 36% of global land at Stages 3 and 4 respectively according to the state-and-transition model (Bestelmeyer, 2015), having crossed their ecological thresholds as a response to land management actions. However, because 10% of land globally is improving (UNCCD, 2016), this means that the trajectory of degradation has been reversed in those areas, thereby bringing those landscapes closer to their reference states. This shift from a degraded landscape to one that is as close as possible to its reference requires that degradation and the causes thereof in that landscape need to be addressed.

The past decade has seen an increase in the awareness and promotion of the sustainable land management concept as an appropriate means of addressing land degradation (Schwilch *et al*., 2014). As defined by the World Overview of Conservation Approaches and Technologies (WOCAT), sustainable land management is 'the use of land resources, including soils, water, animals and plants, for the production of goods to meet changing human needs, while simultaneously ensuring the long-term productive potential of these resources and ensuring their environmental functions' (WOCAT, 2007).

In further recognition of land degradation and the importance of sustainable land management, the UNCCD (UNCCD, 2016) conceptualized the Sustainable Development Goals (SDG) with SDG 15.3 being "Life on Land". This goal states that terrestrial ecosystems should be protected and sustainably managed while degraded ecosystems are restored to achieve land degradation neutrality (previously known as zero-net degradation) (UNCCD, 2016). Land degradation neutrality uses the following land-based indicators to measure progress: trends in land cover, trends in land productivity and functioning and trends in above and below ground carbon stock (Davies *et al*. 2015).

To add further support to sustainable land management, parties to the Convention of Biological Diversity (CBD) convened in Aichi-Nagoya, Japan, adopted the Strategic Plan for Biodiversity 2011-2020, otherwise known as the Aichi Targets (CBD, 2010).

These targets include the following (in no particular order) (CBD, 2010; Hill *et al*. 2015):

1. Increased public awareness of biodiversity values
2. Restoring and protecting ecosystem goods and services
3. Reducing the rate of biodiversity loss by at least 50%
4. Sustainable agriculture, aquaculture and forestry
5. Increasing ecosystem resilience and contribution to carbon sequestration

To combat land degradation and the consequences thereof on nature and society, land rehabilitation and restoration measures are required to be implemented. These measures usually exist along a continuum whereby they are implemented according to a particular order. The process of repairing degraded lands often starts with reclamation and revegetation and ends in rehabilitation and ultimately restoration (Artiola *et al*. 2004; Favas *et al*. 2018).

According to the Society for Ecological Restoration (SER), reclamation refers to the stabilization of terrain, assurance of public safety and aesthetic improvement (Balensiefer *et al*. 2004) for the purposes of restoring biotic function and productivity in severely degraded lands (Artiola *et al*. 2004). Revegetation, a component of land reclamation, is the active planting of native vegetation to repair a disturbed or degraded area (Artiola *et al*. 2004; Balensiefer *et al*. 2004; Munro *et al*. 2007).

The SER goes on to define rehabilitation as the reparation of ecosystem goods, services, processes and productivity (Aronson *et al*. 1993; Artiola *et al*. 2004; Balensiefer *et al*. 2004; Favas *et al*. 2018). Restoration is defined (again by the SER) as “the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed” (Balensiefer *et al*. 2004). While restoration pertains to returning an ecosystem to its historical (though not necessarily pristine) state via the establishment of indigenous vegetation (Aronson *et al*. 1993), rehabilitation is the restoration of ecosystem functions to enhance ecosystem productivity for the benefit of local people (Aronson *et al*. 1993).

As distinct as these two concepts are, in this context it can be said that rehabilitation is the foundation upon which restoration of degraded landscapes is built and

ultimately achieved. This is because in this study, local indigenous vegetation was used in combination with intensive soil rehabilitation techniques to revegetate and rehabilitate degraded areas with the long-term objective of returning the ecosystem to its near-historical state. Therefore, for the purposes of this study, the implementations used will be those of rehabilitation for long-term restoration.

Within the African context, addressing land degradation has prompted the African Union and partners to create and launch the African Forest Landscape Restoration Initiative or AFR100 (Messinger & Winterbottom, 2016). The aim of this initiative is to restore 100 million hectares of degraded forest landscapes by 2030 (Reij & Garrity, 2016).

Active rehabilitation techniques in degraded landscapes are often applied with the aim of improving soil water status through increasing infiltration or decreasing surface runoff and evaporative water loss (Thurow, 2000; Hanke *et al*. 2011) via translocating plant species (Anderson *et al*. 2004), brush packing (Tongway & Ludwig, 1996; Ludwig & Tongway, 1996; Todkill *et al.* 2006) organic mulches and digging pondings to capture run-off (Li *et al*. 2006). These techniques are applied with the aim of re-establishing perennial native vegetation cover, improving foragefor livestock, and protecting soil against erosion, surface compaction and salinization (Botha *et al*. 2008; Hanke *et al.* 2015).

Ecological rehabilitation for restoration can potentially reverse land degradation, increase ecosystem resilience and carbon sequestration as well as restore andprotect biodiversity and ecosystem services (CBD, 2010; Wortley *et al*. 2013). This increased focus on reversing the trend of global land degradation led to the World Bank and the Global Environment Facility (GEF) establishing the Subtropical Thicket Ecosystem Planning (STEP) project to promote the conservation of a globally important biodiversity hotspot in South Africa (Cowling & Knight, 2006) known as the Subtropical Thicket (Low & Rebelo 1996). For the purpose of this study, the Subtropical Thicket Biome is hereafter referred to as the Albany Thicket Biome (Hoare *et al*. 2006).

1.3. Albany Thicket Biome (thicket)

Out of South Africa's current nine biomes (Mucina & Rutherford, 2006), the Albany Thicket Biome (Hoare *et al*. 2006) is the most recently classified Low & Rebelo 1996). Due to its abundance of woody and succulent species, this biome was once thought to be a vegetation type within the Savannah and Nama Karoo biomes; being classified as Valley Bushveld and Noorsveld respectively (Acocks, 1953). However, its structural and floristic characteristics were so unique that it was later classified as a biome on its own (Fabricius *et al*. 2002; Vlok *et al*. 2003; Cowling *et al*. 2005 & Hoare *et al*. 2006). The thicket lies on the eastern seaboard and is most concentrated in the semi-arid river valleys of the Eastern Cape from the Gamtoos River in the west to the Great Kei River in the east (Vlok *et al*. 2003; Cowling *et al*. 2005; Hoare *et al.* 2006).

It is a relatively small biome covering only 2.2% of the land surface area of South Africa's landmass, including Lesotho and Swaziland (Hoare *et al.* 2006; van Oudtshoorn, 2015). The thicket is characterised by a dense matrix of hardy, spinescent, evergreen shrubs (± 5 m in height), succulents and bulbs; all of which grow closely together in bush clumps which eventually expand and connect with eachother (in the absence of disturbance regimes-such as fire - that would prevent consolidation of said bushclumps) (Vlok & Euston-Brown, 2002). In its intact state, thicket vegetation is generally considered to be impenetrable (van Oudtshoorn, 2015). Forming part of the Maputaland-Pondoland-Albany hotspot (Mittermeier *et al.* 2011), the thicket is therefore a habitat supporting high levels of biodiversity and endemism with 20% of thicket species endemic to the biome (Vlok *et al*. 2003).

The climate in this biome ranges from semi-arid to subtropical, with average rainfall varying from 200 mm to 950 mm per annum (Hoare *et al.* 2006; van Oudtshoorn, 2015). However, the rainfall is aseasonal, with peaks in Spring and Autumn but it is unreliable and droughts lasting several months are a common occurrence (van Oudtshoorn, 2015). Furthermore, the Intergovernmental Panel on Climate Change (IPCC) projects with medium to high confidence that temperatures and droughts in South Africa will increase (IPCC, 2021).

In fact, the Eastern Cape is susceptible to year-long peak drought conditions, with a 33% likelihood of experiencing droughts that last over 24 months (Botai *et al*. 2020), with the drought from 2015 to 2019 being especially severe (Mahlalela *et al*. 2020). This drought has cost the province over ZAR 120 million in drought relief measures and has had major socioeconomic effects on the vulnerable rural settlements (and in some urban areas) where water supply services have been restricted and even broken down (Mahlalela *et al*. 2020).

Despite these constraints, intact thicket - particularly thicket rich in *Portulacaria afra* has high levels of litter production for a semi-arid ecosystem. It has been recorded that litter production in intact *P. afra*-rich thicket can reach 4 100 kg ha^{-1} annually in dry matter (Lechmere-Oertel *et al*. 2008). The litter, coupled with the microclimate created by the dense thicket canopy, maintains and improves soil structure, fertility, soil moisture and organic carbon content. This then translates to the biome's extraordinary ability to store up to 245 t C ha^{-1} (in biomass and soils) (Mills *et al*. 2005a), with literature recording a mean value of 209 ± 28 t C ha^{-1} (Mills *et al*. 2005b). These figures suggest the carbon-sequestering ability of *P. afra* to be extraordinarily high for a plant species adapted to xeric landscapes (Mills *et al*. 2005a). These microclimatic conditions enhance plant productivity, thereby contributing to the carrying capacity of the biome; so much so that in its intact state, the thicket can support relatively high densities of wildlife for a semi-arid ecosystem, including mega-herbivores such as the African elephant (*Loxodonta africana*) and black rhinoceros (*Diceros bicornis* subsp. *bicornis*) (IUCN, 2010; Mills *et al*. 2014), which are the main drivers of thicket ecology (Kerley *et al*. 2004).

Thicket degradation

Adapted to browsing by indigenous herbivores as the thicket is, it is highly vulnerable to overutilization by livestock (Stuart-Hill, 1992). Farmers that settled in this part of the Eastern Cape during the 19th Century introduced Angora (*Capra aegagrus*) and Boer goats (*Capra hircus*) to browse on the lush and seemingly endless biomass, a practice which reached a commercial boom in the 1960's (Mills et al. 2005b).

These domestic herbivores eat thicket plants from the bottom up (Stuart-Hill, 1992), feeding on the lower branches of vegetation responsible for shading the soil and organic matter. This differs from the top-down browsing by indigenous herbivores, to which thicket vegetation has adapted over centuries (Vlok & Euston-Brown, 2002). Although overstocking with indigenous herbivores can lead to a reduction in canopy volume (Cowling & Kerley, 2002), thicket is so sensitive to injudicious pastoralism that heavy browsing by domestic herbivores can degrade and transform the ecosystem (Kerley *et al*. 1995; Kerley *et al*. 1999a; Kerley *et al*. 1999b; Lechmere- Oertel *et al*. 2005).

The degradation process in the thicket is most notably observed in areas overstocked with goats and their browsing habits (Stuart-Hill, 1992). The bottom-up browsing opens the thicket canopy and leads to reduced litter production. The topsoil is then exposed to the elements resulting in decreased soil organic carbon, soil moisture content and soil water potential. These changes then lead to increased soil temperatures, decreased infiltration and soil erosion (Lechmere-Oertel *et al*., 2008). According to Mills *et al*., (2005b), overgrazing results in a loss of soil C and biomass C (approximately 24 t C ha^{-1} and 58 t C ha^{-1} respectively). This leakage of nutrients impedes the function of nutrient and water capture; vital functions responsible for the establishment and persistence of thicket (Vlok & Euston-Brown, 2002), thus continuing a vicious cycle of degradation.

Eventually the thicket transforms from a dense closed-canopy shrubland into a pseudo-savanna system, characterised by ephemeral grasses and dwarf karroid shrubs with many of the canopy remnants growing in an umbrella thorn tree-like form because of bottom-up herbivory (Lechmere-Oertel *et al*., 2005). Over the last 150 years, approximately 800 000 hectares of the thicket has been degraded due to overstocking of goats (Lloyd *et al*. 2002), ultimately resulting in functional loss and desertification (Lechmere-Oertel *et al*., 2008; Sigwela *et al*. 2009; Cowling & Mills, 2011). The degradation of this biome drastically diminishes natural resources used for socio-cultural and economic needs (food, fuel, medicinal plants) (Lloyd *et al*., 2002). Another outcome of degradation is the reduction of carrying capacity for livestock and wildlife (Stuart-Hill & Aucamp, 1993), which diminishes opportunities for socio-economic activities such as ecotourism and game farming (Smith & Wilson, 2002). This suggests that compared to alternatives such as game ranching and

ecotourism; both of which have been identified as the most desirable activities in this biome (Kerley *et al.* 1999a), pastoralism is not an ideal land-use option for thicket as it diminishes productivity (Kerley *et al.* 1999b).

Thicket rehabilitation for restoration

Since degradation results in the loss of the biophysical conditions necessary for maintenance of ecosystem structure and function (i.e., soil organic carbon, soil moisture content, etc.) (Mills *et al*. 2005), xeric thicket cannot regenerate autonomously, thus merely removing the cause of degradation (such as injudicious pastoralism) and resting the land is insufficient to restore the ecosystem function (Vlok *et al*., 2003; Lechemere-Oertel *et al.* 2005; Sigwela *et al*. 2009). This is because the xeric thicket (according to the STM) has crossed an ecological threshold and has shifted to a new regime and requires extensive human intervention to reverse its trajectory (Bestelmeyer, 2015). Therefore, without human intervention, the negative impacts of thicket degradation (i.e., soil erosion, reduced base flows in drainage systems, siltation of water bodies) will persist (Lloyd *et al*., 2002).

Active measures need to be implemented for restoration to be successful. Therefore, propagating *P. afra* truncheons on degraded land has been considered the most viable means of xeric thicket restoration (Mills *et al*. 2005; Mills & Cowling, 2006; Mills *et al*. 2007). This evergreen, succulent shrub (reaching up to 5 m in height) (Oakes, 1973; Baran, 1999) has a wide distribution stretching from the Western Cape to Limpopo (Van Jaarsveld & Le Roux, 2021) but is most commonly found in xeric forms of Albany Thicket; currently occupying 1.7 million hectares in the Eastern Cape (Mills *et al*. 2005). While *P. afra* is prominent in Valley Thicket, Noorsveld and Spekboom Thicket, it is especially common in Spekboomveld (a xeric thicket vegetation type that occurs on the mainland) where it is a dominant canopy constituent; making up nearly 90% of the canopy and forming a closed canopy shrubland (Vlok *et al*. 2003; Mills *et al*. 2005). The succulent primarily occurs in frost- free regions as exposure to heavy sustained frost often leaves a negative impact on *P. afra* growth and survival (Duker *et al.* 2015). It is drought-resistant, relatively fast- growing and can be easily propagated via truncheons which root in the soil in a matter of weeks via vegetative reproduction (Swart et al. 1994; Van der Vyver et al. 2012).

Once its roots have successfully established, *P. afra* is resilient to light browsing pressure from indigenous herbivores (Stuart-Hill, 1992; Sigwela *et al*. 2009). The top-down browsing of *P. afra* by elephants and black rhinos stimulates the growthand proliferation of its lower branches which root once the nodes reach the ground surface, all while forming a protective 'skirt' with its lower branches that shades the soil. This allows *P. afra* to act as a nurse plant, creating a shaded, cool microclimate and enriching the soil, thus establishing conditions likely to facilitate recruitment of other thicket species, encouraging a return of thicket biodiversity (Sigwela *et al*. 2009; van der Vyver *et al*. 2013) and thereby enhancing the plant's reputation as a soil binder (Oakes, 1973; Powell, 2009).

Furthermore*, P. afra* has also been found to be a facultative Crassulacean Acid Metabolism (CAM) species (Ting & Hanscom, 1977; Guralnick & Gladsky, 2017). This means that it can perform daytime CO_2 uptake via traditional photosynthesis but can utilize CAM pathways (nighttime CO_2 uptake) when water-stressed (Guralnick & Jackson, 2001). *P. afra* was first shown to utilize nocturnal CO_2 uptake and large acid fluctuation in times of water scarcity by Ting & Hanscom (1977). However, later studies indicated that CAM response is seasonal and related to long day photoperiods, showing more CAM activity during the hot summer months (due to higher temperatures) (Guralnick *et al*. 1984a, b; Guralnick & Gladsky, 2017). This unique ability to switch between C_3 and CAM daily maximizes the carbon uptake by *P. afra* (Guralnick & Gladsky, 2017). Indeed, one hectare of *P. afra* may sequester up to 4.2 t C yr^{-1} over a 27-year period (Mills & Cowling 2006). In addition to these findings, van der Vyver *et al*., (2013) found that a 50-year old post-restoration site can hold large carbon stocks (250.8 ± 14 t C ha^{-1}) similar to those in intact thicket stands (245 ± 28 t C ha^{-1}) (Mills *et al*., 2005a).

Despite these attributes, a detailed cost/benefit analysis of thicket restoration over time is required to determine the overall efficacy of large-scale *P. afra* planting as means of achieving thicket restoration (Mills & Robson, 2017). Nevertheless, current literature states that the restoration of degraded thicket using *P. afra* could potentiallyresult in the rapid return of carbon to the ecosystem (Powell, 2009, van der Vyver *et*

al., 2013). It is this capacity for carbon capture that provides the foundation for soil recovery, a key step towards returning thicket to its former biodiversity through spontaneous recruitment of other canopy species (van der Vyver *et al.,* 2013). Furthermore, it is argued that planting *P. afra* truncheons as a monoculture can encourage a return of biodiversity as the *P. afra* canopy would create a cooler micro-climate and a dense litter layer, ideal conditions for spontaneous recruitment of other thicket canopy species (van der Vyver *et al*., 2012; van der Vyver *et al*., 2013).

1.4. Subtropical Thicket Restoration Project (STRP)

As previously mentioned, (see Chapter 1.3), the degradation of over 800 000 hectares of thicket has led to a drastic reduction of carrying capacity for livestock andwildlife (Lloyd *et al*. 2002). Consequently, natural resources necessary to meet socio-cultural and socio-economic needs are diminished (Lloyd *et al*. 2002) and this further exacerbates what is an already dire situation for the locals that depend on the thicket. South Africa's socio-economic situation is characterized by "high unemployment, slow employment growth, high earnings inequalities, low wages for low-skill workers relative to the cost of living and a level of human development normally associated with a poor country" (Altman & Hemson, 2007). These circumstances are further illustrated by StatsSA (2021) reporting an unemployment rate of 34.4%.

The Eastern Cape is the country's second largest province, supporting the third largest total population of all nine provinces. With a poverty rate of 67.3%, it is the second-most poverty-stricken province in the country (next to Limpopo which has a poverty rate of 67.5%) and is therefore one of the provinces in most need of socioeconomic development (StatsSA, 2018). To combat household poverty while equipping the unemployed with training and skills required for employment, the Expanded Public Works Programme (EPWP) was implemented and has been running since 1994 (Phillips, 2004). It was through the EPWP that the South African government initiated the Subtropical Thicket Restoration Programme (STRP) to facilitate, through public and private sector investment, the restoration of degraded landscapes within the 75 000 km² of thicket spanning across the Eastern and Western Cape Provinces (Mills *et al*. 2015). As part of the EPWP, the STRP

focused on providing the unemployed with temporary work in fields such as government-funded infrastructure projects and public environmental improvement programmes (Phillips, 2004).

Considering the above findings on thicket degradation and the importance of its restoration, the South African government invested over $8 million (US) into the STRP in 2004 (Mills *et al*., 2015). This initiative has focused on restoring large areas of degraded thicket via the propagation of *P. afra* (Powell *et al*. 2006; Mills *et al*. 2007), while also creating livelihood opportunities for the rural poor and standardizing best management practices for entering the carbon market (Marais *et al*. 2009). The single standardised planting protocol used by the STRP for *P. afra* involves planting unrooted truncheons ~ 30 cm in length; ~30 mm in basal stem diameter) in 2 x 2 m rows, equating to a density of 2 500 truncheons per hectare (Powell, 2009; Lagerwall, 2010). The truncheons are planted at a depth of 15 to 20 cm as this is the recommended depth at which truncheons are to be planted for successful establishment (Powell, 2009). In a study conducted by van der Vyver & Cowling (2019), the aboveground carbon (AGC) pool estimates between intact and degraded *P. afra*-rich thicket were compared to one another. Having found the ranges in intact and degraded thicket to vary from 43.0 to 26.0 t C ha^{-1} and from 13to 2.5 t C ha^{-1} in dry matter respectively, the authors suggest that setting an AGC target of 23.8-32.8 t C ha^{-1} after 20-30 years of restoration action is therefore realistic under current restoration protocols (van der Vyver & Cowling 2019).

Despite vast investment into the STRP from the government, many limitations and pitfalls were encountered; chief of which involved the establishment and growth of newly planted *P. afra* truncheons. According to Lagerwall (2010), survivorship of planted *P. afra* cuttings in the restoration trials has been found to be variable across different parts of the thicket, due to local climatic factors and existing soil biophysical conditions. Lagerwall (2010) further states that in an STRP trial conducted in the Addo Elephant National Park (AENP), mean survivorship of *P. afra* planted in 2007 and 2008 amounted to 26% and 33% respectively. In a recent study in AENP, the Great Fish River Nature Reserve (GFRNR) Camdeboo National Park (CNP) and the Baviaanskloof Nature Reserve (BNR), Mills & Robson (2017), monitored a sample size of 200 000 *P. afra* cuttings to estimate survivorship for a population of 21.5 million cuttings planted across these protected areas. In this study, they found

a mean survivorship of 28%. These low levels of survivorship not only undermine the progress of restoration programmes, but they also diminish their cost- effectiveness; with the need to replace dead *P. afra* representing extra costs (Lagerwall, 2010).

Major hindrances to *P. afra* survivorship, as observed in Baviaanskloof include extreme temperatures; hard, capped soil with little to no water infiltration and retention and browsing pressure from domestic and wild herbivores (Powell, 2009). Truncheons that were planted in the soil surface were observed to have ultimately died due to desiccation (Plate 1) because the soil was capped, allowing for little orno infiltration of water down to the base of the truncheon planted at least 10cm deep. Herbivores represent a significant threat to newly planted *P. afra* because the truncheons have not established well enough to withstand the browsing. This often results in the truncheons getting pulled out or pushed over by game before establishing enough roots to keep them anchored to the ground (Plate 1) and/or excessive browsing that ultimately kills the plants (Powell 2009; Lagerwall, 2010). It has also been found that sustained herbivory pressure on newly planted *P. afra* truncheons reduces aboveground biomass and belowground root biomass (Starkey, 2015), further emphasizing the importance of limiting, if not removing, herbivore pressure (Berriman, 2010).

Furthermore, continual browsing pressure on *P. afra* prevents the plant from forming the vegetative skirt required to shade the soil and collect organic matter, as a result that organic matter is lost to the landscape (Powell, 2009; Berriman, 2010). This is an especially prominent pitfall as successful thicket restoration is largely dependent on increasing the rates of organic matter return into the degraded soil (Lechmere-Oertel *et al*. 2008). Finally, *P. afra* is highly palatable and has no natural defence mechanism against herbivores, thus increasing its vulnerability (Mills & Cowling, 2006).

Plate 1.1 Dead *P. afra* truncheons that were planted in the Baviaanskloof as per the STRP planting protocol.

The main pitfall encountered in the STRP is the entrenchment of the restoration protocol in the programme's operations without continual review of its efficacy (Mills *et al*. 2015). According to Mills *et al*., (2015), modifying the existing planting protocol will increase survivorship of plants, reduce restoration costs, increase income streams from restored landscapes and promote new financing mechanisms for restoration.

1.5. Restoration techniques used in the study

The restoration of degraded ecosystems (dryland ecosystems in particular) entails soil and water conservation (SWC) practices which enhance the productivity of areas affected by, or prone to, degradation, known as SWC technologies (WOCAT, 2007). These technologies are divided into nine groups including water harvesting, gully rehabilitation, conservation agriculture, terraces and grazing land management. For the purposes of this study, three SWC technologies were applied, namely water

harvesting, conservation agriculture (which include companion/nurse planting) and grazing land management, while the remaining seven technologies are more agriculture-based and are therefore excluded from this study. Firstly, water harvesting is a relevant technique because rehabilitating degraded thicket requires water harvesting measures to maximise water availability for the indigenous vegetation (Powell 2009; Lagerwall, 2010). Conservation agriculture (WOCAT, 2007) involves the use of companion and nurse plants while grazing land management involves the reduction (if not exclusion) of grazing/browsing pressure on the newly rehabilitated area (Powell 2009; Berriman, 2010; Lagerwall, 2010; Starkey, 2015). The restoration techniques covered below are based on these SWC technologies.

1.5.1. Construction of Pondings

With respect to highly degraded ecosystems, a recommended approach towards their rehabilitation and ultimately restoration would be to increase the number of resource-traps, to facilitate recovery of ecosystem structure and function (de Abreu, 2011). One of the ways in which to carry this approach is through the creation of pondings. These pondings (otherwise known as micro-catchments or micro-dams) are man-made depressions that are dug (often by hand) to harvest rainwater and channel it towards the rooting zone of native plants (Powell, 2009). They are usually 20-60 cm deep and 50-90 cm in diameter, with the loosened soil used as a berm downhill of the ponding (de Abreu, 2010). These depressions are often dug in areas where water and/or resources needed for irrigation are limited and are an economical means of utilizing available water to create healthy and productive vegetation patches (Kimiti *et al.* 2017).

Constructing pondings to conserve water and soil has been a long-standing technique in many countries across the world (Vohland & Barry, 2009). Indeed, in Africa's arid Sahel region, where the pondings are locally referred to as zai pits (Burkina Faso) or "tassa" (Niger) have been applied to harvest water for improved crop yields (Motis *et al.* 2013). Today, under active promotion by NGOs, natural agricultural extension service and government agencies, pondings have mostly been constructed for agricultural crops and much less in the interests of rangeland rehabilitation or ecosystem restoration (Vohland & Barry, 2009).

These artificially created patches act as sinks for vital resources (water, nutrients and soil) (Li *et al*. 2006; Hanke *et al.* 2011), and seeds; thereby creating a suitable microclimate for the recruitment of native plant species and biological soil. Botha *et al*., (2008) substantiate this point in their study to capture restoration knowledge of local resource users in lowland Namaqualand. In the interviews they conducted, the respondents stated that micro-topographic depressions such as bulldozer tracks and cracks in the soil were observed to be sites of plant establishment. Indeed, using pondings in thicket restoration efforts has proven to increase patchiness in degraded thicket and thus create ideal conditions for the establishment of *P. afra* (Powell, 2009) as well as that of other indigenous species (Otto & Maya Beukes, pers. comm. September 2017). Taplin (2019) found that pondings provide a microclimate and become sites of seed production as plants mature in and around the pondings. Furthermore, Mills & Fey (2004a) found that overgrazing and transformation of *P.afra*-rich thicket into savannah reduces soil quality so much so that the soil becomes more prone to crusting and capping, as well as erosion of the organic-rich surface layer. These consequences of thicket degradation reduce infiltration rates and plant productivity, ultimately hindering the natural regeneration, and active restoration, of degraded thicket (Mills & Fey, 2004). The construction of pondings, therefore, can remedy these consequences not only by allowing water to infiltrate into the soil (thus encouraging recovery and storage of soil moisture and baseflow improvement) (van Luijik *et al*. 2013; Taplin, 2019) but also by allowing organic matter to gather within the planting sites, further enriching these islands of fertility (Dastorani *et al*. 2017).

1.5.2. Clump planting

The STRP protocol of planting *P. afra* as mentioned above, involve planting single truncheons in closely packed 2 x 2m rows (Mills *et al*. 2007). The protocol, however, produced mean survivorship levels ranging from 33% (Lagerwall, 2010) to 28% (Mills & Robson, 2017) as previously mentioned (see 1.4 – pg. 14). The variation in survivorship can be attributed to the institutional set up of the programme as well as oversight and management efficiency issues. Because the institutional reflection and evaluation of the protocol would only manifest over a decade after its conceptualization, the STRP hadn't adopted an adaptive learning approach for refining planting protocols (Mills *et al*. 2015). Furthermore the 2 x 2 m protocol has

proven to be largely unsuccessful in the Baviaanskloof (Melloson Allen. pers. comm. September 2017).

However, in a *P. afra* planting project conducted in the CNP, Mills & Robson (2017) were informed by the South African National Parks Board (SANParks) Management that *P. afra* planted as clusters in trenches or micro-basins had greater survivorship compared to *P. afra* planted in 2 x 2 m rows. The depressions in which the clusters of *P. afra* were planted also captured rainwater and nutrients; thereby contributing to the increased survivorship and the creation of new vegetation patches.

Ludwig *et al*., (2005) weigh in on vegetation patchiness by stating that a pulse in plant growth in said patches not only maintains structure, but also increases it so that the vegetation is better able to obstruct and absorb overland flows across the landscape from precipitation and wind transportation of organic matter. Furthermore, the relatively poor water infiltration in the inter-patch area adds to the overland flow that accumulates at the vegetation patches (Stavi *et al.* 2009). This enhances the patches' hydrological function and can assist in mitigating drought conditions while maintaining conditions suitable for plant growth (Ludwig *et al*. 2005).

Planting *P. afra* in clumps therefore has the potential to fast-track canopy closure, thereby contributing towards the creation of a litter layer (Powell, 2009) and thus a return of soil fertility through increased soil carbon in the long-term (Lechmere-Oertel *et al*. 2008).

1.5.3. Brush-packing

Brush-packing is a restoration technique that involves covering the soil surface with brush from woody plants (Gil, 2012; van Oudtshoorn, 2015). Its application is based on the principle that semi-arid landscapes are organized in patches that form a source-sink system in which mature shrubs form islands of fertility (Ludwig & Tongway 1996). This principle is especially relevant to the study since the ecosystem structure and function of thicket is maintained by the presence of these islands of fertility (Vlok & Euston-Brown, 2002) and the microclimate that exists within them that encourages vegetative growth (Padilla & Pugnaire, 2006). In degraded areas where such shrubs

would be scarce or absent, brush-packing can provide newly planted seedlings and cuttings with such a microclimate by providing shade as well as protection from herbivory (Hanke *et al.* 2011).

