Soil and vegetation recovery following *Acacia dealbata* clearing in the Tsitsa catchment, Eastern Cape Province of South Africa: Implications for ecological restoration

By

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Abstract

Invasion by alien plant species in South Africa continues to compromise the stability of ecosystems by causing declines in biodiversity, altering soil nutrients and processes, and subsequently transforming ecosystem functionality. Control of invasive alien plant species has been widely implemented in South Africa to minimize their