Furthermore, brush-packing is suggested to facilitate accumulation of water (via shading and airflow reduction, thereby reducing evaporation) and the establishment of healthy vegetation patches across degraded land (Hanke *et al.* 2011). This is achieved by creating microclimates favourable for vegetative growth while trapping windblown organic matter (i.e., leaf litter) and buffering against rain splash erosion on the ground (Hanke *et al.* 2011). The findings of Todkill *et al*., (2006) substantiate this theory, with brush-packing significantly increasing the percentage of dry litter coverage of the plant area (the litter usually coming from the brush-pack itself as it degrades over time), thus improving the properties and condition of the soil. Furthermore, Todkill *et al*., (2006) found that once the brush-piles started decaying, basic nutrients and cations were released into the soil; thereby enriching it. It has primarily been used as a potentially cost-effective shelter for newly propagated plants (Visser *et al*. 2007), particularly from herbivory which often undermines survivorship of the new plants, especially indigenous thicket species such as *P. afra.* Powell, (2009). Berriman, (2010) and Todkill *et al*., (2006) found, in restoration trials, that brush-packing increased electrical conductivity in degraded thicket soil from 64 µS cm^{-1} to 220 µS cm^{-1} (indicating an accumulation of basic soil nutrients) and increased soil organic matter on the top three centimetres of the soil surface. Furthermore, van Oudtshoorn (2015) states that brush-packing also helps regulate topsoil temperatures. By protecting the soil from erosion and solar radiation, while trapping water, soil particles and organic material, brush-packing may also aid in creating islands of fertility in otherwise harsh conditions (Ludwig *et al*. 2005).

The process of rehabilitating and restoring degraded areas, particularly those within protected areas, should have a focus on implementing environmentally friendly procedures through utilizing locally sourced materials (Gil, 2012). The utilization of natural materials on their own or with constructions to improve soil structure is known as soil bioengineering measures (SBEM), which entails a combination of biological elements and engineering design principles (Evette *et al*. 2009). The concept of SBEM is not new, in fact it has been applied and recorded as far back as 28 BC

(Lewis *et al*. 2001). These historical examples include Chinese tapestries illustrating reparation of the banks of the Yellow River by bundling live stems (Lewis *et al*. 2001) and the establishment of Hong Kong's Feng Shui woodlands for ecological and cultural purposes (Zhuang & Corlett, 1997). With its successful applications in the past, SBEM can be considered an ideal approach to rehabilitating degraded sites.

Benefits of SBEM include the enhancement of natural capital through improved habitat and the reduction of restoration costs via the use of locally sourced materials, labour and relatively less need for machinery (Gil, 2012). Apart from the fencing and companion plants (which were both sourced from outside the Baviaanskloof), SBEM techniques were applied on the experimental treatments, namely via ponding construction, brush-packing and supplemental watering.

1.5.4. Supplemental watering

The alteration of biophysical conditions because of thicket degradation (i.e., capped soils, higher soil temperatures, loss of topsoil and lower soil water infiltration and retention) (Mills *et al.* 2005) will undermine natural succession processes; thereby preventing autogenic recovery (Vlok *et al*. 2003). These environmental constraints are expected to affect survivorship of newly planted *P. afra* (wild-harvested or nursery-grown, rooted and unrooted) as well. The compact clay soil and limited availability of water in degraded thicket have been observed to contribute to desiccation of newly planted cuttings (Powell, 2009). Furthermore, between 2014 and 2019, South Africa experienced a drought so persistent and so severe that it was declared a national disaster in March 2018 (Botai *et al*. 2020). These prolonged drought conditions can further exacerbate *P. afra* mortality (Boshoff *et al*., 2000). Therefore, supplemental watering after planting *P. afra* cuttings can help give the plants a head start by adding moisture to the planting zones, thus mitigating dry spells that are marked with uncertainty as to their duration.

Supplemental watering has been investigated in North Africa's Sahel region by Fox and Rockström (2003). What they found was that compared to non-irrigated sorghum (*Sorghum bicolor*), which produced an average rain yield of 455 kg ha^{-1},

sorghum that received what they referred to as supplemental irrigation produced an average grain yield of 712 kg ha^{-1}.

Water inputs can produce an enhanced pulse of plant growth (Ludwig *et al*. 2005); thus, creating islands of fertility and an improved microclimate in which new thicket clumps can establish (Vlok & Euston-Brown, 2002). Due to the scarcity of water in South Africa as well as the hardiness of *P. afra*, it can be hypothesised that watering rooted cuttings following planting in a ponding, around autumn when the temperatures are relatively cooler, will provide enough moisture on which the succulent can survive until the next rains arrive in the remote valley.

1.5.5. Nurse planting & Companion planting

The current planting protocol for thicket restoration focuses on the large-scale propagation of *P. afra* on degraded soils with no mention of other indigenous plants that can be propagated alongside it (van der Vyver *et al*. 2012). Findings by Louw (2012) and van der Vyver *et al*., (2012) state that using woody thicket species is economically and ecologically unfeasible for restoration owing to low overall establishment (>25%) and low overall survivorship (>20%), with the exception being *Rhigozum obovatum* (70%). Although intensive planting *P. afra* can encourage the autogenic regeneration of thicket canopy constituent species, this can only be achieved after leaving the restored site undisturbed for 35 to 50 years (van der Vyver,2011). Planting *P. afra* alongside or underneath other thicket species that would act as companion or nurse plants may help fast-track this recovery of ecosystem structure, function; all of which are important factors regulating species diversity (Lloyd *et al*. 2002).

A nurse plant is a plant that facilitates and aids the growth of other plant species by providing a canopy that ameliorates biophysical conditions such as light intensity, soil temperature and soil moisture content (Callaway, 2007). This creates an ideal microclimate in which other species can germinate and grow (Padilla & Pugnaire, 2006). A nurse plant can also protect seedlings and saplings from herbivory through associative defence (chemical or mechanical defence) (Smit *et al*. 2006).

A companion plant is a plant that is propagated alongside another plant for the purposes of higher yield, or pest control (Kuepper & Dodson, 2001). Both companion and nurse plants contribute to the growth of plants growing beside/underneath them in a process known as plant facilitation (Navarro-Cano *et al*. 2018). This process that benefits one species without harming the other is increasingly used as a tool for ecological restoration (Navarro-Cano *et al*. 2018). In interactions of facilitation between species, it is expected that functional complementarity between species will promote coexistence, since the possession of different traits might decrease niche overlap and competition (Cadotte *et al*. 2011). These morphological and physiological dissimilarities between co-existing plant species are known as functional distances according to Navarro-Cano *et al*., (2019), who conducted a restoration trial on mine tailings in south-eastern Spain. This experiment involved using 50 pairs of nurse-facilitated species of varying functional distances. The authors found that establishment of seedlings significantly increased with the greater the functional distance between the nurse and facilitated species. In other words, the more dissimilar the nurse and facilitated species are to each other, the more compatible they are to each other. Navarro-Cano *et al*., (2019) elaborate that the co- existence of functionally distant species is a long-term attribute, rather than a short- lived existence. Therefore, the long-term maintenance of restored ecological interactions could very well be an added benefit to using companion and nurse plants.

Navarro-Cano *et al*., (2018) noted that 11 out of the 13 potential nurse plants used in the experiment possess morphological and physiological attributes that not only facilitate other species in establishing themselves, but also facilitate a faster recovery of soil fertility, organic decomposition rates, nutrient cycling and resistance to erosion. This opens the scope for fast-tracking the restoration of species populations and ecological interactions as well as relevant ecosystem functions (Navarro-Cano *et al*. 2016).

1.5.6. Scarification of unrooted truncheons

In earlier studies investigating the potential for *P. afra* to be used as a restoration tool in xeric thicket sites, the main issue identified with planting *P. afra* truncheons was

the low survivorship (Lagerwall, 2010; Mills & Robson, 2017). This was often attributed to lack of root growth in the truncheons from being planted in capped soils that allowed little to no infiltration of surface water to the bottom of the truncheons (Powell, 2009). To determine whether root mass in *P. afra* could be enhanced through treatments, Norman (2016) conducted a laboratory experiment thatinvestigated the use of several pre-planting treatments for *P. afra*, in particular the scarification of unrooted truncheons. The rationale behind the scarification treatments was the theory made by Harmer (1988), stating that plants have epicormics buds can create roots wherever such buds are exposed. Going by that theory, Norman (2016) indeed found that scarification of unrooted truncheons led to root formation on the scars, at various distances from the cut basal stem area. However, the experiment from which these findings were produced were conducted in a controlled environment and was yet to be conducted as a field experiment.

The ability of *P. afra* to form roots at any part of the stem serves as an advantage for *P. afra* as surface water would have less distance along which to infiltrate before reaching the nearest epicormics buds (and thus the sites of root formation and growth). The scarified cuttings, with more exposed epicormics buds developed a greater root mass compared to cuttings with no scarification (Norman, 2016). However, since these findings were obtained in controlled conditions, I decided to investigate whether scarification of truncheons before planting them in the field (i.e., degraded thicket site) would increase their survivorship.

1.6. Aims and objectives

Having highlighted the constraints undermining progress in the STRP in South Africa, the aim of this study was to determine the influence of different planting and soil manipulation techniques on the survivorship of *P. afra.* The objective of which being, to examine innovative ecological restoration techniques that can potentially be used to rehabilitate degraded arid thicket.

1.6.1. Key research questions

1. To what extent does post-planting watering enhance *P. afra* survivorship and growth?
2. What is the survivorship of *P. afra* planted underneath a nurse plant vs *P. afra* planted with companion plants?
3. Are scarification and herbivory simulations potential aids in unrooted *P. afra* establishment and growth?
4. What influence does a modified planting protocol have on the photosynthetic efficiency of *P. afra* and the soil water potential and temperature of the pondings in which they are planted?

The findings of the study can potentially shed light on propagation techniques (or combinations thereof) that can aid in fast-tracking the rehabilitation of arid thicket and the ecosystem goods and services and income-generating opportunities for people who call this semi-arid ecosystem home.

Chapter 2. STUDY AREA

2.1. Location

The study area is located in the Baviaanskloof valley (33° 32' 3.1" S; 23° 59' 5.6" E). The valley, flanked by the Baviaanskloof mountains to the north and the Kouga mountains to the south, stretches from Uniondale in the Western Cape to Uitenhage in the Eastern Cape, with most of the wilderness area occurring within the borders of the Eastern Cape Province. The Baviaanskloof is part of the Cape Floristic Region characterized by high species diversity and endemism (Boshoff, *et al*. 2000; Boshoff & Cowling, 2005; Boshoff, 2008). This is therefore an area of great socio-ecological importance. As one of three protected areas within the Cape Floristic Kingdom identified as a Megareserve (minimum size: 250 000 ha), it was established in the interests of conserving and protecting large-scale ecological processes that sustain biodiversity in the long term (Boshoff, *et al*. 2000; Boshoff, 2008). In recognition of its significance, the Baviaanskloof was established as part of a UNESCO Cape Floristic Region World Heritage Site in 2004 (Boshoff, 2008).

2.2. Geology

The Baviaanskloof is a fault-rounded basin situated within the Cape Fold Mountains, which consist of highly folded and fractured Table Mountain Group sandstones (Holmes & Meadows, 2012). The predominant rocks occurring in the Cape Fold Mountains form part of what is known as the Cape Supergroup (Holmes, 2012). This supergroup consists of alternating layers of quartzitic sandstone and shale that were folded and uplifted together some 210 million years ago, because of increasing compression in the Southern Cape region (then a part of the Gondwana supercontinent) during the late Permian and ensuing Triassic periods (de Villiers, 1941; Boshoff *et al*. 2000; Lewis, 2008; Holmes, 2012). Another member of the supergroup, the Bokkeveld Group shales and the Table Mountain Witteberg Group (the youngest member) occur along the northern karroid plains on the south-facing side of the Baviaanskloof mountains (Boshoff, *et al*. 2000). The collision between the Falkland and African tectonic plates caused the folding of the Cape Supergroup

sediments, giving rise to the numerous basins and watercourses that exist in the Baviaanskloof today (Boshoff, 2008).

Erosion of the weak Bokkeveld Group shales produced the broad intermontane valleys present in the Baviaanskloof today while erosion of the Table Mountain Witteberg Groups formed the longitudinal mountain ranges (Holmes, 2012). The soils derived from the Table Mountain quartzites are nutrient poor, owing to the lack of feldspar, mica or even clay. The Enon Conglomerate consists of densely packed pebbles and rocks occurring in a sandy matrix while the Bokkeveld shales provide clayey, nutrient rich soils (du Toit, 1954; Truswell, 1970). Therefore, the Table Mountain quartzite-derived soils found on the mountaintops and the valley's higher slopes support Fynbos vegetation as it is accustomed to growing in nutrient-poor, acidic soils (du Toit, 1954). The lithology of the lower slopes consists of large outcrops, remnants of the overlying Enon Conglomerate that was formed in the central valley and derived from the erosion of the Cape Fold Belt (de Villers, 1941; Glenday, 2015; Muir *et al*. 2017).

The Bokkeveld shale-derived soils are restricted in the valleys and thus occupy positions favourable for soil production (du Toit, 1954). These deep, well-structured soils on the valley floors support thicket vegetation (Hoare *et al*. 2006), therefore it would be expected that such soils would promote and support the reestablishment of thicket species. However, as the study area is degraded, it can be expected that soil fertility has been reduced and along with unfavourable biophysical changes that arise from land degradation, it is a limiting factor to the rehabilitation of the thicket in the area.

2.3. Climate

The diverse topography of the Baviaanskloof, in addition to the mountains running parallel to each other from west to east influences the local climate. The summers are known to be extremely hot (maximum temperatures can exceed 45°C) and winters to be freezing (temperatures can drop below 0°C) with annual average maximum and minimum temperatures of 32°C and 5°C (Boshoff *et al*. 2000; Hattingh, 2011; Glenday, 2015).

Although the Baviaanskloof falls within the Cape Floral Kingdom's non-seasonal rainfall zone, rainfall in winter is usually dominated by frontal systems whereas summer rainfall is marked by short, intense convective events (Jansen, 2008; Glenday, 2015). The average rainfall experienced in the upper slopes and valley floors of the Baviaanskloof is about 800 mm and 200 mm respectively (Hattingh, 2011).

Although the average annual rainfall was recorded to be 270 mm for the period between 1970 and 2013 (Glenday, 2015), the Eastern Cape experienced a severe drought in all seasons since 2014. The drought has been so severe that in June 2020, the Kouga Dam, which is situated on the eastern edge of the Baviaanskloof, had been recorded to sit at just seven per cent capacity when in the beginning of 2015 it was almost full; its lowest levels since 1985 (Mahlalela *et al*. 2020).

2.4. Vegetation

According to the South African National Biodiversity Institute (SANBI), the Baviaanskloof in its entirety represents five of South Africa's nine biomes, namely thicket, Fynbos, Nama-karoo, Grassland and Forest biomes (SANBI, 2018). However, the immediate study site does not include the latter two biomes (Figure 2.1). The Succulent Karoo was represented in the Baviaanskloof until the latest update in the National Vegetation Map Project (VEGMAP) declassified Willowmore Gwarrieveld from the Succulent Karoo biome and made it part of the thicket (SANBI, 2018; Dayaram *et al*. 2019). This variety of biomes provides an extensive spectrum of ecological structures, patterns and processes in one area, which translates to high levels of biodiversity and endemism over a large area (Boshoff *et al*. 2000).

The thicket, which is well represented in the Baviaanskloof, exists in solid and mosaic vegetation forms in the relatively deep and fertile soils along the lower valley slopes. The main vegetation types of this biome are Sundays Arid Thicket and Baviaanskloof Valley Thicket (SANBI, 2018). Sundays Arid Thicket occurs in the northern edges of the Baviaanskloof and Groot Winterhoek mountains as well as the central and upper Baviaanskloof river valley (Hoare *et al*. 2006). This low and dense succulent thicket is present along moderate to steep slopes on the ridges of mountain ranges where low woody shrubs and succulents are well represented.

Under intact conditions, *P. afra* is abundant in dense stands and the grass component is poorly developed (Hoare *et al*. 2006).

The Baviaans Valley Thicket occurs along the Gamtoos and Baviaanskloof river valleys south of the Baviaanskloof mountains (Hoare *et al*. 2006; SANBI, 2018). Unlike the Sundays Arid Thicket, it is a tall, dense thicket that well represents the tree, shrub and succulent component. It is a vegetation type that mostly occurs in fragments, with dense, contiguous stands being restricted to the fire-resistant south and south-west facing slopes (Hoare *et al*. 2006).

The thicket vegetation in the Baviaanskloof is often dominated by *P. afra, Pappea capensis* (Jacket plum)*, Aloe speciosa* (Tilt-head aloe) and *Schotia latifolia* (Bush Boer-bean) (Hoare *et al.,* 2006). The riverine areas and kloofs support forest vegetation, characterized by *Celtis africana* (white stinkwood), *Podocarpus falcatus* (Outeniqua yellowwood) and *Ficus sur* (Cape fig tree) (Powell, 2009). The fynbos and bonteveld in the upper slopes and plateaux are home to plants such as various *Protea* species which thrive on the nutrient-poor, slightly acidic soils on the upper slopes in the Kloof (Boshoff, 2008). The northern flats of the Baviaanskloof represent Nama-karoo and Fynbos vegetation, marked by the presence of shrubs including *Pentzia incana* (ankerkaroo*)* and *Euphorbia esculenta* (vingerpol) while the deep alluvial soils on the valley bottom supports a savanna ecosystem; dominated by *Vachellia karroo* (Sweet thorn) and *C. africana* (Boshoff *et al*., 2000; Boshoff, 2008).

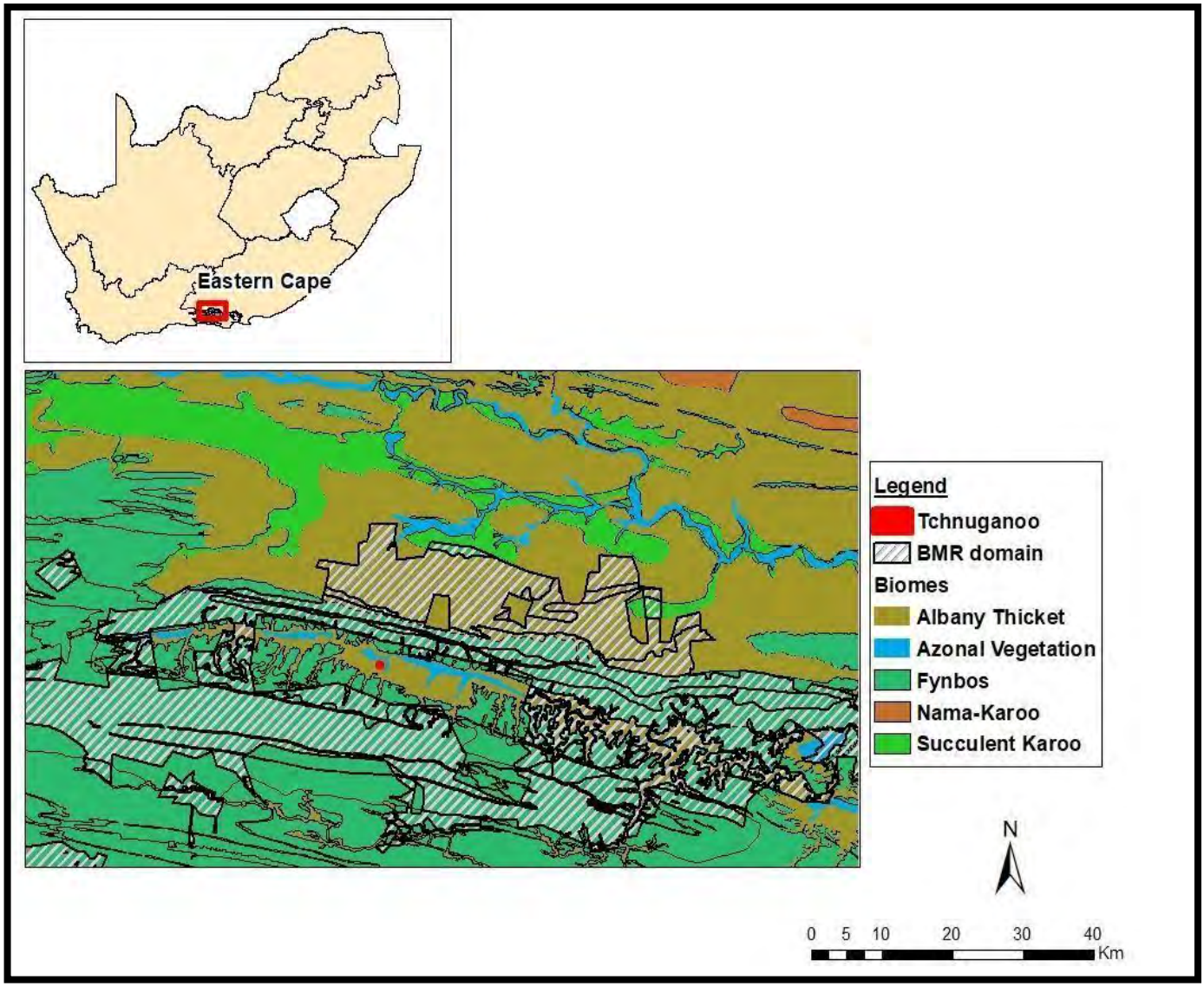

Figure 2.1. Biomes in the Baviaanskloof Megareserve (BMR) and planning domain. Note: Tchnuganoo is the name of the study site. (Shapefile: VegMap 2018; Dayaram *et al*. 2019).

2.5. Fauna

The complexity in topography and habitat, combined with the remote landscape have contributed to the area's richness and diversity in fauna. The Baviaanskloof is home to over 310 recorded bird species; including the globally threatened blue crane (*Anthropoides paradisea*) along with raptors such as the Martial Eagle (*Polemaetus bellicosus*) to which the World Heritage Site is a sanctuary (Boshoff *et al*. 2000).

Out of the 56 reptile species occurring in the Baviaanskloof, about 23 of them are endemic to South Africa, with only three of them having ever been recorded in the wilderness area. The Baviaanskloof is rich in amphibian and fish fauna, providing a home to 17 and 15 species respectively (Boshoff *et al*. 2000).

The Baviaanskloof is also habitat for over 46 species of medium to large-sized mammals. However, recent years of environmental exploitation, alien invasive tree infestation, and human-wildlife conflict have seen 14 of those species extirpated from their former range, according to Boshoff (2008). The Cape Mountain zebra (*Equus zebra zebra*), Cape buffalo (*Syncerus caffer*), eland (*Taurotragus oryx*) and red hartebeest (*Alcelaphus buselaphus caama*) have been re-introduced over the last two decades. Boshoff (2008), goes on to contend that records of large mammals that occurred historically in the greater Baviaanskloof area included all members of thebig five (lion, leopard, buffalo, leopard, and rhinoceros).

Currently three of the big five occur in the wilderness area, namely, Cape buffalo, black rhinoceros (*Diceros bicornis* sub-sp. *bicornis*) and Cape leopard (*Panthera pardus*). While the buffalo and black rhino have been re-introduced; the Cape leopard has persisted in this region for centuries despite conflict with livestock farmers (Minnie *et al*. 2015). A noteworthy species that naturally occurs in large numbers in the Baviaanskloof is the greater kudu (*Tragelaphus strepsiceros*). This large browsing antelope represents the biggest threat to the survivorship of planted *P. afra* cuttings (Powell, 2009). Newly planted cuttings are especially vulnerable as they get pulled out by kudu (as well as other wildlife) when feeding on them (Powell, 2009; Lagerwall, 2010).

2.6. History and land-use

The Baviaanskloof is rich in cultural history, serving as a meeting place for San hunter-gatherers, Khoekhoen herders, Bantu-speaking agro-pastoralists and European settlers (Boshoff *et al*. 2001). A wealth of bone artefacts and rock art foundin the Baviaanskloof is evidence (of up to 120 000 years old) of pre-colonial human occupation in the valley (Boshoff & Cowling, 2005). European settlers arrived in the 18th Century and although pastoralism was their main economic activity, commercial crop cultivation was done where arable soil was available. By the end of the 19th

Century, the San were displaced by the Khoekhoen and European settlers (Boshoff & Cowling, 2005).

By the beginning of the 20th Century, the boom in demand for Angora and Boer goats encouraged private landowners to continue rearing livestock and cultivating lucerne as their main economic activity to this day (De Paoli, 2008; Muchando, 2019). However, centuries of keeping their livestock in the once pristine thicket at unsustainable rates has led to overgrazing and ultimately the degradation of the thicket (Stuart-Hill, 1992; Mills *et al*. 2005b). This was especially true in the Western Baviaanskloof, where the high-density farming of livestock has resulted in the significant reduction of vegetation cover on the hillslopes (Euston-Brown, 2006; Sigwela et al., 2009; Glenday, 2015)

2.7. Field site

The field site is Tchnuganoo (33° 32' 38.5794" S; 23° 59' 8.484" E), an alternative farming community situated in the Western Baviaanskloof Heartland (Plate 2.1). The field site falls under Sundays Arid Thicket (a replacement of Groot Thicket) according to the 2018 version of the VEGMAP (SANBI, 2018) (Figure 2.2). It is characterised by the prominence of *P. afra* and the abundance of woody canopy species such as *Euclea undulata* (Common guarri), *Boscia oleoides* (Shepherd's tree) and *P. capensis* along with woody shrubs such as *Carissa haematocarpa* (Karoo num-num) and *Searsia longispina* (Spiny currant-rhus) (Hoare *et al*. 2006).

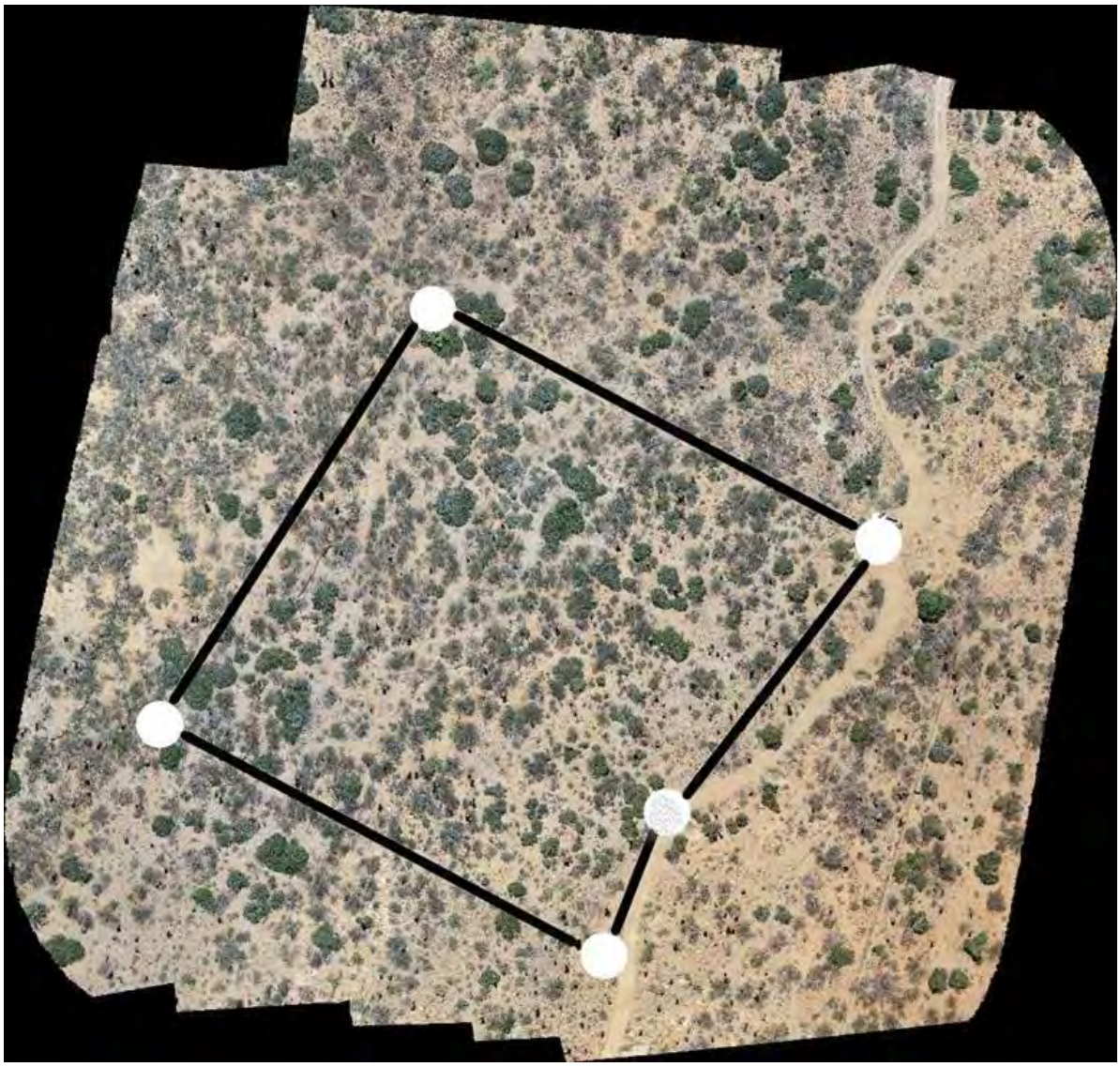

Plate 2.1. An aerial drone shot of the field site. Credits: Luyanda Luthuli (Living Lands). The white dots show the corners of the fenced-in field site.

Since the 19th Century, large-scale commercial agriculture (mainly livestock farming and lucerne cultivation) has been the primary economic activity in the Baviaanskloof Heartland (De Paoli, 2008). The thicket in the study site thus is degraded from decades of overstocking the area with livestock (particularly goats) (Lloyd *et al*. 2002; Boshoff *et al*. 2000). As a result of this degradation, high densities of *V. karroo*, a pioneer woody canopy species, occur within the study site.

Figure 2.2. Map of Tchnuganoo, along with associated biomes and the study site.

Chapter 3. EXAMINING THE INFLUENCE OF MODIFIED PLANTING PROTOCOLS ON ROOTED *PORTULACARIA AFRA* CUTTING SURVIVORSHIPIN DEGRADED ARID THICKET

3.1. Introduction

Land degradation in the thicket results in the alteration of biophysical conditions i.e., capped soils, higher soil temperatures and lower soil water infiltration and retention (van Luijk *et al*. 2013), thereby undermining natural succession processes and preventing autogenic recovery (Vlok *et al*. 2003). Consequently, the compact clay soil and limited availability of water in degraded thicket have been observed to contribute to desiccation of newly planted cuttings (Powell, 2009). Furthermore, the unpredictable rainfall patterns and frequent drought periods characteristic of the Baviaanskloof likely add to the variation in survivorship (Boshoff *et al*. 2000). Therefore, supplemental watering after planting *P. afra* cuttings may help give the plants a head-start by mitigating dry spells that are marked with uncertainty as to their duration.

Supplemental watering

Supplemental watering may potentially mitigate the effects of drought and heat waves as suggested by Rubin *et al*., (2018) when examining the effects of heat waves on C4 grass *Bouteloua gracilis* (blue grama) and the C3 grass *Festuca arizonica* (Arizona fescue). It was found under field conditions and the provision of weekly watering that lasted for a month that the C4 grass displayed a higher survivorship (92%) compared to the C_3 grass (22%) when exposed to heat waves. These findings were similar to those observed by De Boeck *et al.,* (2016) whereby in an alpine grassland, heat waves had a significant effect on fluorescence (an indicator of stress), senescence or aboveground productivity if supplemental irrigation was provided but the heat waves had a significant influence in the absence of irrigation.

Within the local context of thicket-related research, Mills *et al*., (2018) found that adding supplemental watering to *P. afra* in a nursery environment over eight months improved growth. Furthermore, applying four litres of water to unrooted truncheons and six litres to rooted cuttings resulted in a maximal increase in mean basal stem

diameter (5.6 ± 1.0 mm and 5.0 ± 3.7 mm, respectively). However, translating this protocol into the field would cost over ZAR 600 ha^{-1} monthly for truncheons planted 2 m apart compared to the monthly cost of ~R60 a month for truncheons planted in pondings, making the latter option far more cost-effective and feasible to implement than the former (Mills *et al*. 2018).

Having already mentioned the benefits of supplemental watering within a thicket context (see 1.5.4, pg. 37) it can therefore be hypothesised that applying watering regimes to rooted *P. afra* cuttings in pondings can help enhance their survivorship in a degraded thicket site.

Nurse & Companion planting

The current planting protocol for thicket restoration focuses around the large-scale propagation of *P. afra* on degraded soils, with little to no mention of other indigenous plants that can be propagated alongside it. Planting *P. afra* alone can present the risk of establishing monocultures instead of restoring the ecosystem structure and function of a biome renowned for its species richness and diversity. Planting *P. afra* alongside or underneath other thicket species that would act as companion or nurse plants can help fast-track the recovery of ecosystem structure, function; all of which are important factors regulating species diversity (Lloyd *et al*. 2002).

As explained in the previous chapter (see 1.5.5.-pg.38) companion plant is a plant that is propagated alongside another plant for the purposes of higher yield, and or pest control (Kuepper & Dodson, 2001) while improving soil fertility (Nathaniel, 2020). A nurse plant, on the other hand, facilitates and aids in the growth of other plant species by providing a canopy that ameliorates biophysical conditions, thus creating an ideal microclimate in which these beneficiary species can germinate and grow (Padilla & Pugnaire, 2006).

According to Cadotte *et al*., (2011), the successful establishment of functionally diverse species assemblages will ensure provisioning of a wide range of ecosystem functions in rehabilitated areas. This will then fulfil the restoration objectives of maintaining biodiversity as well as its associated ecosystem goods and services. Within the context of the thicket the concept of using companion and nurse plants in

thicket restoration is still a novel one which requires further investigation on its potential to fast-track thicket restoration and a return in biodiversity.

3.2. Experimental design and methods

The study tested the influence of different ecological restoration techniques on the growth of *P. afra*. These modifications consisted of the following experiments and techniques:

3.2.1. Watering regimes. This factor consisted of three levels of *P. afra*. These levels were monthly, bi-monthly and weekly irrigation. All replicates (rooted cuttings planted in pondings) received irrigation with 2L of water. Each of these three levels had 30 replicates. The rooted *P. afra* cuttings were sourced from the Tchnuganoo Nursery.

3.2.2. Nurse planting and companion planting. This experiment also consisted of three levels (all of which had pondings under monthly watering regimes). These levels were nurse planting (planting rooted *P. afra* cuttings underneath a nurse canopy), companion planting (planting rooted *P. afra* alongside nursery-propagated plants locally indigenous to the study area) and planting the *P. afra* in the absence of either nurse or companion species. All three levels had 30 replicates each. The companion plants used were *Lycium ferocissimum, Rhigozum obovatum* and *Aloe ferox*. The nurse plants used were *Carissa haematocarpa*. The companion plants were sourced from the Kouga Dam Nursery while the *C. haematocarpa* used were existing bushclumps on site.

3.2.3. Brushpacking. This experiment involved covering pondings (under monthly watering regimes) containing rooted *P. afra* cuttings with brushpack. There were two levels in this experiment, namely brushpacking and no brushpacking. Both levels had 30 replicates.

3.2.4. Unrooted truncheons. This 2 x 2 factorial experiment involved planting unrooted *P. afra* truncheons (in pondings under monthly watering regimes) under the following levels: Scarification, no scarification, herbivory simulation and no herbivory simulation. This factorial experiment had 30 replicates. The unrooted truncheons were harvested around the study area.

3.2.5. Plant health assessments. To determine the effects of watering regimes, nurse and companion planting, scarification and herbivory simulation on *P.*

afra vigour and survivorship, non-destructive measurements were taken. The variables considered were plant survival and leaf health.

According to agronomy literature, leaf colour (and the observed changes therein) is generally accepted to be a reliable indicator of plant health (Allison *et al*. 1954; Murakami, 2005). In the case of *P. afra* particularly, age and stress are reflected in the yellowing of the leaves, which are usually grey-green when healthy (Baran, 2011). Therefore, leaf colour was measured by observing the mature leaves in the cuttings as new leaves could differ in colour (Becker, 2013).

The cuttings would be given scores ranging from 1-4 (depending on their health) using an index derived from plant health assessments done by Lagerwall (2010) and Becker (2013). In this index, score "1" was given to a plant with green leaves. Score "2" was given to a plant with grey-green to yellow leaves. Score "3" was given to a plant with leaves turning a yellow-brown to burnt orange colour. Usually at this point the leaves would start falling off the plant. If a plant stem is devoid of leaves or signs of life, breaks easily and is non-fleshy upon breakage, that plant was given a scoreof "4" and categorized as 'dead' (Powell, 2009; Lagerwall, 2010). If plants were observed to be somewhat between two scores, then halfway scores were used (e.g., reduced/shrivelled *P. afra* leaves that still retain the healthy dark green colour got a score of 1.5).

The health of the companion species used in this study was assessed in a method similar to the one described above on the basis of leaf colour being a reliableindicator of plant health (Allison *et al*. 1954; Murakami, 2005). However, due to time constraints, the health of these companion species could only be categorized and subsequently assessed as "Alive" (Scores 1-2) "Stressed" (Score 3) and "Dead" (Score 4). These assessments were carried out monthly to ensure that data was collected across all four seasons.

After completing the health assessments, the sum of the *P. afra* cuttings/truncheons per score was calculated across the spring, summer autumn and winter seasons. It is important to note that these plant health assessments were performed for the watering regime, companion planting and nurse planting experiments, as well as for

the unrooted truncheon experiment in the following chapter (Chapter 4). The experiments are illustrated further in Figure 3.1.

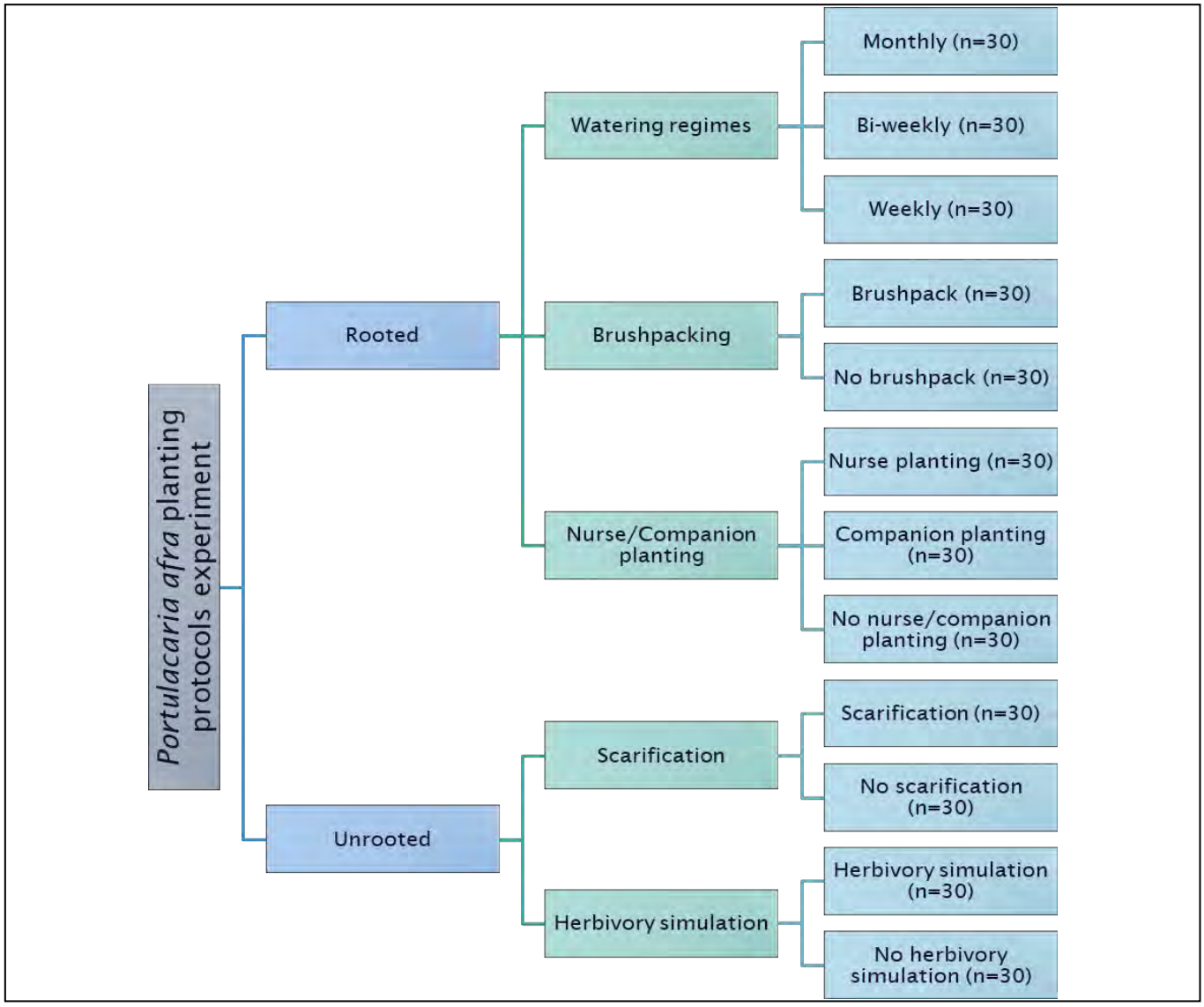

Figure 3.1. Experimental design for *P. afra* planting protocol experiments

It is important to note that while the control was not fully representative of the prevailing natural conditions of the study site as ponding construction, the use of rooted *P. afra* cuttings (with the exception of the unrooted truncheons experiment) and post-planting watering was standardized across the experiments. The reason for standardizing pondings across the experiments lies behind the purpose and effect of rainwater harvesting structures within a thicket restoration context.

Firstly, pondings constructed in the Baviaanskloof were observed by Powell (2009) to significantly enhance survivorship of *P. afra* planted upright (47.4 ± 2.0 % in 2006,

and 39.3 ± 1.9 % in 2008) compared to *P. afra* planted without a ponding (39 ± 1.5% in 2006 and 32.3 ± 1.5% in 2008) The truncheons were planted in clayey, shale-derived soils typical of degraded Sundays Arid Thicket (SANBI, 2018). Secondly, pondings create microsites that favour plant establishment and within thicket mosaics such microsites (that would occur naturally) would enable thicket clump establishment in an otherwise unfavourable environment (Vlok & Euston-Brown, 2002). This technique was therefore used as a foundation for all the experiments.

Secondly, it was essential to not only plant the rooted *P. afra* cuttings as clumps inside pondings but to also water the cuttings to aid in establishment. This is because the pitfalls of planting single unrooted truncheons in hard, capped soils (as per the STRP protocol) have already been discussed (see 1.4 - pg. 30). Furthermore,the lack of water provision for *P. afra* planted in degraded arid thicket according to the STRP protocol (along with the consequences thereof) has also been mentioned already (see 1.4 - pg. 30). I therefore found it unnecessary to repeat the STRP protocol in my experiments when it has already been implemented, critically evaluated and compared with protocol innovations in multiple studies (see 1.4 - pg. 30). This study solely involved experimenting with multiple ecological restoration techniques to determine the influence of such innovations on rehabilitation success.

3.3. Rooted *P. afra* experiments

3.3.1. Watering regimes

To conduct the watering regimes experiment, 120 pondings were hand dug in randomly selected plots within the study site to harvest rainwater and channel it towards the rooting zone of native plants (Powell, 2009). The pondings were ~30 cm deep and ~100 cm in diameter, with the loosened soil used as a berm downhill of the micro-catchment (de Abreu, 2011). Thereafter three rooted *P. afra* cuttings (grown in two litre planting bags and sourced from the Living Lands nursery in Tchnuganoo) were planted in each ponding (Plate 3.1) and were brushpacked with *V. karroo* branches to provide extra protection from herbivory (Plate 3.2).

The cuttings were planted upright as Powell (2009) found that *P. afra* truncheons planted upright showed significantly greater survivorship (53.2 ± 1.65% in 2006; 46.7 ± 1.6% in 2008) compared to truncheons planted flat (33.6 ± 1.56% in 2006; 24.9 ± 1.3% in 2008). Furthermore, Powell's (2009) findings suggest that planting truncheons

upright in pondings significantly enhanced survivorship of *P. afra* while planting them horizontally inside pondings increased mortality. The planting was done in autumn (late May) to avoid the intense summer heat which can prove lethalto newly planted *P. afra* cuttings (Powell, 2009; Lagerwall, 2010).

These 90 pondings were then divided into three watering regimes, each with 30 replicates of pondings that received 2L of water by hand, in varying frequencies. Thirty replicates were watered once every month, the second set of 30 replicates twice a month while the third set was watered weekly. The *P. afra* cuttings were watered under these regimes from winter (beginning of June 2018) until early Spring (beginning of September 2018) when data collection began. The 90 pondings were compared with a Control, which only differed in that it had no brush-packing. It should be noted that the spatial arrangement of the treatments was randomized.

Plate 3.1. Rooted *P. afra* cuttings planted in a ponding before having brushpacking added.

Plate 3.2. A ponding with rooted *P. afra* cuttings and brush-packing (using *V. karroo* branches). This is the standard protocol (modifications notwithstanding) implemented throughout the experiment.

It is important to note that the monthly watering regime and the control have also been added to a subsection in the results section (3.6.1.1, pg. 66) whereby they have been labelled “Brushpacking” and “No brushpacking” respectively. This subsection was added to examine the effects of brushpacking on *P. afra* survivorshipirrespective of watering frequency as both factors received the same amount of water on a monthly basis.

3.3.2. Companion planting

For this treatment, companion plants (in 5L bags) sourced from the Kouga Dam nursery were propagated. The companion plants used were *Lycium ferocissimum* (African boxthorn), *Rhigozum obovatum* (Wild pomegranate) (both woody thicket species) and *Aloe ferox* (a succulent shrub). One of each of the three companion plants were planted in a ponding alongside three *P. afra* cuttings (Plate 3.3). This treatment also received the standard manipulations of brushpacking and was watered once a month over four months. The Control for both the companion plants

and the nurse planting treatment consisted of 30 pondings, each with the aforementioned three *P. afra* cuttings with no brushpacking. The Control was also watered once a month over four months.

Plate 3.3. Companion species planted alongside *P. afra* in a ponding. Note the *A. ferox* planted in the centre of the ponding. White arrow = *A. ferox*; Red arrows = *P. afra* cuttings.

3.3.3. Nurse planting

For the nurse planting treatment, the *P. afra* rooted cuttings were planted in pondings excavated underneath a *C. haematocarpa* (Karoo num-num) canopy (outeredge of the canopy) (Plate 3.4). The plants were planted on the south-facing side of the canopy, which receives less solar radiation that the north-facing slopes and therefore provides milder conditions for plant establishment and growth. This

treatment received no brushpack but was watered once a month over three months. The Control for this experiment was 30 pondings (with three rooted *P. afra* cuttings), that also received no brushpacking but were watered once a month over four months.

Plate 3.4. *P. afra* cuttings planted under a nurse canopy (*C. haematocarpa*).

3.4. Data analysis

Data were analysed using R Software and Microsoft Excel. All data were tested for normality and equality among variances using Shapiro-Wilk's test. Where it was found to not be the case, a Kruskal Wallis test was performed to test for statistical significance ($p > 0.05$) between the different treatments using R and Microsoft Excel. For *P. afra* survivorship, a factorial ANOVA was performed to determine whether the different planting treatments significantly influenced overall survivorship and a repeated measures ANOVA was performed to determine whether the different planting treatments significantly influenced *P. afra* health over time.

3.5. Results

3.5.1. Watering regimes

3.5.1.1. Survivorship

For overall survivorship of *P. afra* a factorial ANOVA was performed to compare the effect of watering regimes on survivorship, revealing that there was a significant difference in mean survivorship amongst the treatments ($F(3,12) = [32.01] = p < 0.05$).

Furthermore, a Tukey HSD found that the mean values of survivorship was not significantly different between winter and autumn ($p = 0.21$, 95% CI, [-16.43, 1.83]) along with between summer and spring ($p = 0.4$, 95% CI, [-16.42, 3.11]). However, significant differences were found between spring and autumn ($p < 0.01$), summer and autumn ($p < 0.01$), winter and spring ($p < 0.01$) and winter and summer ($p < 0.01$).

A pairwise t-test using the Bonferroni method for multiple comparisons found that the mean values for survivorship were significantly different between control (89.3 ± 6.3%), and bi-weekly (78.5 ± 9.7%) ($p = 0.02$)), it was also found that mean survivorship for the weekly treatment (90.9 ± 6.8%) was significantly higher than biweekly (78.5 ± 9.7%) ($p = 0.005$) (Table 3.1) (Figure 3.2).

A pairwise t-test using the Bonferroni method for multiple comparisons found that the mean values for survivorship were significantly different between control (89.3 ± 6.3%), and bi-weekly (78.5 ± 9.7%) ($p = 0.02$) (Table 3.1), it was also found that mean survivorship for the weekly treatment (90.9 ± 6.8%) was significantly higher than biweekly (78.5 ± 9.7%) ($p = 0.005$) (Table 3.1) (Figure 3.2).

Table 3.1. Descriptive statistics for the watering regimes

Treatments	Mean survivorship (%)	Standard deviation
Control	89.3	6.3
Monthly	83.1	11.7
Bi-weekly	78.5	9.7
Weekly	90.9	6.8

A factorial ANOVA performed on the health scores showed that the control had a significantly higher number of healthy *P. afra* cuttings (Score 1) (58 ± 19.5) compared to the monthly (34.8 ± 12.7), weekly (27 ± 14.9) and bi-weekly watering regimes (17.8 ± 6.5) ($p < 0.01$) (Table 3.2).

Table 3.2. Mean number of *P. afra* cuttings receiving scores in response to three different watering regimes (Mean ± SD) over all four seasons of the year (n = 120). Test statistics used: Factorial ANOVA. $F(6,18) = [47.39]$, $p < 0.01$)

Score	1	1.5	2	2.5	3	3.5	4
Control	58.0 ± 19.5	69.8 ± 8.2	101.8 ± 16.2	7.5 ± 5.8	5.8 ± 6.2	23.3 ± 14.0	4.0 ± 4.7
Monthly	34.8 ± 12.7	47.8 ± 3.3	127.3 ± 11.7	14.0 ± 10.6	4.3 ± 4.3	41.3 ± 31.8	0.8 ± 1.5
Bi-weekly	17.8 ± 6.5	55.3 ± 7.0	118.0 ± 14.4	14.0 ± 6.6	12.0 ± 8.6	52.0 ± 19.9	1.0 ± 1.2
Weekly	27.0 ± 14.9	76.3 ± 17.7	122.5 ± 3.7	14.0 ± 11.2	9.8 ± 11.8	20.5 ± 16.4	0.0 ± 0.0

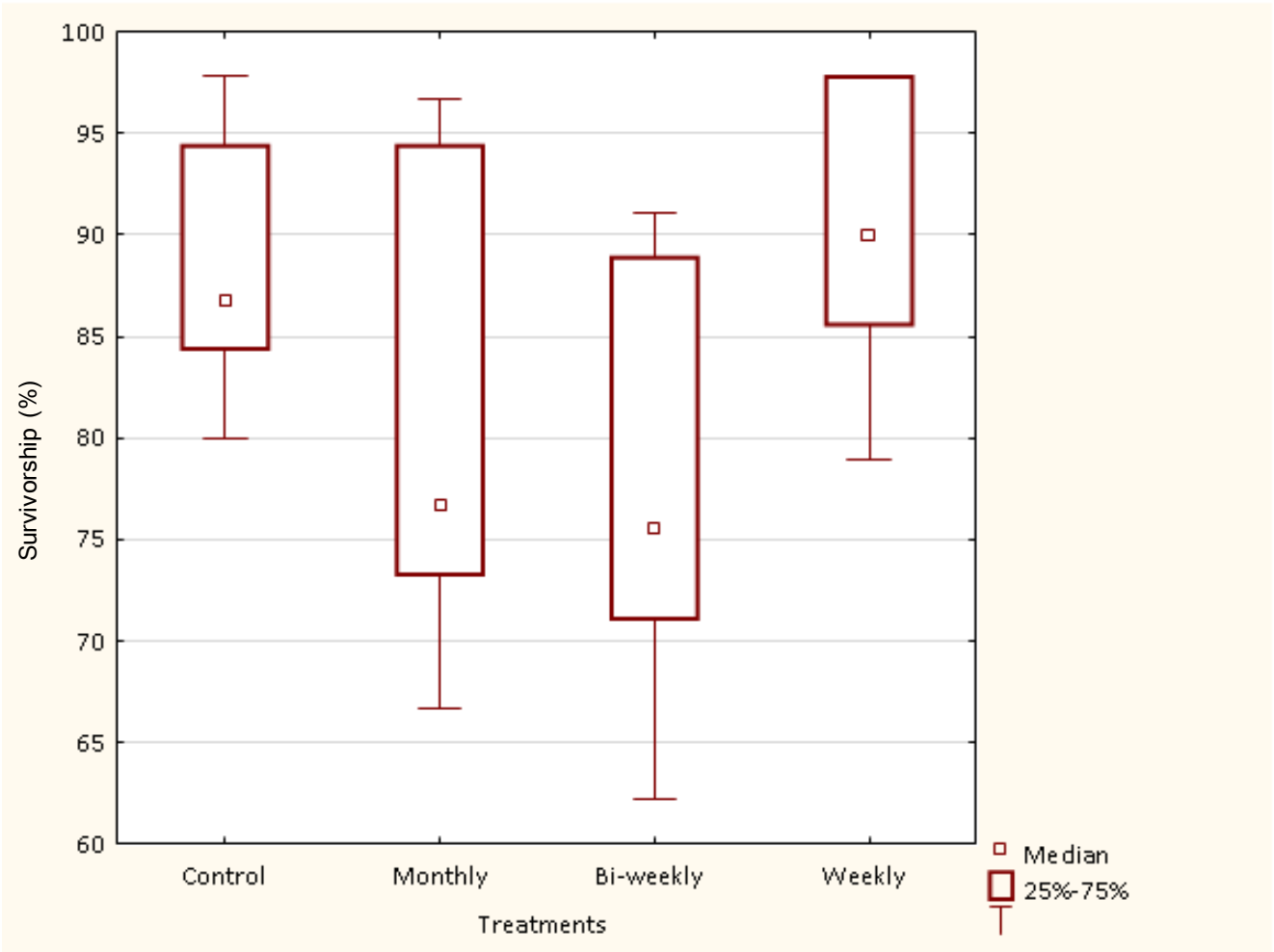

Figure 3.2. Mean survivorship of *P. afra* under different watering regimes.

3.5.1.2. Health observations over time

A repeated measures ANOVA revealed significant effects of the different treatments on survivorship ($p < 0.01$), namely the cuttings bi-weekly watering regime treatment produced significantly sharper fluctuations in health compared to the cuttings under the control and weekly treatments (Figure 3.3).

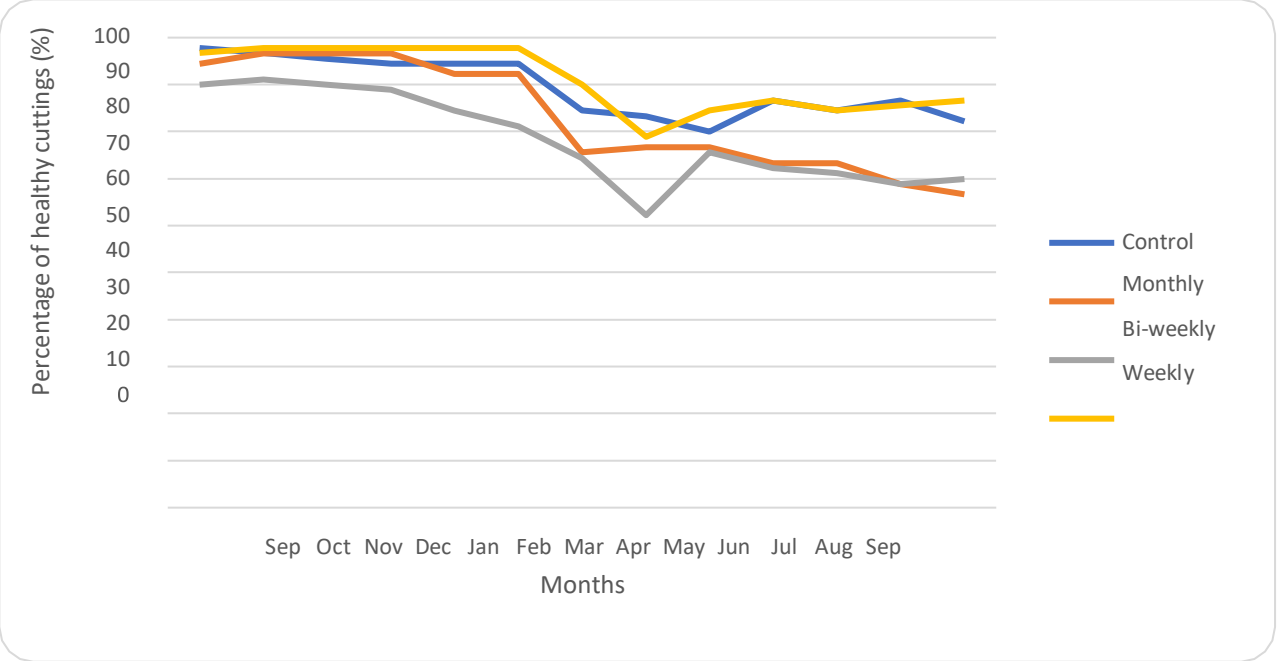

Figure 3.3. Percentage of healthy *P. afra* cuttings (scores 1-3) under three watering treatments over time (September 2018-September 2019).

3.5.2. Brushpack vs no brushpack

The factorial ANOVA found significant effects of brushpacking on survivorship ($F(1,3) = [15.46]$, $p < 0.01$). A Tukey HSD was performed, revealing significant differences between brushpacking and no brushpacking ($p = 0.05$, 95% C.I., [-0.14, 14.02]). Furthermore, there were significant differences between spring and autumn ($p < 0.01$), summer and spring ($p < 0.01$), winter and spring ($p < 0.01$) and winter and summer ($p < 0.01$). The pondings without brushpacking had a significantly higher mean survivorship (89.3 ± 6.3%) compared to the pondings with brushpacking (83.1 ± 11.7%) (Figure 3.4).

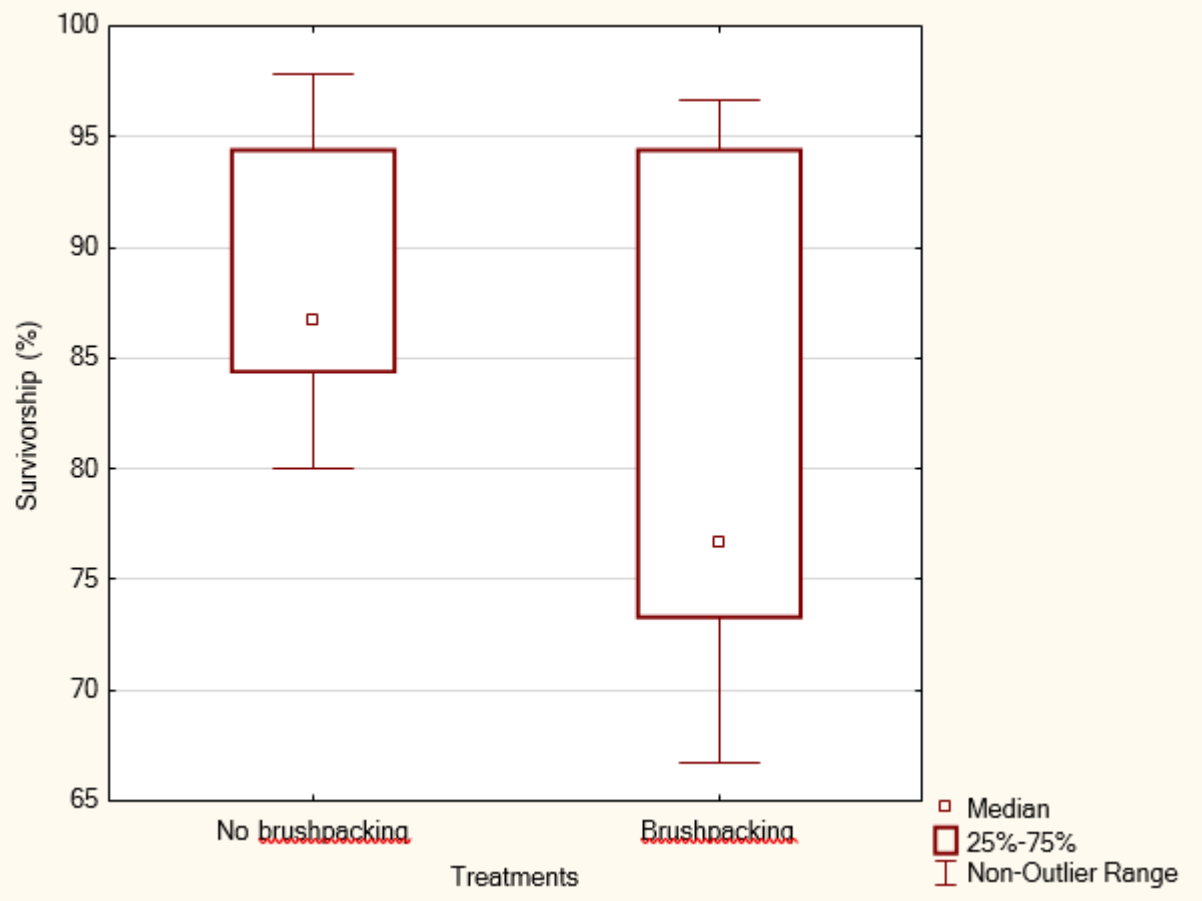

Figure 3.4. Mean survivorship of *P. afra* planted in pondings with brushpacking vs no brushpacking.

3.5.3. Nurse & Companion Planting

3.5.3.1. Survivorship

A Factorial ANOVA performed for this experiment revealed that treatments had a significant effect on survivorship (F (2,3) = [7.760], 0.002) Furthermore, a Tukey HSD revealed that survivorship in the control (89.3 ± 6.3%) was significantly lower than that in the companion plant treatment (94.9 ± 3.6%) (p = 0.04, 95% CI, [-10.91, -0.26] 0.03) (Table 3.3). The nurse planting treatment also had a significantly lower survivorship (87.5 ± 6.3%) compared to the companion planting treatment, (p =0.005, 95% CI, [-12.69, -2.03]) (Figure 3.5). However, there were no significant differences between the nurse planting treatment and the control not significant (p =

0.7, 95% CI, [-7.10, 3.55]). Furthermore, it was found that the seasons of the year were not significantly different from each other, nor did they have a significant effect on survivorship ($p > 0.05$).

Table 3.3. Descriptive statistics for the nurse and companion plant treatment

Treatments	Mean survivorship (%)	Standard deviation
Control	89.3	6.3
Nurse planting	94.9	3.6
Companion planting	87.5	6.3

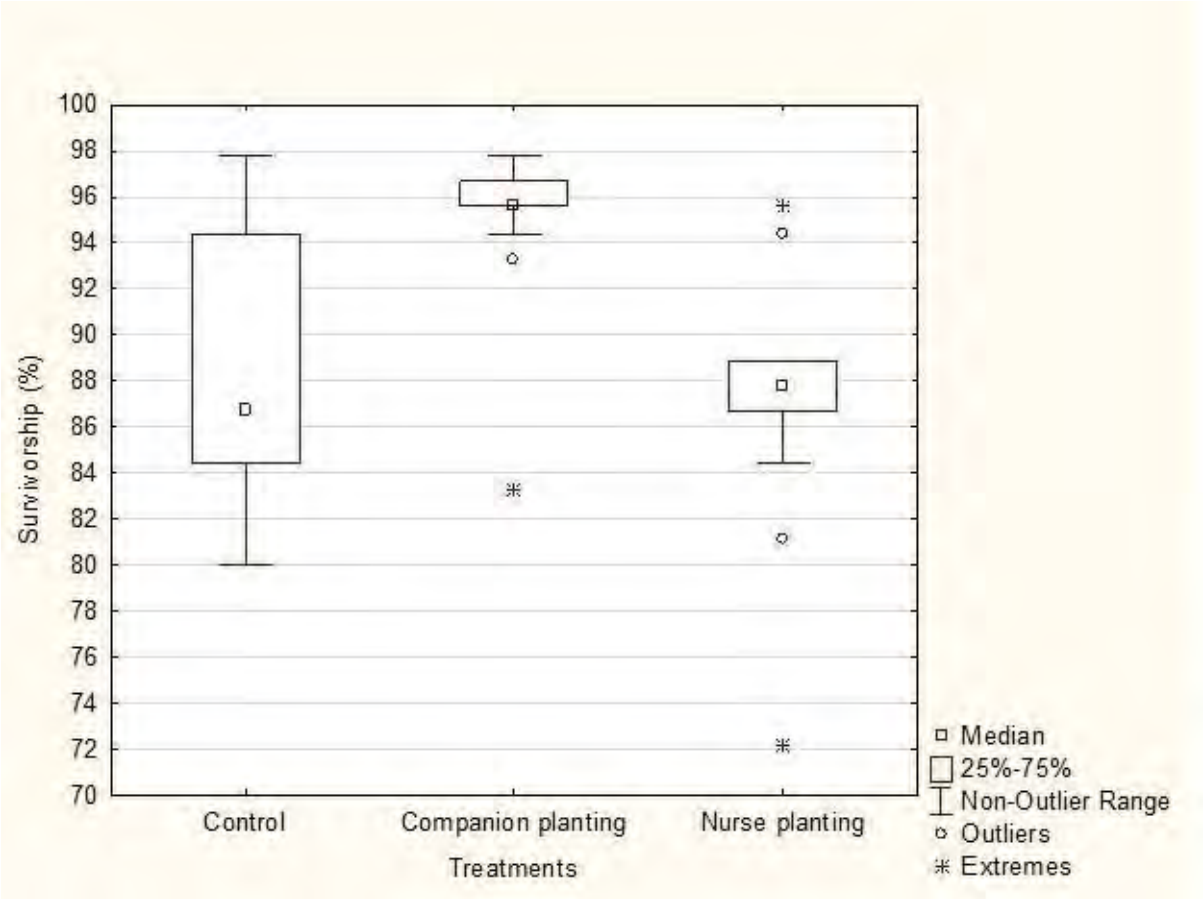

Figure 3.5. Mean survivorship of *P. afra* planted alongside companion and nurse species.

When examining the health scores of *P. afra* cuttings across different treatments, a factorial ANOVA revealed that the companion planting treatment had significantly higher scores of healthy *P. afra* cuttings ($p < 0.01$) (63.5 ± 22.9) compared to nurse (58.5 ± 11.3) and control (58 ± 19.5) (Table 3.4).

Table 3.4: Mean number *P. afra* cuttings per health score planted alongside nurse and companion plants (Mean ± SD) over all four seasons on the year (n = 270). (Test statistic used: Factorial ANOVA. F (6,12) = [155.8], p < 0.01)

Score	1	1.5	2	2.5	3	3.5	4
Control	58.0 ± 19.5	69.8 ± 8.2	101.8 ± 16.2	7.5 ± 5.8	5.8 ± 6.2	23.3 ± 14.0	4.0 ± 4.7
Companion	63.5 ± 22.9	70.3 ± 6.4	113.0 ± 25.6	6.3 ± 4.6	6.5 ± 2.6	10.5 ± 1.3	0.0 ± 0.0
Nurse	58.5 ± 11.3	84.3 ± 32.8	97.0 ± 35.8	8.3 ± 7.1	8.5 ± 8.7	13.5 ± 11.0	0.0 ± 0.0

3.5.3.2. Health observations over time

A repeated measures ANOVA found no significant differences in survivorship between treatments (p = 0.15); however, the control was observed to have a steady decline in growing health, the nurse plants having the most fluctuation and the companion plant displaying a more stable growth trend (Figure 3.6).

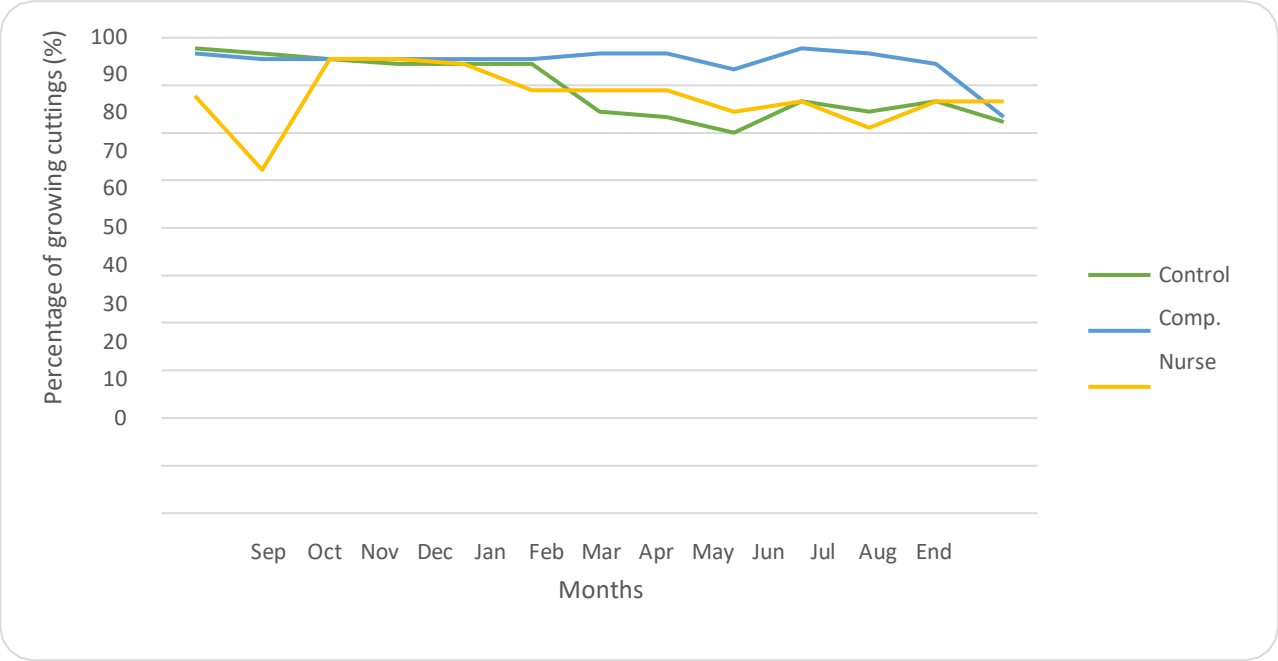

Figure 3.6. Percentage of growing *P. afra* cuttings (Scores 1-3) under nurse and companion planting treatments over time

3.5.3.3. Companion species

By the end of the experiment, none of the woody companion species survived, however the *A. ferox* had varying degrees of survivorship in this treatment. A mortality of 40% was recorded for the succulent while 26.7% were under stress and the remaining 33.3% were alive and in good health (Figure 3.7).

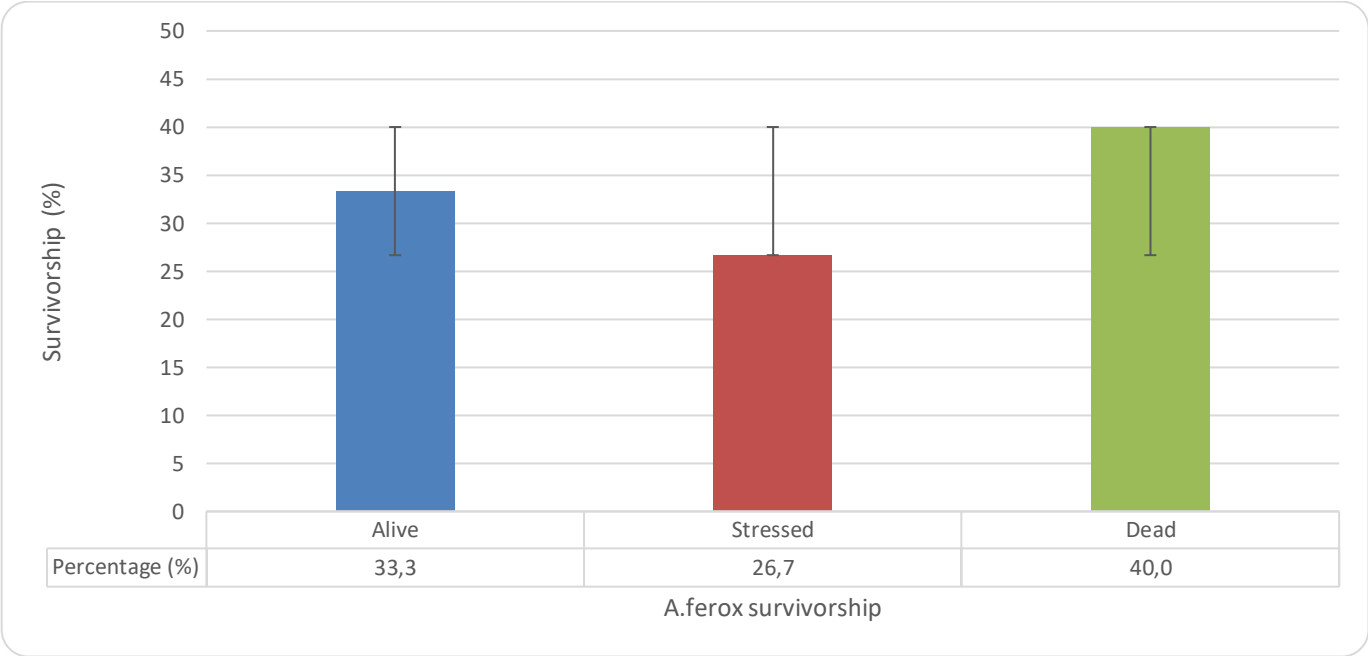

Figure 3.7. Overall mean survivorship of *A. ferox* planted alongside *P. afra*

3.6. Discussion

3.6.1. Watering Regimes

3.6.1.1. Survivorship

The weekly watering treatments had the highest mean survivorship out of the watering regimes (90.9 ± 1.9%), due to the *P. afra* being watered weekly after planting (June-September) (Table 3.1). The higher frequency in water inputs compared to the other watering treatments aided in their establishment and growth (Ludwig *et al*. 2005). However, it had a lower score for peak health (27 ± 14.9) compared to the control (58.0 ± 19.5) (Table 3.2), suggesting that overwatering could have been a factor (McDowell, 1987).

The control had a higher survivorship (89.3 ± 6.3%) than expected (Table 3.1), as it was assumed that, since there was no brush-packing, the lack of extra protection from herbivores and the elements would negatively affect its survivorship, as was observed by Mills & Robson (2017) when the mean survivorship of *P. afra* was no more than 30%. The control also had a higher score of *P. afra* cuttings in peak health(Score 1) (58.0 ± 19.5.) (Table 3.2). A possible explanation could be that the game fencing around the experimental plots greatly limited access to the *P. afra*, despite the signs (spoor) and sightings of kudu entering and traversing the plots, similar to other observations in the Baviaanskloof (Powell, 2009; Berriman, 2010; Starkey, 2015).

The monthly watering treatment was expected to perform better than the control, owing to the brush-packing, however, it displayed a lower survivorship (83.1 ± 11.7%) and lower peak health scores (34.8 ± 12.7). A possible explanation would be that the brush-packing may have shaded out the *P. afra*, thus inhibiting photosynthesis to an extent. The bi-weekly watering had the lowest survivorship (78.5 ± 9.7%) and the lowest scores for peak health (17.8 ± 6.5) despite the higher water inputs to the *P. afra* (compared to the monthly treatment) during establishment. While it is possible for xeric-adapted plants such as *P. afra* to display lower performance and survival with increased water inputs (McDowell, 1987), it still doesn't explain why the weekly watering treatment had a higher survivorship. This suggests that the weekly treatment was significantly different to the biweekly, monthly and control treatments while the monthly, biweekly and control treatments were not significantly different to each other.

While the protocols have resulted in over 50% in *P. afra* survivorship, the only real downside is that using said protocols is time-consuming and expensive. Upon doing a preliminary estimate on the costs of restoration of thicket with less than 25% of its original biomass remaining, Mills *et al*., (2007) found that restoring thicket with *P. afra* alone (using the STRP protocol) would cost approximately ZAR12 668.36 ha^{-1}. Furthermore, it would cost approximately ZAR15 124.82 ha^{-1} to restore said thicket using *P. afra* along with a variety of thicket species (Mills *et al*. 2007), as was done in the case of the companion plant and nurse plant treatments. If one adds the construction of pondings the use of nursery-grown rooted cuttings as opposed to unrooted truncheons and supplemental watering, that means further investment in time and resources will be required (Mills *et al*. 2015).

Nevertheless, using this protocol is a step forward in not only enhancing the restoration process but also the cost-effectiveness thereof. Using pondings has been found to be more effective compared to using narrow planting holes in harvesting rainwater, thereby promoting increased growth and survival of *P. afra* cuttings planted within said pondings (Powell 2009, Mills & Robson, 2017). Furthermore, at the monthly cost of ZAR133.75, watering *P. afra* planted in pondings placed 20 m apart is almost a tenth of the monthly cost of watering individual *P. afra* truncheons planted 2 m apart (ZAR1,269.50) (Mills *et al*. 2018). Ultimately, the benefits of planting clumps of rooted *P. afra* cuttings in pondings and watering them thereafter far outweigh the costs in the long term.

3.6.1.2. Health observations over time

The *P. afra* cuttings in the bi-weekly experienced the sharpest decline out of all the treatments from the spring to the winter months and made the slowest recovery, followed by the monthly, weekly and control, likely because they were overwatered in comparison to the monthly treatment and control (Figure 3.3). It is often stated that rooted *P. afra* cuttings are vulnerable to mortality should they be subjected to overwatering as this encourages the associated fungal pathogens to form (McDowell, 1987). It is unclear, however, as to why the bi-weekly percentage was higher than that of the weekly treatment despite the latter having been watered more often. Another likely theory is that during the autumn months daylight was shortening and the *P. afra* cuttings had to switch to C_3 photosynthesis to cope with the shorter photoperiods (Guralnick *et al*. 1984b). These findings run parallel to those of Ragimana & Potts (2019) that *P. afra* planted vertically experienced greater stress (as reflected by their lower photosynthetic efficiency values) compared to *P. afra* planted horizontally.

It is also unclear why the plants in the monthly treatment experienced a sharper decline compared to those in the control given that they were provided with the same amount and frequency of watering during the establishment phase (Figure 3.3). However, this could be a result of the brushpacking in the monthly treatment serving as somewhat of an obstacle to the plants receiving more sunlight (Ren *et al*. 2008).

The decline observed in the control pondings during the seasons was more staggered, despite being most exposed to environmental stresses (heat, herbivory and

evapotranspiration) as they lacked brushpacking (Hanke *et al*. 2011). A possible theory could be that since the control received no brushpacking and only received monthly watering during establishment, the plants in the control were more exposed to environmental stresses and were thus hardened by them. Indeed, Vitale et al., (2020) found that the CAM photosynthetic pathways help alleviate oxidative stress in *P. afra*, especially when subjected to water stress. However, by the end of the experiment, as the season transitions into summer, the overall vitality of the plants in the control starts to decline once again due to the onset of the harsh summer heat (Figure 3.3).

The weekly treatment had a similar decline to the control but a higher degree of recovery as the seasons progressed. This is likely because the plants received the most frequent watering during the establishment phase. Therefore, the plants in the weekly treatment were provided with the most resources to store and utilize when they switch from CAM to C_3 during winter where the photoperiods are shorter (Guralnick *et al*. 1984a; Guralnick *et al*. 1984b).

3.6.2. Brushpacking vs no brushpacking

The control had a significantly higher survivorship compared to the brushpacked pondings even though it was afforded the least protection (no brushpacking) (Figure 3.4). This is likely because the lack of brushpacking meant that the *P. afra* cuttings received the most direct sunlight compared to the rest of the treatments which were brushpacked. This would suggest that the plants in the control were better able to photosynthesize during the shorter winter days as there were less obstacles between the cuttings and resources (namely sunlight) (Ren *et al*. 2008).

3.6.3. Nurse and companion plant treatments

3.6.3.1. Survivorship

The companion plant treatment contributed to producing the highest *P. afra* survivorship (94.9 ± 3.6%) (Table 3.3) possibly due to facilitation from companion plants (Navarro-Cano *et al*. 2019) which aided in *P. afra* establishment (Figure 3.5). This treatment also displayed the highest number of *P. afra* in peak health (63.5 ± 22.9) (Table 3.4). These results can be compared to the observations of Midega

et al. (2015) whereby they observed the difference in two plots: one with a maize monocrop and the other plot being a maize plot grown with climate-adapted companion species. In this experiment the companion-planted plot showed significant reduction in pest damage and a significant increase in maize plant height and grain yield compared to observations of the monocrop plot (Midega *et al*. 2015).

The Control had a high survivorship (89.3 ± 6.3%) (Table 3.3) and peak health score (58.0 ± 19.5) (Table 3.4) despite having no brush-packing, companion plants or a nurse canopy. It was assumed that the absence of facilitation (from the companion/nurse plants) and the lack of extra protection from herbivores and the elements would negatively affect *P. afra* survivorship.

The nurse plant treatment performed well in terms of survivorship (87.5 ± 6.3%) because nurse plants can limit herbivory and ameliorate the climate, creating a microclimate suitable for *P. afra* establishment and growth. These benefits could have possibly been enhanced by the functional distance of *C. haematocarpa*, a woody shrub, from *P. afra*, a succulent, thus facilitating a more beneficial interaction between the two different plant types (Navarro-Cano *et al*. 2019). The inclusion of post-planting watering also further enhanced the survivorship of the cuttings. These findings are similar to those by Badano *et al*., (2009) who, when planting *Quercus castanea* seedlings in Mexico's degraded oak forests, found that seedlings planted under a nurse canopy and seedlings planted outside a nurse canopy both had a survivorship of less than 20%. However, when water inputs were included, the survivorship of the seedlings under the nurse planted increased up to 58% while that of the seedlings outside a nurse canopy remained at under 20%.

On the other hand, this treatment's survivorship (87.5 ± 6.3%) was outperformed by both the companion plant treatment (94.9 ± 3.6%) and the Control (89,3 ± 6.3%). It is possible that the nurse plants may have negatively influenced the growth of *P. afra* underneath the canopy, thus undermining the number of cuttings growing at peak health (58.5 ± 11.3) (Table 3.6). An observation by Schöb *et al*., (2013) found that the higher the biomass of the beneficiary plant, the higher the level of facilitation. Conversely, the lower the biomass of the beneficiary plant, the higher the level of competition. This could explain why the nurse plant treatment did not show the highest levels of *P. afra* survivorship as expected since the small *P. afra* cuttings were competing with the much larger nurse canopies for resources and root space (Ren *et al*. 2008).

Nevertheless, the benefits of nurse plants for restoration in an arid environment far outweigh the disadvantages as nurse plants (and even nurse objects such as log piles, rocks, etc.) can help the target species (in this case, *P. afra*) establish successfully, thus accelerating the restoration process and promoting biodiversity in harsh environments (Ren *et al*. 2008; Navarro-Cano *et al*. 2016; Oreja *et al*. 2020). A case in point is the finding that *P. afra* planted in more" moderately degraded" thicket sites characterized by a retention of canopy dominants, woody vegetation cover and litter cover had a higher survivorship (48.2 ± 22.9 %) than the *P. afra* planted in more severely degraded sites (35.1 ± 29.1%) (Sholto-Douglas, 2019).

3.6.3.2. Health observations over time

The nurse plant treatment dropped significantly in overall vitality during the first two months of monitoring (Figure 3.6), possibly due to the advent of spring facilitating greater growth and productivity in the nurse plants and therefore increasing competition between the nurse and beneficiary plants (Ren *et al*. 2008). However, the cuttings under the companion plant treatment did not drop in overall health during the spring months, likely due to the facilitative properties that exist in companion species, namely the *A. ferox* that survived. Furthermore, the climate during the spring months in the Baviaanskloof region is mild compared to that of the summer and winter months (Boshoff *et al*. 2000; Hattingh, 2011; Glenday, 2015) and thus the lack of extreme environmental stress meant that the *P. afra* cuttings would thrive.

The increase in overall vitality from November is possibly a result of the nurse plant ameliorating the micro-climate in which the *P. afra* cuttings were sheltered from the intense summer heat (Figure 3.6) (Padilla & Pugnaire, 2006). This protection the *P. afra* received during the harsh summer months from the nurse plants likely mitigated the environmental stress of summer (Gómez-Aparicio *et al*. 2004). The gradual decrease in condition over time among the cuttings under the nurse plant treatments after the summer months may be a result of excessive shading from, and competition for resources between, the nurse plants and the beneficiary plants (Holmgren *et al*. 1997; O'Brien *et al*. 2017). This theory of *P. afra* cuttings competing with nurse plants for dwindling resources is further supported by Holmgren *et al*., (1997) who state that a reduction in resources can shift the interaction between nurse and beneficiary plants from facilitative to competitive.

The companion plant treatment did not have as sharp a decline in the initial data collection process as the nurse plant treatment (Figure 3.6). This is likely because the facilitation mechanism of the companion plants helped mitigate the environmental stresses of the summer months (Padilla & Pugnaire, 2006). The overall vitality of the *P. afra* cuttings remained above 90% until gradually declining to 83.3% by the end of the study as the summer heat intensified. However, overall vitality of the *P. afra* cuttings gradually decreased during the autumn and winter months, likely due to competition between the cuttings and companion species asthe seasons progressed and resources dwindled (Holmgren *et al*. 1997).

The control experienced a steeper decline in overall vitality compared to the companion plants, ending up at 80% by May (Figure 3.6). This is very likely because the absence of a nurse or companion plant meant that the *P. afra* cuttings in the control were exposed to the elements with no mitigating factor compared to the other treatments (Hanke *et al*. 2011). However, the recovery to 86.7% in June and ultimately 82.2% by the end of the experiment (Figure 3.6) suggests that the cuttings had acclimated to the harsh climate.

3.6.3.3. Overall survivorship of companion plants

The total mortality of woody species could be because the lack the ability to switch between the CAM and C_3 photosynthetic pathways, unlike *P. afra*, therefore transplants were going to experience high levels of mortality during transplantation (Figure 3.7) (Louw, 2012). The mortality rate of these woody species was far higher than was previously observed in the study conducted by van der Vyver *et al*., (2012), where survival of *L. ferocissimum* was 19% and that of *R. obovatum* was 70%. However, that study involved planting no less than 150 propagules of both *L. ferocissimum* and *R. obovatum* and survivorship data was gathered for up to 24 months after planting.

The *A. ferox* showed varying degrees of survivorship (Figure 3.7) and this couldlikely be a result of a multitude of factors. Waterlogging from the pondings collecting water during rainfall events could have caused root rot on the *A. ferox* plants. Furthermore, the succulent is highly susceptible to a wide range of diseases such as *Aloe* cancer, *Aloe* scale and infestation of ants and aphids (Bhaludra *et al*. 2013). *Aloe* cancer is a

viral infection spread by mites, causing growths to form on the plant. *Aloe* scale is a white scale found on *Aloe* leaves and if the infestation becomes severe, it is usually fatal for the host plant.

Ants and aphids can also cause severe damage to the plant when together (Bhaludra *et al*. 2013). The variation in survivorship could be a result of one of, or a combination of, these factors affecting its health. Another possible theory is that, in addition to *A. ferox* being highly susceptible to diseases, it is also a succulent much like *P. afra*, thus making these two plants less functionally distant to one another (Navarro-Cano *et al*. 2019) and this could have resulted in the more resistant *P. afra* outcompeting *A. ferox*.

While the costs and benefits of using these protocols have already been discussed earlier in the chapter (3.7.1- pg. 71), further research is required to determine whether other thicket canopy species would serve as a suitable nurse plant as wellas to determine xeric thicket species most suited to be used as a companion plantfor *P. afra.*

Chapter 4. EXAMINING THE USE OF A MODIFIED PLANTING PROTOCOL TO INCREASE SURVIVORSHIP OF UNROOTED *PORTULACARIA AFRA* TRUNCHEONS IN DEGRADED ARID THICKET

4.5. Introduction

Since 2003, the Subtropical Thicket Restoration Programme (STRP) have established and followed a regional planting protocol for *Portulacaria afra* in hopes of successfully restoring vast areas of the xeric thickets which has undergone over 150 years of degradation, mainly through overbrowsing (Stuart-Hill, 1992; Mills *et al*. 2005b). This planting protocol involves the planting of unrooted *P. afra* truncheons (~30 cm in length; 20 mm in diameter) in 2x2 m rows (Lagerwall, 2010; Mills *et al*., 2015). These truncheons were planted in narrow, shallow holes made with augers or an iron bar locally referred to as a "koevoet' (Mills *et al*. 2015).

However, the protocols have proven to be less than feasible under the harsh conditions in which the *P. afra* truncheons are planted. Authors have stated that on average, ~ 30% of unrooted truncheons survive after being planted in degraded thicket sites (Lagerwall, 2010; Mills & Robson, 2017). The low survivorship is attributed to biophysical factors including capped soil, high soil temperatures and low soil moisture content and lower soil water potential, conditions that are highly stressful to the truncheons (Powell, 2009). Furthermore, plants getting uprooted or pushed over by animals (particularly herbivores) also contributes significantly to plant mortality (Powell 2009; Lagerwall, 2010). Cases of the existing planting protocol resulting in higher *P. afra* survivorship, while relatively few and far between, are present. Lagerwall (2010) examined the cost-effectiveness of restoration in northern AENP using the STRP planting protocol. In this study it was found that surrounding thicket vegetation in which the *P. afra* truncheons had a significant effect ($p = 0.029$) on *P. afra* survivorship. Illustrating thisfinding was that *P. afra* truncheons had a higher mean survivorship where the surrounding vegetation was Spekboom Thicket (43.5%) compared to Pentziaveld (31%) and Noorsveld (15.14%) (Lagerwall, 2010).

4.1.1. Scarification

The scarification of *P. afra* has been proven to be successful in generating roots from other parts of the stem (Norman, 2016). Scarifying the stems exposes sites of growth known as epicormic buds (Harmer, 1998). Planting scarified truncheons at least 2 months before winter, when temperatures are still relatively warm, was observed to produce a higher mean root biomass than if the truncheons were simply planted in the soil medium (Norman, 2016). Since unrooted *P. afra* truncheons can form roots in scarified parts of the stem, thus increasing their vigour and their chances of survival, it begs the question whether *P. afra* can exhibit a similar response when pruned (to simulate herbivory).

4.1.2. Herbivory simulation

As previously stated in Chapter 1, established *P. afra* responds well to top-down browsing by indigenous herbivores (Stuart-Hill, 1992) through extending its lower branches and forming a protective 'skirt' on the soil (Sigwela *et al*. 2009). If this response can be stimulated on unrooted truncheons by cutting off the apical meristem (thus simulating natural herbivory), then there is potential to 'prime' *P. afra* truncheons for successful establishment and stimulation of accelerated growth of the 'skirt' along with its biophysical benefits (Adie & Yeaton, 2014; Lechmere-Oertel *et al*. 2008; Sigwela *et al.* 2009).

4.6. Methods

This experiment is a 2 x 2 factorial experiment that consists of two factors, namely scarification and herbivory simulation. The factors were stratified as follows:

Truncheon 1: Scarification (S)

Truncheon 2: Herbivory simulation x Scarification (H/S)

Truncheon 3: Herbivory simulation (H)

Truncheon 4: Control (C) (no scarification and no herbivory simulation)

To implement this experiment, 30 pondings were excavated. Thereafter, 120 unrooted truncheons were sourced in the valleys near the study area for logistical ease as well as ensuring that the *P. afra* used is a local genotype. The truncheons, having been cut with gardening secateurs (at a 45-degree angle to allow for greater rooting area) and immediately planted in the soil measured at ~30 cm in length and ~20 mm in basal stem diameter. Four truncheons were selected for planting in each of the 30 pondings however, prior to planting, three of the four truncheons (one would serve as a control and thus be planted as is) would undergo the following treatments:

4.6.1. Scarification

The scarification treatment of *P. afra* stems was performed by nicking two small indentations (area:1 cm x 1 cm; Depth: ~0,5 cm) using garden secateurs. This was performed at two opposite sides of the stem along 10 cm from the bottom of the stem which was planted ~10cm deep.

4.6.2. Herbivory simulation

Using a pair of garden secateurs, the apical meristem of the *P. afra* truncheon (the top 10cm of the stem) would be cut off to simulate the top-down browsing done by indigenous large herbivores in a thicket landscape. The truncheons were then planted from left to right along the ponding according to the following order of combinations:

4.6.3. Plant health assessments

To determine the effects of varying watering regimes on *P. afra* vigour and survivorship, the health index created for this component study was used (see 3.2.5 – pg 53). After completing the health assessments, the scores were consolidated across the four seasons.

4.7. Data analysis

Data analysis was done using R Software and Microsoft Excel. All data were tested for normality and equality among variances using Shapiro-Wilk's test. Where it was found to not be the case, a Kruskal Wallis test was performed to test for statistical significance ($p > 0.05$) between the different treatments using R and Microsoft Excel.

For *P. afra* survivorship, a 2 x 2 Factorial ANOVA was performed to determine whether the different planting treatments have significantly influenced overall survivorship. To determine whether the different planting treatments have significantly influenced plant health over time, a repeated measures ANOVA was performed.

4.8. Results

4.8.1. Survivorship

A 2 x 2 Factorial ANOVA was performed for this experiment, revealing that the treatments did not have a significant effect significant on the survivorship of *P. afra* truncheons ($F(3,1) = [1.377]$ $p = 0.261$). Among the pre-planting applications between unrooted truncheons, the Control showed the highest survivorship (76.2 ± 17.6%), followed by the herbivory simulation treatment (74.9 ± 18.3%) (Table 4.1).

Table 4.1. Descriptive statistics of the unrooted truncheons treatments. (S = Scarification; H/S = Herbivory simulation + scarification; H = Herbivory simulation; C = Control).

Treatments	Mean	SD
S	72.1	19.3
H/S	70.8	20.8
H	74.9	18.3
C	76.2	17.6

The two lowest percentages of survivorship were observed in the scarification (72.1 ± 19.3%) and herbivory x scarification (70.8 ± 20.8%) applications respectively (Figure4.1).

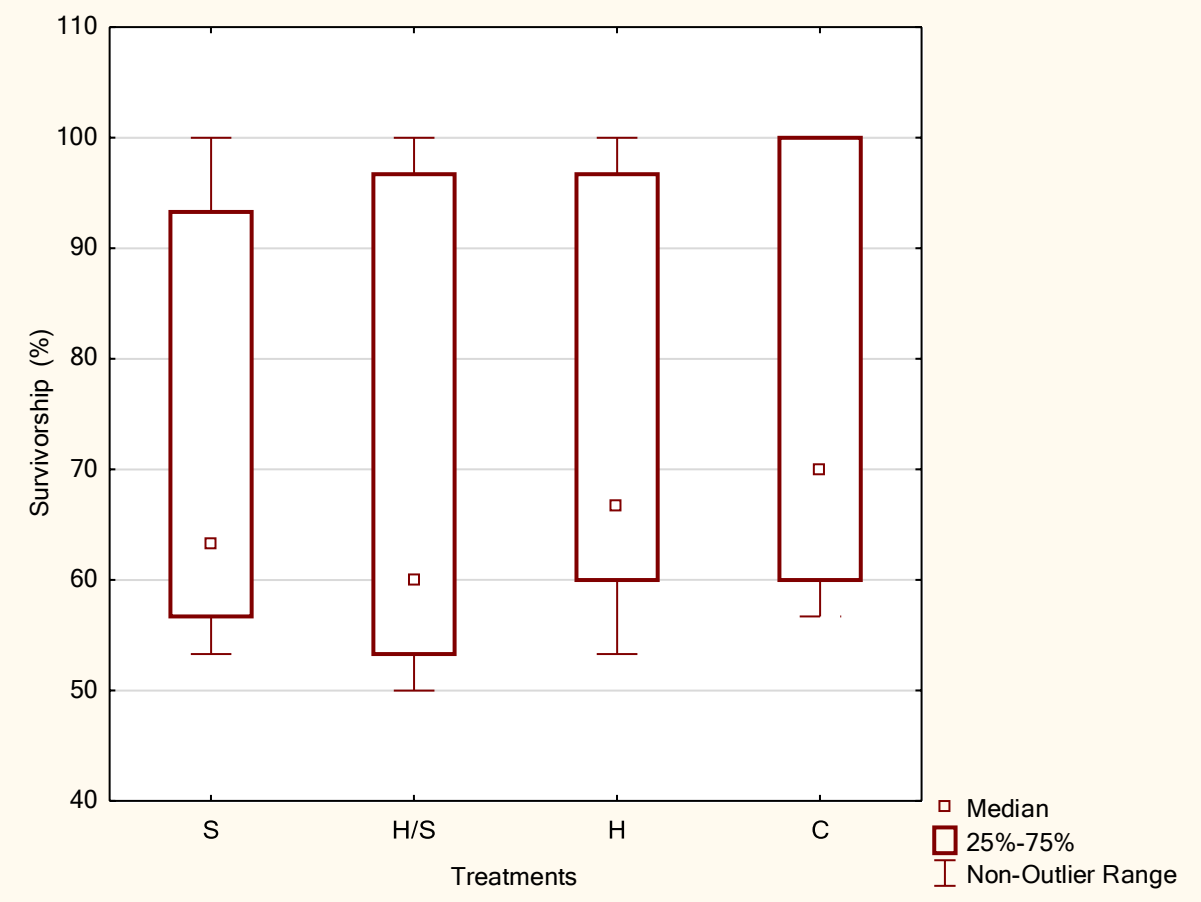

Figure 4.1. Mean survivorship of unrooted truncheons subjected to different treatments. (S = Scarification; H/S = Herbivory simulation + scarification; H = Herbivory simulation; C = Control).

When examining the number of *P. afra* truncheons that have scored for peak health (Score 1) (Table 4.2), a factorial ANOVA found that HS had the significantly highest number of P. *afra* truncheons in peak condition (8.3 ± 5.0) followed by the S1 (6.3 ± 5.4), H (4.0 ± 4.0) treatments and the Control (4.5 ± 6.1) ($p < 0.01$) (Table 4.3).

Table 4.2: Health scores for different unrooted truncheons treatments (Mean ± SD) over all four seasons of the year (n=120). Test statistic used: Factorial ANOVA. F (6,12) = [133.4], p<0.01)

Score	1	1.5	2	2.5	3	3.5	4
S1	6.3 ± 5.4	10.3 ± 3.8	43.3 ± 21.2	2.5 ± 1.3	4.0 ± 1.2	23.8 ± 18.0	0.0 ± 0.0
HS	8.3 ± 5.0	11.3 ± 2.8	39.3 ± 18.6	2.8 ± 2.9	3.5 ± 4.1	25.0 ± 18.8	0.0 ± 0.0
H	4.0 ± 4.0	15.0 ± 5.0	43.3 ± 24.3	4.5 ± 2.4	2.3 ± 2.6	21.0 ± 16.8	0.0 ± 0.0
C	4.5 ± 6.1	17.8 ± 1.9	41.0 ± 20.2	4.5 ± 3.3	2.5 ± 3.1	19.8 ± 15.9	0.0 ± 0.0

4.8.2. Health observations over time

A repeated measures ANOVA was performed, revealing no significant differences in treatments on health over time (F (3,1) = [1.377], p = 0.261), however all treatments show similar trends of health, initially declining after the first few months of establishment and slowly recovering (Figure 4.2).

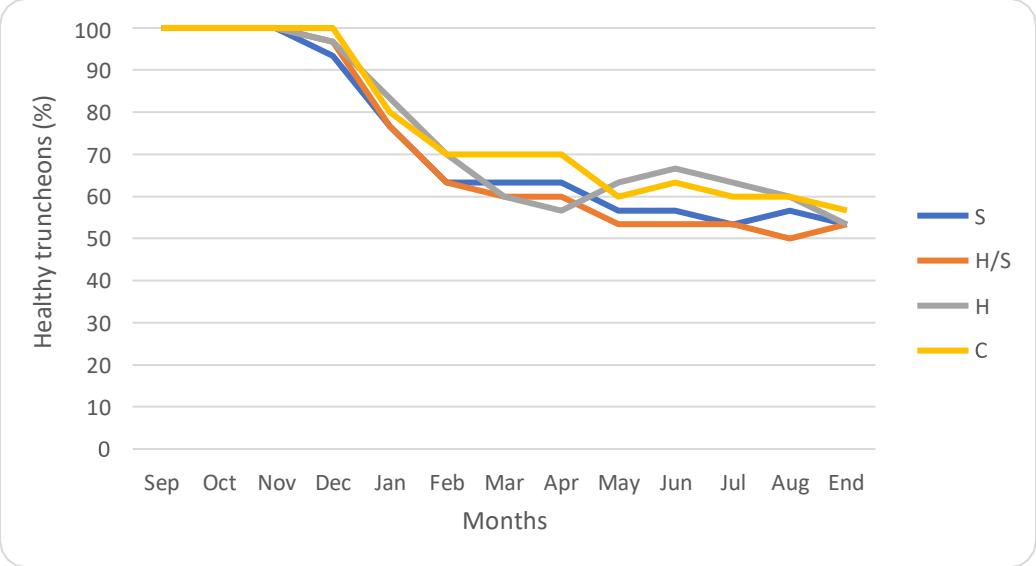

Figure 4.2. Percentage of healthy unrooted *P.afra* truncheons (Scores 1-3) under different treatments over time.

4.9. Discussion

4.9.1. Survivorship

The S treatment likely provided more sites of root establishment among the truncheons (Harmer, 1998) and while that entail a greater degree of stress compared to the treatments mentioned above, it still worked favourably for the establishment of the truncheons, as illustrated by its mean survivorship (72.1 ± 19.3%) (Table 4.1) (Figure 4.1) and mean number of truncheons scoring for peak health (6.3 ± 5.4) (Table 4.2)

Although the HS treatment had the highest mean number of truncheons in peak health (8.3 ± 5.0) (Table 4.2), it showed the lowest percentage of survivorship (70.8 ± 20.8.8%), (Table 4.1). likely because the compounded stress of scarification and apical meristem removal meant that more wounds through which water can be lost were created, thereby increasing the likelihood of the truncheon desiccating to death. Although Mills *et al*., (2018) noted that withholding water from nursery-grown truncheons over eight months resulted in a 50% mortality due to desiccation, it can be said that the same would apply on the field where the truncheons have more wounds through which water can be lost. This suggests that this treatment is more stressful to *P. afra* in comparison to the above applications on unrooted truncheons prior to planting (Figure 4.1).

The Control survivorship (76.2 ± 17.6%), likely outperformed all the other applications to truncheons because it underwent the least stress and could, therefore allocate more resources to establishing itself rather than having to use said recovery recourses for establishment thereafter (Figure 4.1), although it produced a lower mean of truncheons scoring for peak health (4.5 ± 6.1) (Table 4.2).

The H treatment was second to the control (74.9 ± 18.3%) (Figure 4.1) because while the truncheons under this treatment were stressed, they underwent a similar kind of a stress that further encourages growth in *P. afra*, namely top-down herbivory and the removal of the apical meristem (Vlok & Euston-Brown, 2002; Sigwela *et al*.

2009). However, it's likely that such stress contributed to this treatment having less truncheons in peak health (4.0 ± 4.0) compared to the control (Table 4.2).

4.9.2. Health observations over time

The S treatment stayed at 100% survivorship from September till November, until finally starting to dip from December (93.3%) and February (63.3%) (Figure 4.2). This dip is probably due to the environmental stresses of heat and drought coming into play during the summer months (Glenday, 2015). The overall vitality of the truncheons stays constant until April, finally dropping down to 53.3% by the end of the experiment.

The HS treatment went down a similar trajectory, also ending up at 53.3% by the end of the experiment. This is likely because the HS treatment went through both scarification of the stems as well as having the apical meristem cut off, causing the truncheons in this treatment to undergo double the stress (Louw, 2012; Mills *et al*. 2018) compared to the H and S treatments as well as the control.

The Control had a more gradual decline in health, only dropping to 80% from January and ultimately ending up at 56.7% by the end of the experiment. It makes sense that the control would have the highest percentage as it was subjected to the least initial stress prior to planting. As the HS and Control had already been mentioned above, it is worth noting that the H followed a similar trend, dropping steadily to 56.7%. It makes sense as to why the H treatment had a higher percentage compared to the HS treatment as it was subjected to less stress in comparison. The truncheons within the HS treatment experienced a steadier decline likely because the two pre-planting stresses took twice the toll on the truncheons and thus, they had to cope with the most initial stress (Louw, 2012; Mills *et al*. 2018),

A possible explanation as to why all the unrooted truncheon treatments had similar trend of declining overall vitality during the summer months could be due to the intense heat and drought that occur in the Baviaanskloof during that period, the increasing temperatures thus compounding drought stress (Boshoff *et al*. 2000; Hattingh, 2011; Louw, 2012; Glenday, 2015). Another possible explanation for the decline in overall vitality would be the length and diameter of the truncheons. It has

been argued that planting smaller, shorter truncheons as opposed to larger ones is a more ideal protocol. This is because smaller truncheons, once successfully established, have a greater root to shoot ratio compared to the larger truncheons (Ragimana & Potts, 2019), although this is contrary to the findings of Van der Vyver et al (2021), who found that larger truncheons (>22.5 mm diameter) had a higher survivorship (~45%) compared to smaller truncheons (<15 mm diameter) which had a survivorship of ~25%.

However, it is unclear why the H and C despite the truncheons in the former having been subjected to more stress than those in the latter, share the same percentage values. It is possible that the truncheons within the H, having recovered from the initial stress of having their apical meristem cut off (Stuart-Hill, 1992), were more resistant to stress.

Chapter 5. USING ECOPHYSIOLOGICAL DATA TO ASSESS THE INFLUENCE OF PLANTING INNOVATIONS ON RESTORATION SITES IN AN ARID THICKET LANDSCAPE

5.1 Introduction

Broad studies relating to plant biology have identified several functional traits that are good indicators of plant responses to water and nutrient availability and good indicators of plant defences against herbivory and plant effects on litter decomposition (Bernhardt-Römermann et al. 2008; Rusch et al. 2009; Carmona et al. 2011). Trait selection has been advocated to include ecophysiology as a starting point. However, the importance of ecophysiology is perceived to be largely overlooked by functional ecologists (Rosado et al. 2013), despite its usefulness as an additional metric for measuring restoration success by comparing ecophysiological traits between plants in restored and intact communities (Kimball et al. 2016).

Ecophysiology examines variables such as growth, reproduction, survival, abundance and geographical distribution of plants. These variables are focused on, because they are affected by the interactions between plants with their physical, chemical and biotic environment (Lambers *et al*. 2008). Ecophysiological knowledge is fundamental for functional ecology and this knowledge can help explain plant responses to key ecological drivers such as environmental stress and disturbances (Lambers *et al*. 2008).

Although the physiological responses of *P. afra* to drought and re-watering have been examined and documented (Guralnick & Ting, 1986), these studies were largely lab-based and focused on the plant's photosynthetic responses via Crassulacean acid metabolism (CAM) and C_3 (Ting & Hanscom 1977 Guralnick *et al*.1984a; Guralnick *et al*. 1984b; Guralnick & Gladsky, 2017). It is important to examine how a plant's photosystem (a biochemical mechanism through which photosynthesis is performed) (Ball *et al*.1994) changes in response to environmental changes.

Functioning of photosystem II (PSII) is the most sensitive indicator of environmental stress in plants. Changes in PSII activity can be measured rapidly and non-destructively by measurement of chlorophyll fluorescence.

Most studies have emphasised the fluorescence parameter Fv/Fm which is well correlated with the quantum efficiency of photosynthetic carbon dioxide assimilation or oxygen evolution in PSII (Baker, 2008). This parameter reveals information which can be attributed to diurnal and seasonal variation in photosynthesis, plant growth and plant community dynamics. Therefore, fluorescence measurement techniques are powerful tools with which to link photosynthesis with higher levels of plant functioning and have great potential for research in forest ecology (Ball *et al*. 1994).

Therefore, PSII is also accepted to be the most vulnerable part of the photosynthetic apparatus to light-induced damage. Damage to PSII will usually be the first manifestation of stress in a plant leaf (Maxwell & Johnson, 2000).

Moreover, the functioning and ultimately survival of plants depends on the availability of water in the soil, otherwise referred to as soil water potential in this particular context of research and land management. Soil water potential, the force that governs water movement in soil, is essentially the amount of energy required for a plant to extract moisture from the soil (Dane & Hopmans, 2002; Al-Kaisi *et al*. 2017). It is therefore of vital importance when considering soil and soil health as availability of water is crucial to plant survival and growth. Commonly measured as -kPa, the more negative the value, the lower the soil water potential and therefore the more work a plant needs to do in order to extract moisture from the soil (Dane & Hopmans, 2002). In the context of agriculture, it is generally accepted that a soil water potential value of 0 kPa is saturated soil; -33 kPa is field capacity (optimal soil water potential for plants) and -1500 kPa being the permanent wilting point at which the plant dies (Dane & Hopmans, 2002; Bitelli, 2010). Soil water potential is therefore useful for determining how much moisture is available in the soil for plants to utilize.

Soil temperature is another important environmental parameter that influences plant growth. While low soil temperatures (~5°C) have a negative effect on plant growthvia biomass reduction (Domisch et al. 2001), it has been reported that high soil temperatures (>30°C) also negatively influence plant growth by inhibiting photosynthesis and shoot growth (Arai-Sanoh *et al*. 2010).

The environmental parameters explained above play a crucial role in not only determining plant health and function, but also in making inferences about the climate and ecosystems as a whole. Therefore, the aim of this study was to examine how soil water potential, soil temperature and chlorophyll fluorescence respond to different restoration techniques in degraded thicket.

5.2 Methods

The study used ten replicates of pondings grouped under four different locations (a total of 40 pondings) within the field site. For these treatments, ecophysiological measurements were taken to determine whether the different planting innovations used in this study for *P. afra* have any influence on the ecophysiology of the succulent. These measurements were taken from October 29, 2019, until November, 7, 2019. The four ponding types used in the study were:

- Nurse canopy
- *V. karroo* canopy
- Open area
- Companion plants

For the ponding types, the measurements taken are discussed below.

5.2.1. Measuring soil water potential (kPa)

Soil water potential and temperature were measured across the *P. afra* sites that were mentioned above. It should be noted that soil water potential was used as an indicator of soil moisture content. The measurements were taken in the morning, midday and late afternoon over a period of 10 days. Both soil water potential and soil temperature were measured using Decagon MP6 soil moisture probes (± 1,5% accuracy). A probe was placed in a ponding at a depth of ~10 cm, to measure at root level. The probes were left in the soil for an hour to ensure accurate readings when the measurements were done.

5.2.2. Measuring soil temperature (°C)

The same methods as 5.2.1. were applied in measuring this variable. It should be noted that the measurements mentioned in 5.2.1 and 5.2.2. included ten replicates of unrooted *P. afra* truncheons that were planted in the field site without a ponding or brushpacking by Working for Water (WfW)- a subdivision of Working for Ecosystems (WfE)- but due to time constraints these truncheons could only be included in the soil water potential and soil temperature measurements.

5.2.3. Measuring chlorophyll fluorescence (Fv/Fm)

5.2.3.1 Assessing P. afra cuttings

Chlorophyll fluorescence was measured using a fluorometer (OS1p Modulated Fluorometer) on dark-adapted *P. afra* leaves. These measurements were taken during midday (ideally under sunny weather). Before conducting fluorometry measurements, fluorescence clips were inserted on a new green (healthy) leaf from one *P. afra* cutting per ponding to adapt the leaves to darkness. The chlorophyll fluorescence was measured an hour after the clips were inserted, whereby thecovering in each clip was removed. This was done so that the fluorometer's short, high intensity light flash could be applied directly to the leaf. Thereafter, the fluorometer console was used to measure the chlorophyll fluorescence of the leaf. The selected leaves were new, exposed to full sun and situated in a north-facing direction. These measurements were done during midday every day over a period of 10 days.

5.2.4.2. Assessing nurse plants

In addition to these four ponding types, four woody thicket species were selected as potential nurse species for *P. afra*. One individual of each species was selected and had their chlorophyll fluorescence (Fv/Fm) measured (as outlined in 5.2.4). These species were *E. undulata, C. haematocarpa, Grewia robusta* and *S. longispina*.

5.3 Data analysis

Data analysis was done using R Software and Microsoft Excel. All data were tested for normality and equality among variances using Shapiro-Wilk's test. Where it was found to not be the case, a Kruskal Wallis test was performed to test for statistical significance ($p > 0.05$) between the different treatments. A repeated measures ANOVA (followed by a Post-hoc Tukey HSD) was performed to determine whether the variables were significantly different across the different ponding types.

5.4 Results

The repeated measures ANOVA found that the soil water potential was significantly different among the different treatments ($F(2.4) = [8.43]$ $p = 0.0368$). A pairwise comparison also found that the morning values were significantly different to those of the late afternoon ($p < 0.01$) but not to those taken in the midday ($p > 0.01$)

A repeated measures ANOVA revealed significant differences in soil temperature values between the WfW planting sites and the pondings with companion plants ($F(2.2) = [29.93]$ $p = 0.02$). Again, pairwise comparisons revealed significant differences between the morning and late afternoon values only.

A repeated measures ANOVA showed that there is a significant difference in soil temperature values between the pondings in the open area, nurse canopy and *V. karroo* canopy ($F(2,4) = [67.06] = p < 0.01$). A pairwise comparison also showed that the values take during midday were significantly different to those taken in the morning ($p < 0.01$) and late afternoon ($p < 0.01$).

A repeated measures ANOVA showed that there is a significant difference in soil temperature values between the WfW planting sites and pondings with companion plants ($F(2,2) = [191.6]$ $p = 0.005$). A pairwise comparison also showed that the values take during late afternoon were significantly different to those taken in the values take during late afternoon were significantly different to those taken in the morning ($p = 0.002$) and midday ($p < 0.01$).

5.4.1. Mean Soil Water Potential

Morning

Out of all the treatments nurse canopy treatment had the lowest mean soil water potential in the morning (-44.5 ± 45.5 kPa), followed by the Open area pondings (-18.1 ± 5.8 kPa). The pondings under the *V. karroo* canopy had a soil water potential value of -16.1± 3.9 kPa (Figure 5.1). The WfW *P. afra* plantings had a soil water potential of -22.7 ± 5.9 kPa, lower than that of the companion plant treatment (-14.1 ± 1.7 kPa).

Midday

During the midday, the mean soil water potential of the pondings under the nurse canopy treatment remained the lowest (-160.9 ± 200.5 kPa) followed by that of the open area pondings (-73.4 ± 55.7 kPa) and the pondings under the *V. karroo* canopy (-56.5 ± 75.7 kPa) (Figure 5.1). The companion plant pondings had much lower mean soil water potential (-141.1 ± 192.6 kPa) compared to the WfW *P. afra* planting sites (-88.3 ± 37.3 kPa) (Figure 5.2).

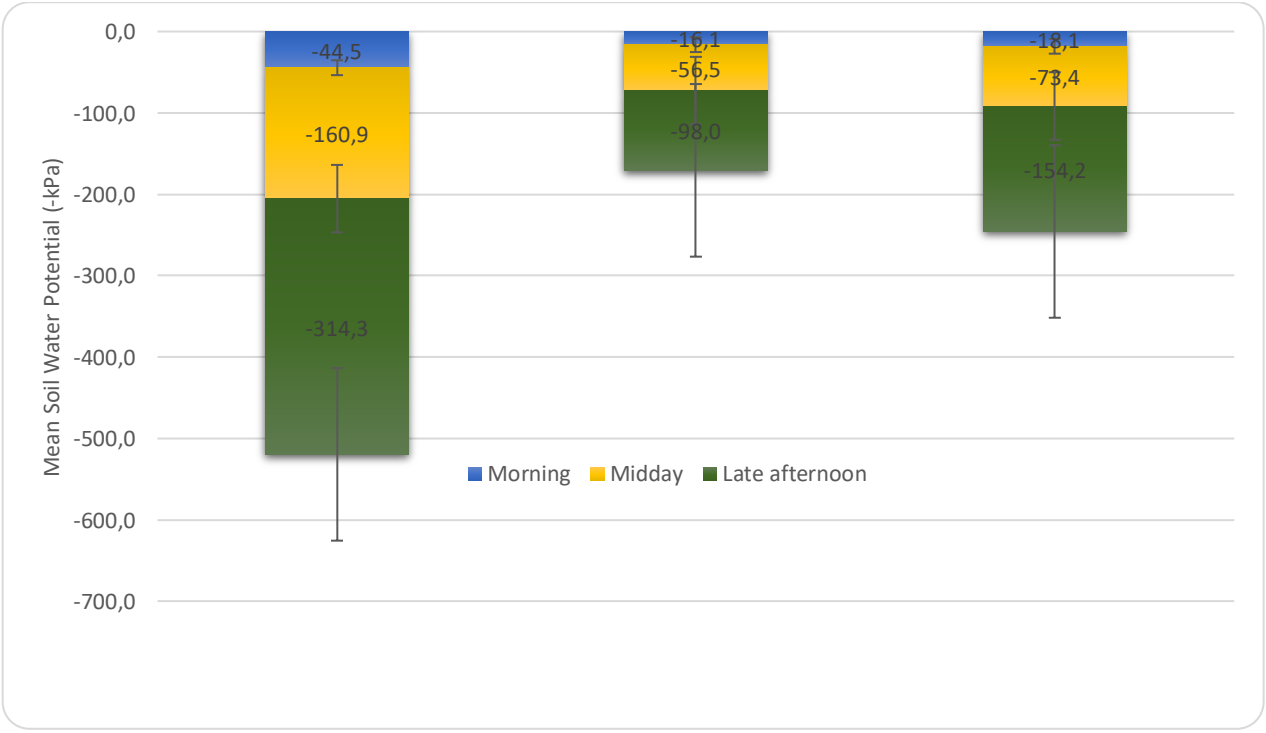

Figure 5.1. Mean soil water potential across pondings in Nurse canopy, *V. karroo* canopy and Open areas.

Late afternoon

In the late afternoon, the pondings under the nurse canopy had a very low mean soil water potential (-314.3 ± 423 kPa), lower than that of the pondings in the open area (-154.2 ± 133.6 kPa). The pondings under the *V. karroo* canopy displayed a higher mean soil water potential compared to those under the aforementioned two treatments (-98 ± 140.1 kPa) (Figure 5.1). The pondings with the companion plans displayed a lower mean soil water potential (-367.2 ± 607.4 kPa) in comparison to the WfW *P. afra* planting sites (-265 ± 149.1 kPa) (Figure 5.2).

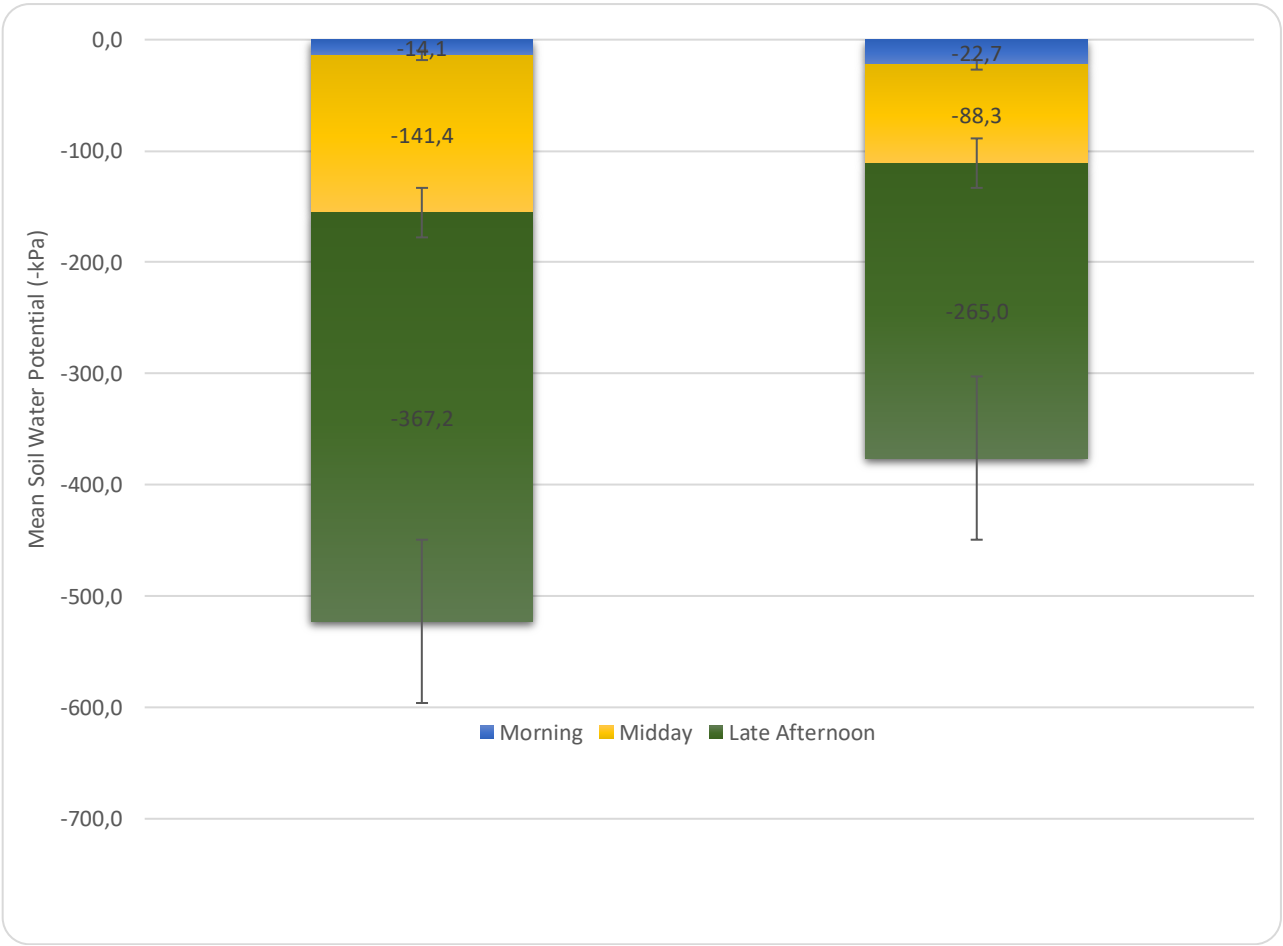

Figure 5.2. Mean soil water potential across *P. afra* planted with companion species and *P. afra* truncheons that were planted prior to the study.

5.4.2. Mean Soil Temperature

Morning

During the morning measurements, the nurse canopy had the lowest mean soil temperature (26 ± 5.6 °C) (Figure 5.3) while the highest mean soil temperature values were found in the WfW *P. afra* (36.1 ± 4.6°C), followed closely by the Companion plant treatments (34.5 ± 3.2°C) (Figure 5.4). Compared to the aforementioned treatments, the Open area treatment was observed to have a lower mean value (31.4 ± 6.3°C) (Figure 5.1). The second-lowest value was recorded in the *V. karroo* canopy (29.8 ± 4.4°C).

Midday

The Open area treatment had the highest mean soil temperatures during the midday (38.5 ± 2.7°C) (Figure 5.3), only slightly surpassing the values observed inthe WfW *P. afra* treatment (37.8 ± 3.7°C). The Companion plant treatment had a mean soil temperature value of (36.5 ± 2.8°C) (Figure 5.4) while the *V. karroo* treatment had a slightly lower value in comparison (35.2 ± 4.2°C). The Nurse canopy had the lowest mean soil temperature out of all the treatments (31.4 ± 5.2°C).

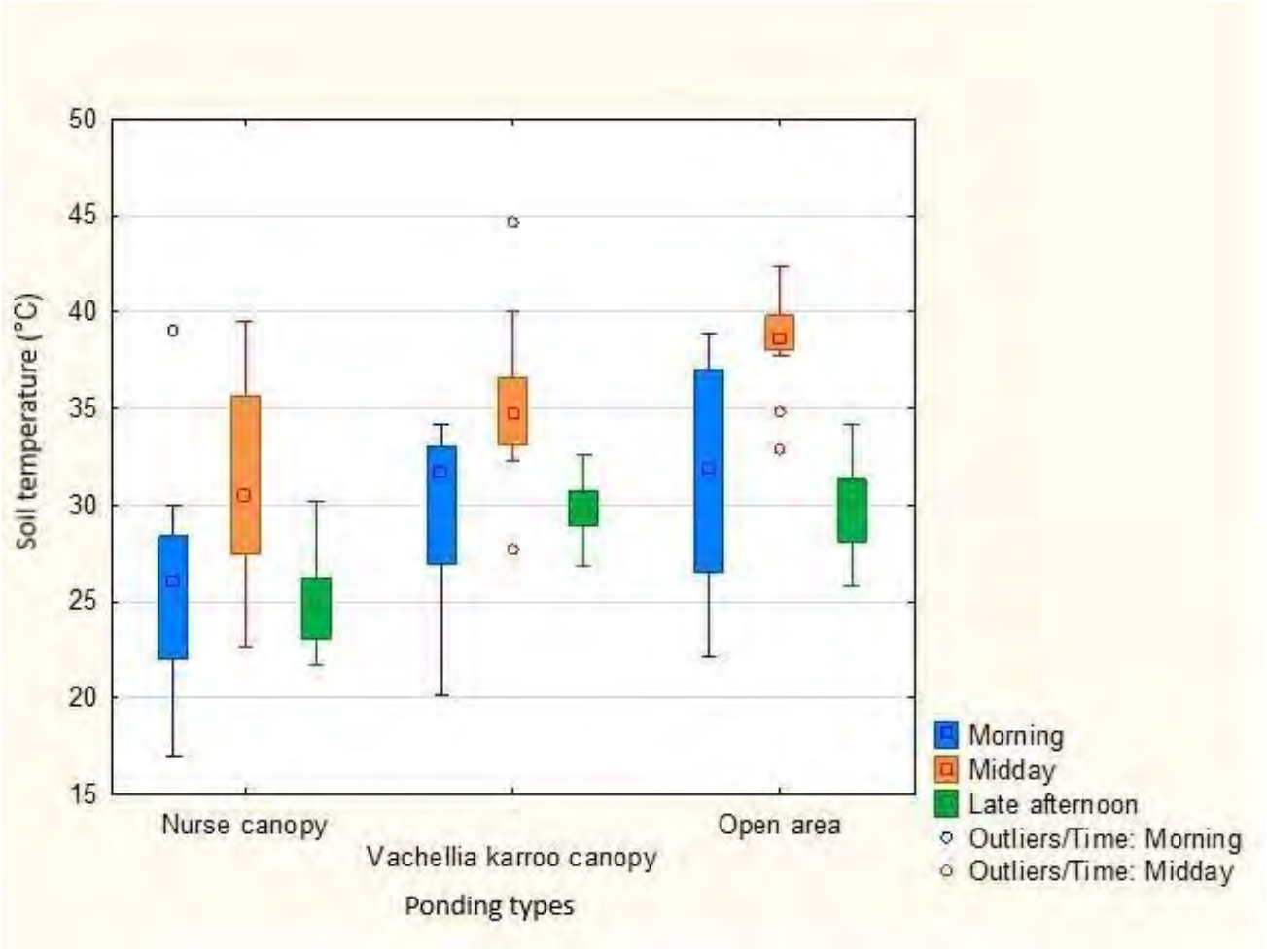

Figure 5.3. Mean soil temperature across pondings in Nurse canopy, *V. karroo* canopy and open areas.

Late afternoon

The late afternoon readings showed the Nurse canopy pondings having the lowest mean soil temperatures (24.9 ± 2.5°C), as it had throughout the course of the day. (Figure 5.3), while the WfW *P. afra* planting sites having the highest mean soil temperature (31.6 ± 3.2°C) (Figure 5.4). The Open area treatment by comparison had a slightly lower reading (30 ± 2.3°C). The *V. karroo* treatment, however, had a reduced mean soil temperature (29.7 ± 1.6°C) but the readings thereof were higherthan those of the Companion plant treatment (29.4 ± 1.4°C).

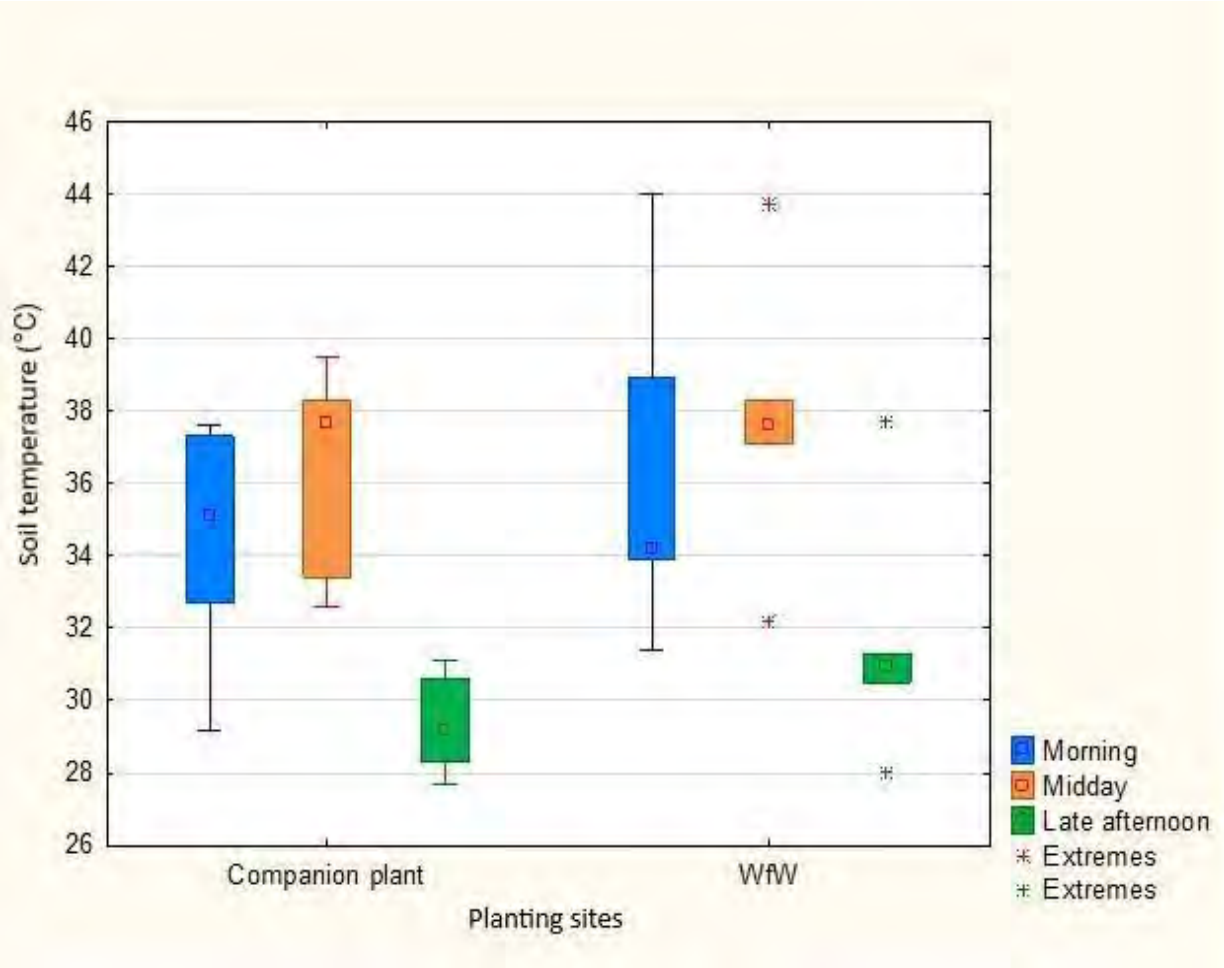

Figure 5.4. Mean soil temperature across *P. afra* planted with companion species and *P. afra* truncheons that were planted prior to the study

5.4.3. Mean Chlorophyll Fluorescence

Using a repeated measures ANOVA with a post-hoc Tukey HSD, the mean chlorophyll fluorescence of *P. afra* was found to not be significantly different between the different treatments (F (79, 237) = [1.211] p = 0.138) A repeated measures ANOVA determined that the chlorophyll fluorescence was not significantly different between the different potential nurse species (F (9,27) = [0.646]p = 0.749).

Maximum fluorescence

The Nurse plant treatment displayed the highest mean maximum fluorescence (Fv/m) (0.76 ± 0.06). The *V. karroo* treatment were slightly lower than the nurse plant treatment by comparison (0.74 ± 0.08). The open area treatment had the second- lowest reading

of maximum fluorescence (0.73 ± 0.13) and the Companion had the lowest out of the treatments (0.69 ± 0.09) (Figure 5.5).

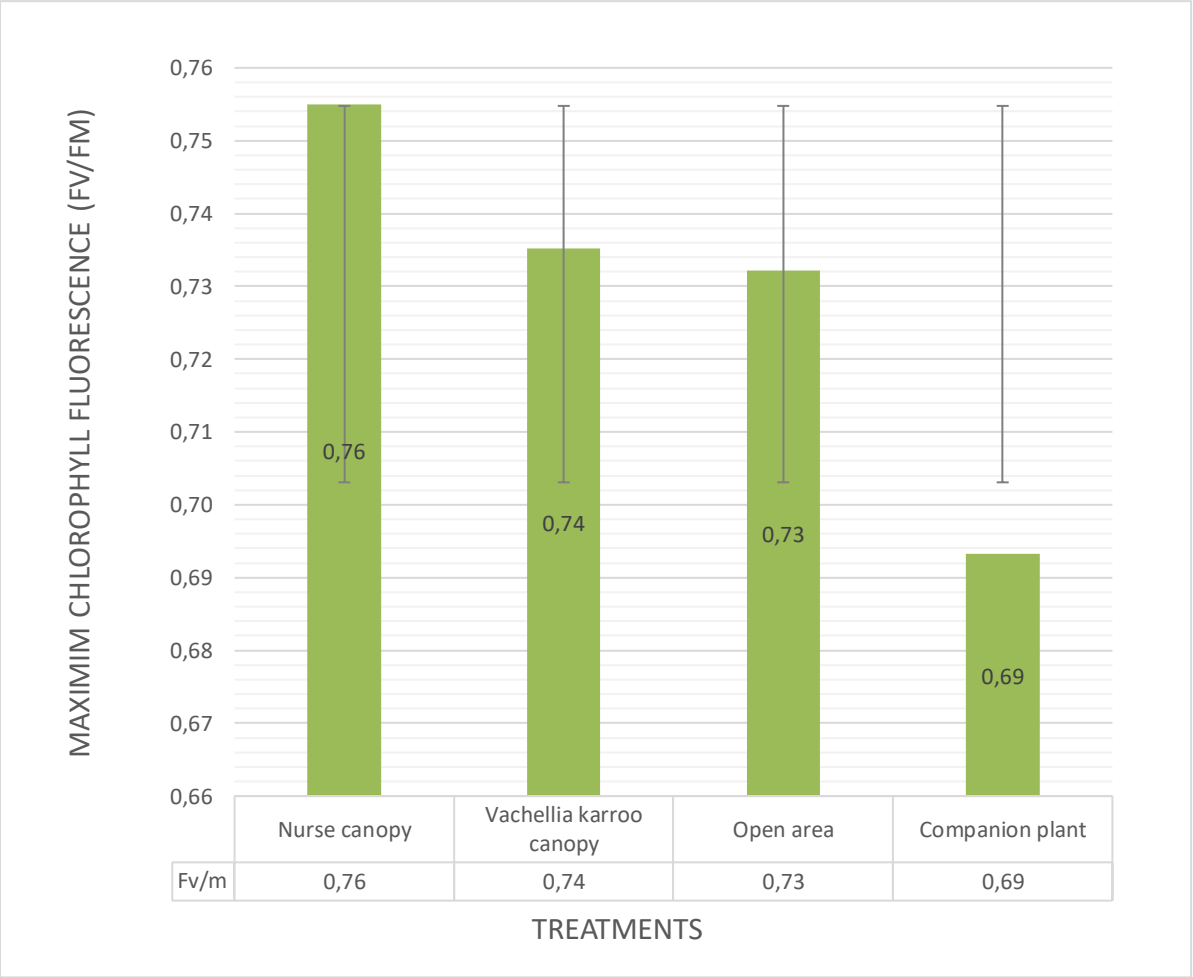

Figure 5.5. Mean values maximum fluorescence (Fv/Fm) in *P. afra* cuttings across different ponding types.

Potential nurse species

In the case of potential nurse species, *S. longispina* showed the highest maximum fluorescence (0.76 ± 0.02) *E. undulata* not being too far behind (0.74 ± 0.07). The companion species used in the study, *C. haematocarpa,* showed lower maximum fluorescence in comparison (0.65 ± 0.06) with *G. robusta* showing the lowest values (0.53 ± 0.08) (Figure 5.6).

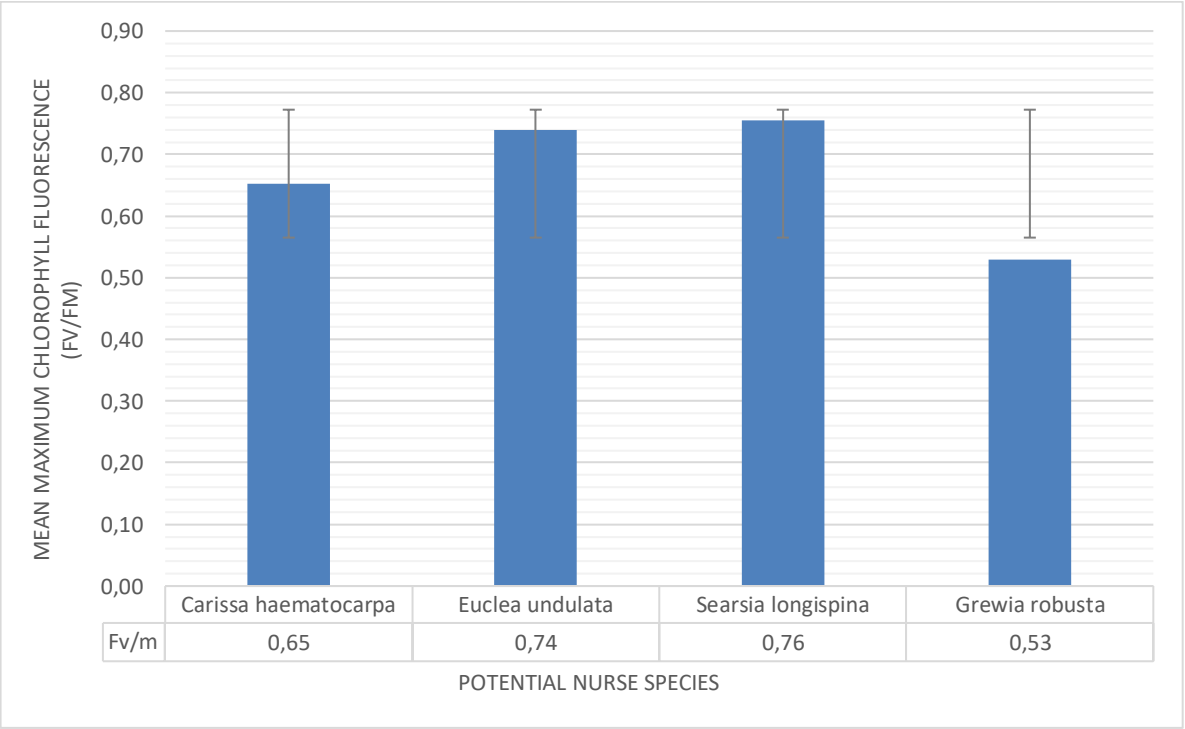

Figure 5.6. Mean values of and maximum fluorescence (Fv/Fm) in potential nurse plant species (bars represent 1 ± SD).

5.5 Discussion

5.5.1. Mean Soil Water Potential

The pondings under the nurse canopy retained the lowest mean soil water potential, likely due to competition between the nurse plant and the *P. afra* cuttings. Indeed, a study done by Hao *et al*., (2019) in the Great Plains, USA found that woody encroachment on prairie grasslands exacerbated the depletion of soil moisture during

periods of drought. This is merely a possible explanation as woody encroachment has been found to have mixed effects on soil moisture and groundwater recharge (Acharya *et al*. 2018).

Another plausible explanation is that the nurse canopy intercepts much of the precipitation during small rainfall events, thereby limiting the amount of water available in the soil. This theory is supported by the findings by Cowling & Mills (2011) who found that throughfall in spekboom thicket ranges from 49 to 63% of gross rainfall, which is lower compared to the South African savannas (75-84%) and Amazonian rainforests (74-91%). Although the study site is situated in degraded thicket the throughfall theory still applies due to the presence of the *C. haematocarpa* shrubs on site.

The pondings under the *V. karroo* canopy also had a higher mean soil water potential compared to those under the nurse canopy. A possible explanation could be that lower *V. karroo* has less dense foliage compared to *C. haematocarpa* and thus allows for more throughfall in comparison as discussed above (Cowling & Mills, 2011). Furthermore, *V. karroo* is able to utilize water from deep underground due to its long taproot and by extension can increase the availability of water in the soil for other plants to access and utilize (Dingaan & du Preez, 2017).

Pondings in open areas, as expected, showed a relatively lower mean soil water potential compared to pondings under the nurse and *V. karroo* canopies, likely due to their exposure to solar radiation and therefore evaporation throughout the course of the day.

The WfW *P. afra* had fluctuating values of soil water potential, likely because the moisture from the morning dew was not picked up in the readings until during the afternoon. This is a plausible theory as Kosmas *et al*., (1998) along with Agam & Berliner (2005) state that dew formation and direct water adsorption are two mechanisms by which water can be added to the soil in semi-arid regions, increasing its availability.

In comparison to the WfW planting areas, the companion plant pondings had a lower mean soil water potential for the rest of the day likely due to competition between the *P. afra* cuttings and the companion plants (Holmgren *et al*. 1997).

5.5.2. Mean Soil Temperature

The pondings under the *V. karroo* canopy showed the lowest mean soil temperature. This is likely because while the *V. karroo* provides shade to the pondings it doesn't shield them from wind and precipitation as well as the nurse plants in the study. This combined effect of shading and reduced protection from wind and rain would consequently lower soil temperature (Ren *et al*. 2008).

Since the nurse plant's dense vegetation protects the pondings from non-optimal environmental conditions (Ren *et al*., 2008), it would be expected then, that the pondings would also be buffered from frost and generally cooler temperatures (Padilla & Pugnaire, 2006).

The pondings under the open area experienced the highest mean soil temperatures due to their exposure to solar radiation during the course of the day. WfW *P. afra* had higher soil temperatures because they were planted in hard capped soil (Powell 2009; Lagerwall, 2010), whereas companion plant pondings had slightly lower soil temperatures because they retained soil moisture, thus slightly lowering soil temperature.

5.5.3. Mean chlorophyll fluorescence in *P. afra*

The nurse canopy (0.76 ± 0.06) and *V. karroo* canopy pondings (0.74 ± 0.08) shared similar maximum fluorescence values. This is likely because light infiltration was limited to a similar degree underneath the respective canopies (Roque Marca *et al*. 2021).

The second-lowest Fv/Fm values were found in the open area (0.73 ± 0.13). This is despite the assumption that in the open area, the *P. afra* cuttings would be most exposed to direct sunlight. Although this would result in an increase in their photosynthetic rates, as reflected by the Fv/Fm value (Ball *et al*. 1995), it still doesn't explain why the Fv/Fm values in the open areas were lower than those of the cuttings underneath the aforementioned canopies.

Companion plant pondings had the lowest values (0.69 ± 0.09), which is inexplicable because while a reduction in Fv/Fm indicates photo-inhibitory damage to the leaves

in response to adverse environmental conditions such as water stress (Maxwell & Johnson, 2000; Prieto *et al*. 2009), none of the treatments in this study had a mean Fv/Fm value below 0.7.

These findings are similar to those by Louw (2012) who, when examining the chlorophyll fluorescence of *P. afra*, found that the Fv/Fm values were mostly between 0.7 and 0.8. This suggests that the *P. afra* in this study did not experience a discernible decrease in photosynthetic efficiency and stomatal conductance during drought as the Fv/Fm values were measured during the hot and dry summer.

5.5.4. Mean Chlorophyll fluorescence in potential companion species

The highest Fv/Fm values were observed in *S. longispina* (0.76 ± 0.02) (Figure 5.6). This suggests that among the potential nurse plants *S. longispina* has the highest photosynthetic rates and most resistant to photo-inhibitory damage (Ball *et al*.1995; Prieto *et al*. 2009).

The second highest was *E. undulata* (0.74 ± 0.07) and the second lowest was *C. haematocarpa* (0.65 ± 0.06). The values found in these two species, in comparison to those found in *S. longispina* and *G. robusta*, reflect moderate photoinhibition resistance and photosynthetic efficiency (Ball *et al*.1995). Although *E. undulata* and *C. haematocarpa* may not have the same degree of photosynthetic efficiency compared to *S. longispina*, they do have means of coping with drought such as leaf abscission (thus, producing much litter) and the resprouting of new leaves after re-watering (Sigwela et al. 2009; Louw, 2012).

The shrub *G. robusta* had the lowest Fv/Fm value (0.53 ± 0.08). This suggests that *G. robusta* is most susceptible to environmental stress compared to the rest of the nurse species (Maxwell & Johnson, 2000; Prieto et al. 2009). These findings differ from those of Louw (2012) where the Fv/Fm values of *G. robusta* remained above 0.7, except when the volumetric soil moisture content was below 30%. This suggests that *the G. robusta* in this study were found in soils with low volumetric soil moisture content and thus their photosynthetic efficiency and stomatal conductance were reduced.

Chapter 6. CONCLUSION

6.1. Watering regimes

The inclusion of post-planting irrigation further aided in *P. afra* plants establishing themselves in pondings that capture and store the water. However, bi-weekly watering regimes (and their influence of plant survivorship) require further investigation due to its rather confounding findings compared to those of monthly and weekly regimes. These findings are similar to those of Fick *et al*., (2020), who suggest that microsite enhancements (i.e., watering and shading) can induce the formation of biological soil crusts, which further enhance active restoration efforts. Furthermore, the protocols used in this study mirror those done in the CNP where *P. afra* cuttings were planted in pondings that harvested and stored rainwater, increasing growth rate compared to cuttings planted outside of pondings (Mills & Robson, 2017; Mills *et al*. 2018).

The ideal regime for achieving best results was found to be the weekly watering, as it served as the best buffer against drought due to increased water inputs. Ideally, this regime must be maintained for approximately three months to give the cuttings as much of a head start as possible in establishing themselves. However, post-planting irrigation in a degraded rangeland is expensive at a large scale (Mills *et al*. 2018) so watering the plants at least once a month can still produce satisfactory results.

All of the treatments implemented in the study achieved the primary objective of improving *P. afra* survivorship to 70-95%, thereby reducing costs of replacing dead *P. afra* via replanting and thus lowering the overall cost of restoration. These results are even more astounding given that the study area has been experiencing severe drought for three years.

6.2. Companion planting

Companion planting has the potential to aid restoration efforts, as the facilitation factor of the companion plants (Navarro-Cano *et al*. 2019) can enhance plant survivorship but it also gives the companion species a chance to establish themselves, thereby adding a biodiversity component to the restoration process. However, within the context of degraded arid thicket, the concept of companionplanting needs to be researched and experimented further. In particular, the death of most of

the *A. ferox* and that of all the woody companion species used in the study warrants further investigations on which locally indigenous companion species would be best suited for planting and most resistant to both desiccation and waterlogging. It is possible that the alteration of the local microclimate due to degradation (Sigwela *et al.* 2009) has made the planting sites unsuitable for the nursery-grown woody species. Therefore, using nursery-grown propagules is likely to be unfeasible for restoration (van der Vyver *et al*. 2012). A possible solution to this limitation may be to condition nursery-grown companion species to drought. Indeed, Valliere *et al*., (2019)found that when native plant species were exposed to episodic drought, some species responded positively to the pre-conditioning trials, although such responses may be species-specific. Navarro-Cano *et al*., (2018) propose a protocol for selecting key species for restoration based on their ability to restore ecosystem functions.

Planting companion species underneath a nurse canopy may also help enhance survivorship. This option is suggested by Badano *et al*., (2009) when they observed that *Quercus castanea* seedlings they planted in a restoration site fared better underneath *Mimosa* and *Senecio* canopies compared to being planted in the open where they were exposed to high levels of solar radiation and therefore photoinhibition. Using local xeric-adapted succulents (such as *Crassula ovata*) as companion species could potentially be the most feasible option (Louw, 2012). However, as was seen with the *A. ferox* in the study, measures to mitigate waterlogging (which is the downfall of succulents) in the pondings need to be investigated further.

6.3. Nurse planting

Although the nurse planting treatment did not perform as well as expected in comparison to the companion planting treatment and the Control, it still produced more than satisfactory results, whereby over 80 percent of *P. afra* planted in pondings underneath a nurse canopy survived. These results are likely because of the ameliorated microclimate that exists within nurse plants, whereby the soil underneath the canopy is ameliorated, thus aiding in establishment and improved survival of beneficiary plants (Mihoč, 2016). With that said, it would be useful to conduct further research on what other shrubs and thicket canopy species would make for the best nurse environment for *P. afra* planting.

6.4. Unrooted truncheons

While planting unrooted truncheons in a ponding has proven to be feasible, it is ideal to plant them in early spring (along with post-planting irrigation) to give them as much time as possible to strike roots under milder spring conditions. The concept of planting *P. afra* key seasonal windows was suggested by Powell (2009) and further investigation of these seasonal periods and their potential for incorporation in an improved planting protocol is needed.

Furthermore, the findings suggest that the pre-planting treatments of unrooted truncheons may vary in efficacy depending on the context of the experimental plots. For example, while scarification has proven to create more sites of root formation; it may not be as useful when the truncheons are being planted in relatively loose soil that allows sufficient infiltration to the bottom of the stems (which is where most roots would form and grow). In this case, the scarification may cause undue stress on the truncheons thereby reducing their chances of successful establishment. Similarly, van der Vyver (2018) found that pruning unrooted truncheons proved to be less effective than simply planting the truncheons deeper into the soil.

The simulation of herbivory through the removal of apical meristems is a potentially useful tool for encouraging establishment and growth in unrooted truncheons as the results suggest, it being the second-least stressful application next to not treating the truncheons at all (Control). It is worth further investigating whether similar results could be achieved with rooted cuttings as a means of encouraging the lower branches to form the skirt growth vital to the establishment and maintenance of thicket clumps (Vlok & Euston-Brown, 2002; Sigwela *et al*. 2009).

One method that has proven to have a significantly positive effect on truncheon survival is inoculating them with mycorhizzal fungi while the plants are being grown in nurseries (Aviwe, 2014). Inoculating *P. afra* prior to planting them in the field has the potential to significantly enhance survivorship and restoration efforts.

6.5. Ecophysiology

The use of nurse plants, whenever possible, is a sound way of planting *P. afra* in an environment that not only retains moisture for growth but also helps mitigate the negative effects of solar radiation. The results suggest that nurse plants, in this case

C. haematocarpa, represent a suitable microhabitat for *P. afra* and other thicket species to establish and thrive. This is because the biophysical conditions under such a nurse canopy are more ideal those under which *P. afra* are planted in the STRP programme (Powell, 2009; Lagerwall, 2010). These findings concur with those of Panter & Ruwanza (2019) who found that improved microsites within degraded thicket areas are associated with improved water-holding capacity and water availability, high litter cover and ultimately vegetation recovery. Furthermore, the restoration of thicket can help in not only flood control but also in increasing absorption of solar radiation for photosynthesis, thus decreasing albedo (Smart, 2016).

Local indigenous companion plants can also be a very useful tool in enhancing survivorship of *P. afra* and the biodiversity of the degraded planting sites via facilitation.

Although *V. karroo* is not a xeric thicket species, its abundance in degraded xeric thicket has the potential to serve as an advantage in restoring the degraded areas. Using pre-existing *V. karroo* as a potential nurse plant requires further investigation as the experiment has produced mixed results. Further investigation can also go into using shrubs such as *S. longispina* and woody canopy species such as *G.robusta* as nurse plants, should *C. haematocarpa* be scarce or absent across such sites. This is further emphasised by Sholto-Douglas (2019) who found that *P. afra* survivorship was highest (81 ± 1.3%) in planting areas that retained canopy dominant speciesand higher litter cover.

The planting of single truncheons in open degraded areas and the efficacy thereof needs to be re-evaluated, as these conditions are far from ideal for *P. afra* establishment compared to truncheons planted in clumps and inside pondings (Mills & Robson, 2017).

6.6. Final conclusion and recommendations

With respect to the context and findings of the study, the following can be surmised:

1. The inclusion of post-planting irrigation (at least once a month over three months) further aid in *P. afra* establishing itself.
2. The construction of pondings aids *P. afra* establishment as the final survivorship

in each treatment was well over 50%.

3. The high mortality rates of the companion species used in the study warrants further investigation on species selection.
4. In comparison, nurse planting can be a potentially more reliable approach to planting *P. afra* in degraded arid thicket. Firstly, because the nurse planting treatment produced high *P. afra* survivorship. Secondly, because the nurse plants would ideally bewell established and therefore have a lower risk of mortality. This is justified by the fact that none of the nurse plants died throughout the course of the study. With that being said, further studies are required to confirm whether nurse planting can be a feasible option for large scale arid thicket restoration.
5. While using unrooted truncheons has proven to be feasible, it is ideal to plant them in a ponding during early spring (along with post-planting irrigation) to give them as much time as possible to strike roots as under milder spring conditions.
6. The planting protocols used in the study achieved the primary objective of improving *P. afra* survivorship to at least 50%. However, further research on enhancing the cost-effectiveness of these protocols is required to incentivise landowners into investing time and resources into these protocols and

ensuring that local communities benefit from the potential income opportunities that may arise from such restoration action

From these conclusions I would recommend the following techniques to improve the current *P. afra* planting protocol:

- The construction of pondings can be a very useful technique in creating conditions favourable for not only *P. afra* survivorship and growth, but also for the recruitment of other locally indigenous thicket species.
- In the presence of a game fence, *P. afra* can be successfully planted in pondings without the need to brush-pack.
- Nurse planting has the potential to be a game-changer for *P. afra* propagation as the *P. afra* cuttings benefit from the micro-climate and protection (from herbivory and harsh weather conditions) the nurse canopy provides. This was the case with the nurse species *C. haematocarpa* in the study area (where it is indigenous). This spinescent shrub, with its resistance to drought and herbivory, coupled with the ability to produce organic matter (through its deciduous leaves) and shading underneath its canopy make it an ideal nurse plant. Furthermore, nurse plantingcan also enhance the establishment of bush clumps in degraded thicket, therefore facilitating further recruitment of thicket species on the surrounding bare patches (Vlok & Euston-Brown, 2002; Adie & Yeaton, 2013). Furthermore, using other thicket species such as *S. longispina*, *G. robusta* and *Azima tetracantha* as nurse plants is a concept worth looking further into and testing out, as they may achieve similar results to *C. haematocarpa.*
- Companion planting is a method that requires further investigation on its influence on the improved survivorship and growth of *P. afra*. Further research is required to investigate ways to mitigate transplant shock for the companion plants in question.
- Propagating local succulent companion plants can also be a viable alternative for rehabilitation of degraded xeric thicket as there is less aftercare required since these plants seem to be more drought-resistant and naturally recruit in degraded landscapes. However, further investigation is required to confirm the feasibility of this option.

- Companion plants and unrooted truncheons would likely fare better if planted in early spring when temperatures get progressively warmer and rainfall gradually increases (theoretically).
- Ultimately, these innovations need to be compared with the existing STRP planting protocol to determine whether they are more effective in rehabilitating degraded arid thicket.

It must be noted that, these intensive protocols, while proven to be effective during the study are still unlikely to be feasible for large scale restoration programmes at the moment. Even at the bare minimum estimate of USD 722 ha^{-1} (which equates to ZAR 10 656.6 ha^{-1}) (Mills et al. 2007), the cost is simply too high for most landowners to afford, especially when considering the fact that hundreds of thousands of hectares of thicket need to be rehabilitated and restored. Economic incentives and further innovation of the planting protocols need to be investigated so that the restoration of the thicket at large becomes more cost-effective and economically beneficial for all stakeholders that utilize this unique and fragile ecosystem.

7. Reference List

ACHARYA, B.S., KHAREL, G., ZOU, C.B., WILCOX, B.P. and HALIHAN, T., 2018. Woody plant encroachment impacts on groundwater recharge: A review. *Water,* **10**(10), pp. 1466.

ACOCKS, J.P.H., 1988. Veld types of South Africa*. Memoirs of the Botanical Survey of South Africa,* **28**, pp.1-128*.*

ADIE, H. and YEATON, R.I., 2013. Regeneration dynamics in arid subtropical thicket, South Africa. *South African Journal of Botany*, ***88***, pp.80-85.

ADIE, H. and YEATON, R.I., 2014. Directed dispersal and decomposition drive cyclic succession in arid subtropical thicket. *Plant Ecology,* **215**(5), pp. 507-515.

AGAM, N. and BERLINER, P.R., 2006. Dew formation and water vapor adsorption in semi-arid environments–A review. *Journal of Arid Environments*, **65**(4), pp.572-590.

AL-KAISI, M.M., LAL, R., OLSON, K.R. and LOWERY, B., 2017. Fundamentals and functions of soil environment. Academic Press. *Soil Health and Intensification of Agroecosystems,* pp. 1-23.

ALLISON, L. E., BERNSTEIN, L., BOWER, C. A., BROWN, J. W., FIREMAN, M., HATCHER, J. T., HAYWARD, H. E., PEARSON, G. A., REEVE, R. C., RICHARDS, A. and WILCOX, L. V. 1954. Diagnosis and improvement of saline and alkali soils. United States Department of Agriculture, Virginia.

ALTMAN, M. and HEMSON, D., 2008. The role of expanded public works programmes in halving unemployment.

ANDERSON, P., HOFFMAN, T. and HOLMES, P.M., 2004. The potential of *Cephalophyllum inaequale* (L. Bolus) for the restoration of degraded arid landscapes in Namaqualand, South Africa. *Restoration Ecology,* **12**(3), pp. 343-351.

APOGEE INSTRUMENTS, INC. 2016. Conversion - PPFD to Watts. [https://www.apogeeinstruments.com/conversion-ppfd-to-watts/] Accessed: February 6, 2023

ARAI-SANOH, Y., ISHIMARU, T., OHSUMI, A. and KONDO, M., 2010. Effects of soil temperature on growth and root function in rice. *Plant Production Science*, **13**(3), pp. 235-242.

ARONSON, J., FLORET, C., LE FLOC'H, E., OVALLE, C. and PONTANIER, R., 1993. Restoration and rehabilitation of degraded ecosystems in arid and semi-arid lands. I. A view from the south. *Restoration Ecology,* **1**(1), pp. 8-17.

ARTIOLA, J.F., BRUSSEAU, M.L. and PEPPER, I.L., 2004. Environmental monitoring and characterization. Academic Press.

ASH, A.J. and BELLAMY, J.A., 1994. State and transition models for rangelands. 4. Application of state and transition models to rangelands in northern Australia. *Tropical Grasslands,* **28**, pp. 223-228.

BADANO, E.I., PÉREZ, D. and VERGARA, C.H., 2009. Love of nurse plants is not enough for restoring oak forests in a seasonally dry tropical environment. *Restoration Ecology*, ***17***(5), pp.571-576.

BAKER, N.R., 2008. Chlorophyll fluorescence: a probe of photosynthesis in vivo. *Annual Review of Plant Biology.,* **59**, pp. 89-113.

BALENSIEFER, M., ROSSI, R., ARDINGHI, N., CENNI, M. and UGOLINI, M., 2004. SER International Primer on Ecological Restoration. Society for Ecological Restoration [Online] Available at: https://floridalivingshorelines.com/wp-content/uploads/2015/05/Clewell.Aronson.Winterhalder.2004-SER-Primer.pdf Accessed: January 2022

BALL, M.C., BUTTERWORTH, J.A., RODEN, J.S., CHRISTIAN, R., EGERTON, J., WYDRZYNSKI, T.J., CHOW, W.S. and BADGER, M.R., 1995. Applications of chlorophyll fluorescence to forest ecology. *Functional Plant Biology,* **22**(2), pp. 311-

319.

BARAN, R.J., 1999. *Portulacaria afra* the Elephant's food or spekboom: an ongoing monograph which contains some of the areas of both knowledge and ignorance pertaining to this plant, https://www.magiminiland.org/Portulacaria.html

BECKER, C.H., 2013. The influence of soil properties on the growth and distribution of *Portulacaria afra* in subtropical thicket, South Africa. Doctoral dissertation, Nelson Mandela Metropolitan University, Port Elizabeth, South Africa, pp. 106.

BEISNER, B.E., HAYDON, D.T. and CUDDINGTON, K., 2003. Alternative stable states in ecology. *Frontiers in Ecology and the Environment,* **1**(7), pp. 376-382.

BERNHARDT-RÖMERMANN, M., RÖMERMANN, C., NUSKE, R., PARTH, A., KLOTZ, S., SCHMIDT, W. and STADLER, J., 2008. On the identification of the most suitable traits for plant functional trait analyses. *Oikos*, **117**(10), pp.1533-1541.

BERRIMAN, S. 2010. The effect of herbivory on survivorship and growth rate of subtropical thicket species following restoration plantings. Unpublished honours thesis, Rhodes University, Grahamstown, South Africa. pp.22-35.

BESTELMEYER, B.T., TUGEL, A.J., PEACOCK JR, G.L., ROBINETT, D.G., SHAVER, P.L., BROWN, J.R., HERRICK, J.E., SANCHEZ, H. and HAVSTAD, K.M., 2009. State-and-transition models for heterogeneous landscapes: a strategy for development and application. *Rangeland Ecology & Management,* **62**(1), pp. 1-15.

BESTELMEYER, B.T., MOSELEY, K., SHAVER, P.L., SANCHEZ, H., BRISKE, D.D. and FERNANDEZ-GIMENEZ, M.E., 2010. Practical guidance for developing state-and-transition models. *Rangelands,* **32**(6), pp. 23-30.

BESTELMEYER, B.T., 2015. National assessment and critiques of state-and-transition models: the baby with the bathwater. *Rangelands,* **37**(3), pp. 125-129.

BESTELMEYER, B.T., OKIN, G.S., DUNIWAY, M.C., ARCHER, S.R., SAYRE, N.F., WILLIAMSON, J.C. and HERRICK, J.E., 2015. Desertification, land use, and the transformation of global drylands. *Frontiers in Ecology and the Environment,* **13**(1), pp. 28-36.

BHALUDRA, C.S.S., BETHAPUDI, R.R., MURUGULLA, A.C., PULLAGUMMI, C., LATHA, T., VENKATESH, K., BHEEMAGANI, A.J., PUDUTHA, A. and RANI, A.R., 2013. Cultivation, phytochemical studies, biological activities and medicinal uses of *Aloe ferox*, grandfather of aloes an important amazing medicinal plant. *International Journal of Pharmacology,* **9**(7), pp. 405-415.

BITTELLI, M., 2010. Measuring soil water potential for water management in agriculture: A review. *Sustainability*, **2**(5), pp.1226-1251.

BOSHOFF, A., COWLING, R.M. and KERLEY, G.I.H., 2000. The Baviaanskloof Conservation Area: a conservation and tourism development priority. Terrestrial Ecology Research Unit, University of Port Elizabeth. **58**. pp.1-52.

BOSHOFF, A., COWLING, R. and KERLEY, G., 2001. Baviaanskloof. *Veld & Flora*, **87**(2), pp.74-77.

BOSHOFF, A.F., 2008. The Baviaanskloof Mega-Reserve: from concept to implementation. Centre for African Conservation Ecology, Nelson Mandela Metropolitan University, Port Elizabeth, South Africa*,* **66***,* pp. 1-54.

BOSHOFF, A. and COWLING, R., 2005. The Baviaanskloof Mega-Reserve: An environmentally, socially and economically sustainable conservation and development initiative. Terrestrial Ecology Research Unit, Nelson Mandela Metropolitan University, Port Elizabeth, South Africa.

BOTAI, C.M., BOTAI, J.O., ADEOLA, A.M., DE WIT, J.P., NCONGWANE, K.P. and ZWANE, N.N., 2020. Drought risk analysis in the Eastern Cape Province of South Africa: the copula lens. *Water,* **12**(7), pp. 1938.

BOTHA, M.S., CARRICK, P.J. and ALLSOPP, N., 2008. Capturing lessons from land-users to aid the development of ecological restoration guidelines for lowland Namaqualand. *Biological Conservation,* **141**(4), pp. 885-895.

CADOTTE, M.W., CARSCADDEN, K. and MIROTCHNICK, N., 2011. Beyond species: functional diversity and the maintenance of ecological processes and services. *Journal of Applied Ecology,* **48**(5), pp. 1079-1087.

CALLAWAY, R.M., 2007. Positive interactions and interdependence in plant communities. Springer.

CARMONA, D., LAJEUNESSE, M.J. and JOHNSON, M.T., 2011. Plant traits that predict resistance to herbivores. *Functional Ecology*, **25**(2), pp.358-367.

CONVENTION ON BIOLOGICAL DIVERSITY, 2010. Strategic Plan for Biodiversity (2011-2020) and the Aichi Biodiversity Targets. Montreal: Convention on Biological Diversity. [Online] Available at https://www.cbd.int/sp/ Accessed: January, 2022

CORTINA, J., MAESTRE, F.T., VALLEJO, R., BAEZA, M.J., VALDECANTOS, A. and PÉREZ-DEVESA, M., 2006. Ecosystem structure, function, and restoration success: are they related? *Journal for Nature Conservation,* **14**(3-4), pp. 152-160.

COWLING, R. and KERLEY, G.I.H., 2002. Impacts of elephants on the flora and vegetation of subtropical thicket in the Eastern Cape. In Elephant conservation and management in the Eastern Cape: Workshop Proceedings. Terrestrial Ecology Research Unit, University of Port Elizabeth, South Africa, Report. **35**, pp. 55-72.

COWLING, R.M., PROCHES, S., VLOK, J. and VAN STADEN, J., 2005. On the origin of southern African subtropical thicket vegetation. *South African Journal of Botany,* **71**(1), pp. 1-23.

COWLING, R.M. and KNIGHT, A.T., 2006. Into the thick of it: bridging the research-implementation gap in the thicket biome through the Thicket Forum: news & views. *South African Journal of Science*, **102**(9), pp.406-408.

COWLING, R.M. and MILLS, A.J., 2011. A preliminary assessment of rain throughfall beneath *Portulacaria afra* canopy in subtropical thicket and its implications for soil carbon stocks. *South African Journal of Botany,* **77**(1), pp. 236- 240.

DANE, JH and HOPMANS JW., 2002. Water retention and storage. In: Methods of Soil Analysis, Part 4 Physical Methods. pp 671 – 720. Soil Science Society of America, Inc, Madison, Wisconsin, USA

DASTORANI, M.T., KOUHZAD, B., SEPEHR, A. and TALEBI, A., 2017. The role of rainwater harvesting on providing vegetation water requirement in arid and semi-arid regions. *Eur Water,* **58**(2017), pp. 521-527.

DAVIES, J., GUDKA, M., LABAN, P., METTERNICHT, G., ALEXANDER, S., HANNAM, I., WELLING, L., VASSEUR, L., SILES, J. and AGUILAR, L., 2015. Land Degradation Neutrality: Implications and Opportunities for Conservation. *IUCN: Nairobi, Kenya.*

DAYARAM, A., HARRIS, L.R., GROBLER, B.A., VAN DER MERWE, S., REBELO, A.G., POWRIE, L.W., VLOK, J.H., DESMET, P.G., QABAQABA, M., HLAHANE, K.M. and SKOWNO, A.L., 2019. Vegetation Map of South Africa, Lesotho and Swaziland 2018: A description of changes since 2006. *Bothalia-African Biodiversity & Conservation*, **49**(1), pp.1-11.

DE ABREU, P., 2011. The effect of rehabilitation on ecosystem services in the semi-arid Succulent Karoo lowlands of the Little Karoo, South Africa. (Doctoral dissertation, Council for Scientific and Industrial Research).

DE BOECK, H.J., BASSIN, S., VERLINDEN, M., ZEITER, M. and HILTBRUNNER, E., 2016. Simulated heat waves affected alpine grassland only in combination with drought. *New Phytologist,* **209**(2), pp. 531-541.

DE PAOLI, G., 2008. River System Restoration for A Sustainable Land and Water Management in The Baviaanskloof Mega-Reserve. Wageningen, Alterra.

DE VILLIERS, J., 1941. The geology of the Baviaanskloof. *Transactions of the Geological Society of South Africa*, pp.151-159.

DINGAAN, M. and DU PREEZ, P.J., 2018. *Vachellia (Acacia) karroo* Communities in South Africa: An Overview. *Pure and Applied Biogeography,* pp. 109.

DOMISCH, T., FINÉR, L. and LEHTO, T., 2001. Effects of soil temperature on biomass and carbohydrate allocation in Scots pine (*Pinus sylvestris*) seedlings at the beginning of the growing season. *Tree Physiology*, **21**(7), pp.465-472.

DU PLESSIS, M., 2016. Agriculture: Facts and trends South Africa. *Online: http://awsassets.wwf.org.za/downloads/facts_brochure_mockup_04_b. pdf.WWF-SA. Accessed: October, 2016.*

DU TOIT, A.L., 1939. *The Geology of South Africa.* Oliver and Boyd. United Kingdom, 611 pp.

DUKER, R., COWLING, R.M., DU PREEZ, D.R. and POTTS, A.J., 2015. Frost, *Portulacaria afra* Jacq., and the boundary between the Albany Subtropical Thicket and Nama-Karoo biomes. *South African Journal of Botany,* **101**, pp. 112-119.

EVETTE, A., LABONNE, S., REY, F., LIEBAULT, F., JANCKE, O. and GIREL, J., 2009. History of bioengineering techniques for erosion control in rivers in Western Europe. *Environmental management,* **43**(6), pp. 972.

FABRICIUS, C., PALMER, A.R. and BURGER, M., 2002. Landscape diversity in a conservation area and commercial and communal rangeland in Xeric Succulent Thicket, South Africa. *Landscape Ecology,* **17**(6), pp. 531-537.

FAVAS, P.J., MARTINO, L.E. and PRASAD, M.N., 2018. Abandoned mine land reclamation–Challenges and opportunities (Holistic Approach). *Bio- Geotechnologies for Mine Site Rehabilitation.* Elsevier, pp. 3-31.

FICK, S.E., DAY, N., DUNIWAY, M.C., HOY-SKUBIK, S. and BARGER, N.N., 2020. Microsite enhancements for soil stabilization and rapid biocrust colonization in degraded drylands. *Restoration Ecology,* **28**, pp. S139-S149.

FOX, P. and ROCKSTRÖM, J., 2003. Supplemental irrigation for dry-spell mitigation of rainfed agriculture in the Sahel. *Agricultural Water Management,* **61**(1), pp. 29-50.

FULMAKA, A., 2016. Investigating soil microbial interactions of *Portulacaria afra*. Masters thesis. Department of Biochemistry and Microbiology. Rhodes University, Grahamstown, South Africa.

GIL, R.T., 2012. Soil Bioengineering Measures A case study in Western Baviaanskloof. Unpublished master's thesis. University of Kiel (Christian-Albrechts-Universität zu Kiel), Germany. pp.1-82.

GLENDAY, J.A., 2015. Modeling the Hydrologic Impacts of Vegetation and Channel Network Change for a Semi-arid, Mountainous, Meso-scale Catchment: The Baviaanskloof, South Africa. Doctoral dissertation, University of California, Santa Barbara.

GOLDBLATT, A., 2010. *Agriculture: facts and trends South Africa.* World Wildlife Forum. [Online] Available at: http://awsassets.wwf.org.za/downloads/facts_brochure_mockup_04_b.pdf Accessed: October, 2016.

GÓMEZ-APARICIO, L., GÓMEZ, J.M., ZAMORA, R. and BOETTINGER, J.L., 2005. Canopy vs. soil effects of shrubs facilitating tree seedlings in Mediterranean montane ecosystems. *Journal of Vegetation Science*, **16(**2), pp.191-198.

GORDON, I.J. and PRINS, H.H., 2008. The ecology of browsing and grazing (No. 195). Berlin, Germany: Springer.

GURALNICK, L.J. and GLADSKY, K., 2017. Crassulacean acid metabolism as a continuous trait: variability in the contribution of Crassulacean acid metabolism (CAM) in populations of *Portulacaria afra*. *Heliyon,* **3**(4), pp. e00293.

GURALNICK, L.J. and JACKSON, M.D., 2001. The occurrence and phylogenetics of crassulacean acid metabolism in the Portulacaceae. *International journal of plant sciences,* **162**(2), pp. 257-262.

GURALNICK, L.J., RORABAUGH, P.A. and HANSCOM III, Z., 1984a. Influence of Photoperiod and Leaf Age on Crassulacean Acid Metabolism in *Portulacaria afra* (L.) Jacq. *Plant Physiology,* **75**(2), pp. 454-457.

GURALNICK, L.J., RORABAUGH, P.A. and HANSCOM III, Z., 1984b. Seasonal shifts of photosynthesis in *Portulacaria afra* (L.) Jacq. *Plant Physiology,* **76**(3), pp. 643-646.

GURALNICK, L.J. and TING, I.P., 1986. Seasonal response to drought and rewatering in Portulacaria afra (L.) Jacq. *Oecologia,* **70**(1), pp. 85-91.

HANKE, W., GRÖNGRÖFT, A., JÜRGENS, N. and SCHMIEDEL, U., 2011. Rehabilitation of arid rangelands: Intensifying water pulses from low-intensity winter rainfall. *Journal of Arid Environments,* **75**(2), pp. 185-193.

HANKE, W., WESULS, D., MÜNCHBERGER, W. and SCHMIEDEL, U., 2015. Tradeoffs in the rehabilitation of a succulent Karoo rangeland. *Land Degradation & Development,* **26**(8), pp. 833-842.

HAO, Y., LIU, Q., LI, C., KHAREL, G., AN, L., STEBLER, E., ZHONG, Y. and ZOU, C.B., 2019. Interactive Effect of Meteorological Drought and Vegetation Types on Root Zone Soil Moisture and Runoff in Rangeland Watersheds. *Water,* **11**(11), pp. 2357.

HARMER, R., 1988. Production and use of epicormic shoots for the vegetative propagation of mature oak. *Forestry: An International Journal of Forest Research,* **61**(4), pp. 305-316.

HATTINGH L., 2011. “Baviaanskloof: A World Apart”, Willowmore, South Africa: Baviaanskloof Tourism Association.

HILL, R., DYER, G.A., LOZADA-ELLISON, L., GIMONA, A., MARTIN-ORTEGA, J., MUNOZ-ROJAS, J. and GORDON, I.J., 2015. A social-ecological systems analysis of impediments to delivery of the Aichi 2020 Targets and potentially more effective pathways to the conservation of biodiversity. *Global Environmental Change,* **34**, pp. 22-34.

HOARE, D.B., MUCINA, L., RUTHERFORD, M.C., VLOK, J.H., EUSTON-BROWN, D.I., PALMER, A.R., POWRIE, L.W., LECHMERE-OERTEL, R.G., PROCHES, S.M. and DOLD, A.P., 2006. Albany thicket biome. *The vegetation of South Africa, Lesotho and Swaziland. Strelitzia,* **19**, pp. 541-567.

HOFFMAN, M.T. and TODD, S., 2000. A national review of land degradation in South Africa: the influence of biophysical and socio-economic factors. *Journal of Southern African Studies,* **26**(4), pp. 743-758.

HOLMES, P., 2012. Southern African geomorphology: Recent trends and new directions. African Sun Media

HOLMES, P. and MEADOWS, M., 2012. Southern African Geomorphology. Recent Trends and New Directions, SUN MEDIA, Bloemfontein.

IUCN, S., 2010. African Rhino Specialist Group. 2008. *Diceros bicornis*. Unpublished report.

IPCC, 2021. Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [Masson-Delmotte, V., P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M.I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J.B.R. Matthews, T.K. Maycock, T. Waterfield, O. Yelekçi, R. Yu, and B. Zhou (eds.)]. Cambridge University Press. In Press.

JANSEN, H.C., 2008. Water for Food and Ecosystems in the Baviaanskloof Mega Reserve. *Land and water resources assessment in the Baviaanskloof, Eastern Cape Province, South Africa.*

KERLEY, G.I., KNIGHT, M.H. and DE KOCK, M., 1995. Desertification of subtropical thicket in the Eastern Cape, South Africa: are there alternatives? *Environmental monitoring and assessment,* **37**(1), pp. 211-230.

KERLEY, G.I., BOSHOFF, A.F. and KNIGHT, M.H., 1999a. Ecosystem integrity and sustainable land-use in the Thicket Biome, South Africa. *Ecosystem Health,* **5**(2), pp. 104-109.

KERLEY, G., TONGWAY, D.J. and LUDWIG, J.A., 1999b. Effects of goat and elephant browsing on soil resources in Succulent Thicket, Eastern Cape, South Africa. In: Eldridge, D. & Freudenberger, D. (eds). Proceedings of the VIth International Rangelands Congress. Volume 1. Townsville, Australia. Pg.116–117

KIMBALL, S., FUNK, J.L., SANDQUIST, D.R. and EHLERINGER, J.R., 2016. Ecophysiological considerations for restoration. In *Foundations of restoration ecology*. pp. 153-181. Island Press, Washington, DC.

.

KIMITI, D.W., RIGINOS, C. and BELNAP, J., 2017. Low-cost grass restoration using erosion barriers in a degraded African rangeland. *Restoration Ecology,* **25**(3), pp. 376-384.

KNIGHT, A.T. and COWLING, R.M., 2006. Into the thick of it: bridging the research-implementation gap in the thicket biome through the Thicket Forum: news & views. *South African Journal of Science,* **102**(9), pp. 406-408.

KUEPPER, G., DODSON, M. and DUNCAN, J., 2016. Companion Planting & Botanical Pesticides: Concepts & Resources. *ATTRA Sustainable Agriculture, updated April, 2016.* pp.1-20.

KOSMAS, C., DANALATOS, N.G., POESEN, J. AND VAN WESEMAEL, B., 1998. The effect of water vapour adsorption on soil moisture content under Mediterranean climatic conditions. *Agricultural Water Management*, **36**(2), pp.157-168.

LAMBERS, H.L., PONS, T.L. and PONS, T.L., 2008. Plant physiological ecology. Springer Science & Business Media, New York, USA.

LAGERWALL, D. 2010. Is the Subtropical Thicket Restoration Project cost effective? Unpublished honours thesis, Rhodes University, Grahamstown, South Africa. pp.1- 26.

LECHMERE-OERTEL, R.G., KERLEY, G. and COWLING, R.M., 2005. Patterns and implications of transformation in semi-arid succulent thicket, South Africa. *Journal of Arid Environments,* **62**(3), pp. 459-474.

LECHMERE-OERTEL, R.G., KERLEY, G., MILLS, A.J. and COWLING, R.M., 2008. Litter dynamics across browsing-induced fenceline contrasts in succulent thicket, South Africa. *South African Journal of Botany,* **74**(4), pp. 651-659.

LEWIS, L., SALISBURY, S.L., HAGEN, S. and MAURER, L.A., 2001. Soil bioengineering for upland slope stabilization. WSDOT Research Project WA-RD, pp. 491.

LEWIS, C.A., 2008. Geomorphology of the Eastern Cape, South Africa. National Inquiry Services Centre, Grahamstown.

LI, X., ZHANG, L. and ZHANG, Z., 2006. Soil bioengineering and the ecological restoration of riverbanks at the Airport Town, Shanghai, China. *Ecological Engineering,* **26**(3), pp. 304-314.

LINIGER, H., and CRITCHLEY, W. (Eds.). 2007. *Where the land is greener—case studies and analysis of soil and water conservation initiatives worldwide.* CTA, FAO, UNEP, CDE, WOCAT. Berne, Switzerland.

LLOYD, J.W., VAN DEN BERG, E C and PALMER, A.R., 2002. Patterns of transformation and degradation in the thicket biome, South Africa. Unpublished Report, Terrestrial Ecology Research Unit, Dept of Zoology, University of Port Elizabeth, South Africa., **39**, pp. 86.

LOUW, M., 2012. Propagation and stress physiology of selected Subtropical Thicket species: towards increasing biodiversity at rehabilitation sites. Master's thesis. Nelson Mandela Metropolitan University, South Africa.

LUDWIG, J.A. and TONGWAY, D.J., 1996a. Rehabilitation of semiarid landscapes in Australia. II. Restoring vegetation patches. *Restoration Ecology,* **4**(4), pp. 398-406.

LUDWIG, J.A. and TONGWAY, D.J., 1996b. Rehabilitation of semiarid landscapes in Australia. II. Restoring vegetation patches. *Restoration Ecology,* **4**(4), pp. 398-406.

LUDWIG, J.A., WILCOX, B.P., BRESHEARS, D.D., TONGWAY, D.J. and IMESON, A.C., 2005. Vegetation patches and runoff-erosion as interacting ecohydrological processes in semiarid landscapes. *Ecology,* **86**(2), pp. 288-297.

MA, H. and ZHAO, H., 1994. United Nations: Convention to combat desertification in those countries experiencing serious drought and/or desertification, particularly in Africa. *International.Legal Mater,* **33**, pp. 1328-1382.

MAESTRE, F.T. and CORTINA, J., 2004. Insights into ecosystem composition and function in a sequence of degraded semiarid steppes. *Restoration Ecology,* **12**(4), pp. 494-502.

MAHLALELA, P.T., BLAMEY, R.C., HART, N. and REASON, C., 2020. Drought in the Eastern Cape region of South Africa and trends in rainfall characteristics. *Climate Dynamics,* **55**(9), pp. 2743-2759.

MARAIS, C., COWLING, R.M., POWELL, M. and MILLS, A., 2009. Establishing the platform for a carbon sequestration market in South Africa: the working for woodlands subtropical thicket restoration programme, *XIII World Forestry Congress (October)*

2009, pp. 1-13.

MAXWELL, K. and JOHNSON, G.N., 2000. Chlorophyll fluorescence–a practical guide. *Journal of experimental botany,* **51**(345), pp. 659-668.

MCDOWELL, C., 1987. Growing succulents under cover in South Africa: water and soil. *Veld & Flora*, **73**(4), pp.126-128.

MEADOWS, M.E. and HOFFMAN, T.M., 2003. Land degradation and climate change in South Africa. *The Geographical Journal*, **169**(2), pp.168-177.

MESSINGER, J. and WINTERBOTTOM, B., 2016. African forest landscape restoration initiative (AFR100): restoring 100 million hectares of degraded and deforested land in Africa. *Nature & Faune,* **30**(2), pp. 14-17.

MICROSOFT CORPORATION, 2018. Microsoft Excel, Available at: https://office.microsoft.com/excel.

MIDEGA, C.A., BRUCE, T.J., PICKETT, J.A., PITTCHAR, J.O., MURAGE, A. and KHAN, Z.R., 2015. Climate-adapted companion cropping increases agricultural productivity in East Africa. *Field Crops Research,* **180**, pp. 118-125.

MIHOČ, M., GIMÉNEZ-BENAVIDES, L., PESCADOR, D.S., SÁNCHEZ, A.M., CAVIERES, L.A. and ESCUDERO, A., 2016. Soil under nurse plants is always better than outside: a survey on soil amelioration by a complete guild of nurse plants across a long environmental gradient. *Plant and Soil,* **408**(1), pp. 31-41.

MILLS, A. and FEY, M., 2004. Transformation of thicket to savanna reduces soil quality in the Eastern Cape, South Africa. *Plant and Soil,* **265**(1), pp. 153-163.

MILLS, A.J., O'CONNOR, T.G., DONALDSON, J.S., FEY, M.V., SKOWNO, A.L., SIGWELA, A.M., LECHMERE-OERTEL, R.G. and BOSENBERG, J.D., 2005a. Ecosystem carbon storage under different land uses in three semi-arid shrublands and a mesic grassland in South Africa. *South African Journal of Plant and Soil,* **22**(3), pp.

183-190.

MILLS, A.J., COWLING, R.M., FEY, M.V., KERLEY, G., DONALDSON, J.S., LECHMERE-OERTEL, R.G., SIGWELA, A.M., SKOWNO, A.L. and RUNDEL, P., 2005b. Effects of goat pastoralism on ecosystem carbon storage in semiarid thicket, Eastern Cape, South Africa. *Australian Ecology,* **30**(7), pp. 797-804.

MILLS, A.J. and COWLING, R.M., 2006. Rate of carbon sequestration at two thicket restoration sites in the Eastern Cape, South Africa. *Restoration Ecology,* **14**(1), pp. 38-49.

MILLS, A.J., TURPIE, J.K., COWLING, R.M., MARAIS, C., KERLEY, G.I., LECHMERE-OERTEL, R.G., SIGWELA, A.M. and POWELL, M., 2007. Assessing costs, benefits, and feasibility of restoring natural capital in subtropical thicket in South Africa. *Restoring natural capital. Science, business and practice.* Island Press, Washington DC, United States of America.

MILLS, A.J., SPEAR, D., NDHLOVU, T., KNIGHT, M.H., PEINKE, D.M., HOFMEYR, S. and COWLING, R.M., 2014. Biomass of large herbivores in South African subtropical thicket. *African Journal of Ecology,* **52**(4), pp. 577-580.

MILLS, A.J., VYVER, M.V.D., GORDON, I.J., PATWARDHAN, A., MARAIS, C., BLIGNAUT, J., SIGWELA, A. and KGOPE, B., 2015. Prescribing innovation within a large-scale restoration programme in degraded subtropical thicket in South Africa. *Forests,* **6**(11), pp. 4328-4348.

MILLS, A.J. and ROBSON, A., 2017. Survivorship of spekboom (*Portulacaria afra*) planted within the Subtropical Thicket Restoration Programme. *South African Journal of Science,* **113**(1-2), pp. 1-3.

MILLS, A.J., MILTON, S.J., TAPLIN, B.D. and ALLEN, J.L., 2018. Viability of *watering Portulacaria afra* truncheons to facilitate restoration of subtropical thicket: Results from a nursery experiment and cost model. *South African Journal of Botany*, **115**, pp.58-64.

MINNIE, L., BOSHOFF, A.F. and KERLEY, G.I., 2015. Vegetation type influences livestock predation by leopards: implications for conservation in agro- ecosystems. *African Journal of Wildlife Research,* **45**(2), pp. 204-214.

MITTERMEIER, R.A., TURNER, W.R., LARSEN, F.W., BROOKS, T.M. and GASCON, C., 2011. Global biodiversity conservation: the critical role of hotspots. *Biodiversity hotspots.* Springer, pp. 3-22.

MIRALLES, D.G., HOLMES, T.R.H., DE JEU R.A.M., GASH, J.H., MEESTERS, A.G.C.A. and DOLMAN, A.J. Global land-surface evaporation estimated from satellite-based observations. *Hydrology and Earth System Sciences*., **15** (2) (2011), pp. 453-469

MOTIS, T., D'AIUTO, C. & LINGBEEK, B., 2013. Zai pit system. ECHO Technical Note **78**, North Fort Myers, Florida, United States of America, pp. 9.

MUCHANDO, A., 2019. Remote sensing assessment of land restoration interventions in South Africa. Master's thesis, University of Twente, Netherlands.

MUIR, R.A., BORDY, E.M., REDDERING, J. and VILJOEN, J., 2017. Lithostratigraphy of the Enon Formation (Uitenhage Group), South Africa. *South African Journal of Geology,* **120**(2), pp. 273-280.

MUNRO, N.T., LINDENMAYER, D.B. and FISCHER, J., 2007. Faunal response to revegetation in agricultural areas of Australia: a review. *Ecological Management & Restoration,* **8**(3), pp. 199-207.

MURAKAMI, P. F. 2005. An instructional guide for leaf color analysis using digital imaging software. US Department of Agriculture, Forest Service, Northeastern Research Station.

NATHANIEL, J.A., 2020. Biology 4920G: Companion Planting in the Community. Western University, Ontario, Canada, pp. 11. [Online] Available at: https://ir.lib.uwo.ca/se-ccel/30/ Accessed on: January 2022

NAVARRO-CANO, J.A., GOBERNA, M., VALIENTE-BANUET, A. and VERDÚ, M., 2016. Same nurse but different time: temporal divergence in the facilitation of plant lineages with contrasted functional syndromes. *Functional Ecology,* **30**(11), pp. 1854-1861.

NAVARRO-CANO, J.A., GOBERNA, M. and VERDÚ, M., 2019. Using plant functional distances to select species for restoration of mining sites. *Journal of Applied Ecology,* **56**(10), pp. 2353-2362.

NAVARRO-CANO, J.A., VERDÚ, M. and GOBERNA, M., 2018. Trait-based selection of nurse plants to restore ecosystem functions in mine tailings. *Journal of Applied Ecology,* **55**(3), pp. 1195-1206.

NORMAN, Y.M. 2016. Effects of rooting hormone, stem scarification and time of planting on root biomass of *Portulacaria afra* truncheons. Unpublished honours thesis, Rhodes University, Grahamstown, South Africa. pp.1-20.

OAKES, A.J., 1973. *Portulacaria afra* Jacq.: a potential browse plant. *Economic Botany,* pp. 413-416.

O'BRIEN, M.J., PUGNAIRE, F.I., ARMAS, C., RODRÍGUEZ-ECHEVERRÍA, S. and SCHÖB, C., 2017. The shift from plant-plant facilitation to competition under severe water deficit is spatially explicit. *Ecology and evolution*, **7**(7), pp.2441-2448.

OREJA, B., GOBERNA, M., VERDÚ, M. and NAVARRO-CANO, J.A., 2020. Constructed pine log piles facilitate plant establishment in mining drylands. *Journal of environmental management,* **271**, pp. 11-1015.

PADILLA, F.M. and PUGNAIRE, F.I., 2006. The role of nurse plants in the restoration of degraded environments. *Frontiers in Ecology and the Environment,* **4**(4), pp. 196-202.

PANTER, B. and RUWANZA, S., 2019. Spekboom (*Portulacaria afra*) planting in degraded thickets improves soil properties and vegetation diversity. *Ecological Restoration,* **37**(2), pp. 76-80.

PARNELL, S. and WALAWEGE, R., 2011. Sub-Saharan African urbanisation and global environmental change. Global Environmental Change, **21**, pp. 12-20.

PETERSON, G., ALLEN, C.R. and HOLLING, C.S., 1998. Ecological resilience, biodiversity, and scale. *Ecosystems,* **1**(1), pp. 6-18.

PHILLIPS, S. 2004. The Expanded Public Works Programme (EPWP). Presentation to the UNDP, HSRC & DBSA Conference on Overcoming Under-development in South Africa's Second Economy, Pretoria, South Africa, 29 October 2004.

POTTS, A.J., 2016. Spekboom restoration: the rooting window hypothesis. Available: Pottsresearch.blogspot.co.za. URL: http://pottsresearch.blogspot.co.za/2016/12/spekboom-rehabilitation-rooting-window.html [Accessed, 18 June 2017].

POWELL, M.J., 2009. Restoration of degraded subtropical thickets in the Baviaanskloof Megareserve, South Africa. The role of carbon stocks and *Portulacaria afra* survivorship. Master's thesis. Rhodes University, Grahamstown, South Africa, pp 151.

PRIETO, P., PENUELAS, J., LLUSIA, J., ASENSIO, D. and ESTIARTE, M., 2009. Effects of long-term experimental night-time warming and drought on photosynthesis, Fv/Fm and stomatal conductance in the dominant species of a Mediterranean shrubland. *Acta Physiologiae Plantarum,* **31**(4), pp. 729-739.

QUÉTIER, F., THÉBAULT, A. and LAVOREL, S., 2007. Plant traits in a state and transition framework as markers of ecosystem response to land-use change. *Ecological Monographs,* **77**(1), pp. 33-52.

RAGIMANA, M.B. and POTTS, A.J., 2019. Planting orientation of *Portulacaria afra* cuttings for thicket restoration: vertical versus horizontal, (No. e27543v1), PeerJ Preprints.

R CORE TEAM, 2021. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

REIJ, C. and GARRITY, D., 2016. Scaling up farmer-managed natural regeneration in Africa to restore degraded landscapes. *Biotropica,* **48**(6), pp. 834-843.

REN, H., YANG, L. and LIU, N., 2008. Nurse plant theory and its application in ecological restoration in lower subtropics of China. *Progress in Natural Science,* **18**(2), pp. 137-142.

RIETKERK, M. and VAN DE KOPPEL, J., 1997. Alternate stable states and threshold effects in semi-arid grazing systems. *Oikos*. pp. 69-76.

ROQUE MARCA, N., LÓPEZ, R.P. AND NAOKI, K., 2021. Effect of shade and precipitation on germination and seedling establishment of dominant plant species in an Andean arid region, the Bolivian Prepuna. *Plos one*, **16**(3), pp.0248619.

ROSADO, B.H., DIAS, A.T. and DE MATTOS, E.A., 2013. Going back to basics: importance of ecophysiology when choosing functional traits for studying communities and ecosystems. *Natureza & Conservação,* **11**(1), pp. 15-22.

RUBIN, R.L., KOCH, G.W., MARTINEZ, A., MAU, R.L., BOWKER, M.A. and HUNGATE, B.A., 2018. Developing climate-smart restoration: Can plant microbiomes be hardened against heat waves? *Ecological Applications,* **28**(6), pp. 1594-1605.

RUSCH, G.M., SKARPE, C. and HALLEY, D.J., 2009. Plant traits link hypothesis about resource-use and response to herbivory. *Basic and Applied Ecology*, **10(**5), pp.466-474.

SCHÖB, C., ARMAS, C. and PUGNAIRE, F.I., 2013. Direct and indirect interactions co-determine species composition in nurse plant systems. *Oikos,* **122**(9), pp. 1371-1379.

SCHOLES, R.J., 2009. Syndromes of dryland degradation in southern Africa. *African Journal of Range & Forage Science,* **26**(3), pp. 113-125.

SCHOLES, R.J., MONTANARELLA, L., BRAINICH, E., BARGER, N., TEN BRINK, B., CANTELE, M., ERASMUS, B., FISHER, J., GARDNER, T. and HOLLAND, T.G., 2018. IPBES (2018): Summary for policymakers of the assessment report on land degradation and restoration of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.

SCHWILCH, G., LINIGER, H.P. and HURNI, H., 2014. Sustainable land management (SLM) practices in drylands: how do they address desertification threats? *Environmental management,* **54**(5), pp. 983-1004.

SHACKLETON, C.M. and GAMBIZA, J., 2008. Social and ecological trade offs in combating land degradation: the case of invasion by a woody shrub (Euryops floribundus) at Macubeni, South Africa. *Land Degradation & Development,* **19**(4), pp. 454-464.

SHOLTO-DOUGLAS, C., 2019. Factors influencing survivorship of *Portulacaria afra* cuttings in restoration of degraded lands. Master's thesis. Rhodes University, Grahamstown, South Africa. pp. 1-104.

SIGWELA, A.M., KERLEY, G., MILLS, A.J. and COWLING, R.M., 2009. The impact of browsing-induced degradation on the reproduction of subtropical thicket canopy shrubs and trees. *South African Journal of Botany,* **75**(2), pp. 262-267.

SMART, K.G., 2016. The role of vegetation in regional climate regulation feedback processes (Doctoral dissertation). Department of Botany. Rhodes University, Grahamstown, South Africa.

SMIT, C., DEN OUDEN, J. and MÜLLER-SCHÄRER, H., 2006. Unpalatable plants facilitate tree sapling survival in wooded pastures. *Journal of Applied Ecology,* **43**(2), pp. 305-312.

SMITH, N. and WILSON, S.L., 2002. Changing land use trends in the thicket biome: pastoralism to game farming. Port Elizabeth, South Africa: Terrestrial Ecology Research Unit, University of Port Elizabeth.p*,* **29**.

SOCIETY FOR ECOLOGICAL RESTORATION INTERNATIONAL, 2004. SER international primer on ecological restoration. Society for Ecological Restoration International.

SOUTH AFRICAN NATIONAL BIODIVERSITY INSTITUTE. 2018. The Vegetation Map of South Africa, Lesotho and Swaziland, Mucina, L., Rutherford, M.C. and Powrie, L.W. (Editors), Online, http://bgis.sanbi.org/Projects/Detail/186, Version 2018.

STARKEY, R. 2015. Effect of herbivory on below-ground root carbon storage of Spekboom (*Portulacaria afra*). Baviaanskloof Nature Reserve, Eastern Cape, South Africa. Unpublished honours thesis, Rhodes University, Grahamstown, South Africa.

STATS, S.A., 2018. Men, women and children. Findings of the Living Conditions Survey, 2014/15 (Vol. 15). Report No. 03-10-02 2014. Available at: https://www.statssa.gov.za/publications/Report-03-10-02%20/Report-03-10-02%202015.pdf Accessed on: January 2022

STATS, S.A., 2021. Statistical Release P0211 Quarterly Labour Force Survey. Quarter 2. [Online] Available at: https://www.statssa.gov.za/publications/P0211/P02112ndQuarter2021.pdf Accessed on: January 2022

STATSOFT, INC. 2012. Electronic Statistics Textbook. Tulsa, OK: StatSoft. WEB: http://www.statsoft.com/textbook/.

STAVI, I., LAVEE, H., UNGAR, E.D. and SARAH, P., 2009. Ecogeomorphic feedbacks in semiarid rangelands: A review. *Pedosphere,* **19**(2), pp. 217-229.

STRINGHAM, T.K., KRUEGER, W.C. and SHAVER, P.L., 2003. State and transition modeling: an ecological process approach. *Rangeland Ecology & Management/Journal of Range Management Archives,* **56**(2), pp. 106-113.

STUART-HILL, G.C. and AUCAMP, A.J., 1993. Carrying capacity of the succulent valley bushveld of the Eastern Cape. *African Journal of Range & Forage Science,* **10**(1), pp. 1-10.

STUART-HILL, G.C., 1992. Effects of elephants and goats on the Kaffrarian succulent thicket of the eastern Cape, South Africa. *Journal of Applied Ecology*, pp. 699-710.

SUDING, K.N., GROSS, K.L. and HOUSEMAN, G.R., 2004. Alternative states and positive feedbacks in restoration ecology. *Trends in ecology & evolution,* **19**(1), pp. 46-53.

SWART, M., HOBSON, F.O. and STUART-HILL, G.C., 1994. Establishment of spekboom. *Dohne Bulletin,* **3**(1), pp. 10-13.

TAPLIN, B. 2019. An Introduction to Soil Degradation and Practical Rehabilitation Guidelines Conservation And Management Services. Unpublished manual. Conservation Management Services. Rhodes University. South Africa.

THUROW, T.L., 2000. Hydrologic effects on rangeland degradation and restoration processes. *Rangeland desertification.* Springer, pp. 53-66.

TING, I.P. and HANSCOM III, Z., 1977. Induction of acid metabolism in *Portulacaria afra*. *Plant Physiology,* **59**(3), pp. 511-514.

TODKILL, W.B., KERLEY, G. and CAMPBELL, E.E., 2006. Brushpiles and dung as rehabilitation patches: effect on soil resources in degraded succulent thicket, Eastern Cape, South Africa. *African Journal of Range and Forage Science,* **23**(1), pp. 39-48.

TONGWAY, D.J. and LUDWIG, J.A., 1996. Rehabilitation of semiarid landscapes in Australia. I. Restoring productive soil patches. *Restoration Ecology,* **4**(4), pp. 388- 397.

TROLLOPE, W.E., TROLLOPE, L.A. and BOSCH, O.J.H., 1990. Veld and pasture management terminology in southern Africa. *Journal of the Grassland Society of Southern Africa.* **7**(1), pp. 52-61.

TRUSWELL, J.F., 1970. An Introduction to the Historical Geology of South Africa: Purnell. *Cape Town, 167p.*

TURNBULL, L., WAINWRIGHT, J. and BRAZIER, R.E., 2008. A conceptual framework for understanding semi-arid land degradation: Ecohydrological interactions across multiple-space and time scales. *Ecohydrology: Ecosystems, Land and Water Process Interactions, Ecohydrogeomorphology,* **1**(1), pp. 23-34.

UNCCD. 2016. Land Degradation Neutrality Target Setting –A Technical Guide. Bonn, Germany: Global Mechanism of the UNCCD, pp.1-64

VALLIERE, J.M., ZHANG, J., SHARIFI, M.R. and RUNDEL, P.W., 2019. Can we condition native plants to increase drought tolerance and improve restoration success? *Ecological Applications,* **29**(3), pp. e01863.

VAN DER VYVER, M.L., 2011. Restoring the biodiversity of canopy species within degraded spekboom thicket. Doctoral dissertation, Nelson Mandela Metropolitan University, Port Elizabeth. South Africa.

VAN DER VYVER, MARIUS L, COWLING, R.M., CAMPBELL, E.E. and DIFFORD, M., 2012. Active restoration of woody canopy dominants in degraded South African semi-arid thicket is neither ecologically nor economically feasible. *Applied Vegetation Science,* **15**(1), pp. 26-34.

VAN DER VYVER, MARIUS L, COWLING, R.M., MILLS, A.J. and DIFFORD, M., 2013. Spontaneous return of biodiversity in restored subtropical thicket: *Portulacaria afra* as an ecosystem engineer. *Restoration Ecology,* **21**(6), pp. 736-744.

VAN DER VYVER, MARIUS LODEWYK, 2018. Factors affecting effective ecological restoration of *Portulacaria afra* (spekboom)-rich subtropical thicket and aboveground carbon endpoint projections. Nelson Mandela University, Port Elizabeth, South Africa.

VAN DER VYVER, MARIUS L and COWLING, R.M., 2019. Aboveground biomass and carbon pool estimates of *Portulacaria afra* (spekboom)-rich subtropical thicket with species-specific allometric models. *Forest Ecology and Management,* **448**, pp. 11-21.

VAN DER VYVER, MARIUS L, MILLS, A.J. and COWLING, R.M., 2021. A biome-wide experiment to assess the effects of propagule size and treatment on the survival of *Portulacaria afra* (spekboom) truncheons planted to restore degraded subtropical thicket of South Africa. *PloS one,* **16**(4), pp. e0250256.

VAN JAARSVELD, E. and LE ROUX, A., 2021. *Portulacaria afra* (L.) Jacq.: Variability and distribution. *Bradleya,* **2021**(39), pp. 138-152.

VAN LUIJK, G., COWLING, R.M., RIKSEN, M. and GLENDAY, J., 2013. Hydrological implications of desertification: Degradation of South African semi-arid subtropical thicket. *Journal of Arid Environments,* **91**, pp. 14-21.

VAN OUDTSHOORN, F., 2015. Veld management: principles and practices. Briza. Cape Town, South Africa.

VISSER, N., MORRIS, C., HARDY, M.B. and BOTHA, J.C., 2007. Restoring bare patches in the Nama-Karoo of South Africa. *African Journal of Range and Forage Science,* **24**(2), pp. 87-96.

VITALE, L.; VITALE, E.; COSTANZO, G.; DE MAIO, A. and ARENA, C. 2020. Photo-protective mechanisms and the role of poly (ADP-ribose) polymerase activity in a facultative CAM plant exposed to long-term water deprivation. *Plants*, **9**, pp. 1192.

VLOK, J. and EUSTON-BROWN, D., 2002. The patterns within, and the ecological processes that sustain, the subtropical thicket vegetation in the planning domain for the Subtropical Thicket Ecosystem Planning (STEP) project. *Terrestrial Ecology Research Unit report,* **40**, pp. 142.

VLOK, J., EUSTON-BROWN, D., COWLING, R.M., and HOFFMAN, M.T.,2003. Acocks' Valley Bushveld 50 years on: new perspectives on the delimitation, characterisation and origin of subtropical thicket vegetation. *South African Journal of Botany,* **69**(1), pp. 27-51.

VOHLAND, K. and BARRY, B., 2009. A review of in situ rainwater harvesting (RWH) practices modifying landscape functions in African drylands. *Agriculture, Ecosystems & Environment,* **131**(3-4), pp. 119-127.

WESTOBY, M., WALKER, B. and NOY-MEIR, I., 1989. Opportunistic management for rangelands not at equilibrium. *Rangeland Ecology & Management/Journal ofRange Management Archives,* **42**(4), pp. 266-274.

WHISENANT, S., 1999. Repairing damaged wildlands: a process-orientated, landscape-scale approach. Cambridge University Press.

WOCAT. 2007. Where the land is greener—case studies and analysis of soil and water conservation initiatives worldwide. Liniger HP, Critchley W (eds) CTA, FAO, UNEP, CDE, Berne. Switzerland.

WORTLEY, L., HERO, J.M. and HOWES, M., 2013. Evaluating ecological restoration success: A review of the literature. Restoration Ecology. **21**(5), pp.537- 543.

ZHUANG, X.Y. and GORLETT, R.T., 1997. Forest and forest succession in Hong Kong, China. *Journal of Tropical Ecology,* **13**(6), pp. 857-866.

Personal communications

Beukes, M. 2017. Conversation with Otto and Maya Beukes, September 2017 (Living Lands)

Beukes, O. 2017. Conversation with Otto and Maya Beukes September 2017, (Living Lands)

Allen, M. 2017. Conversation with Melloson Allen, September 2017 (Living Lands)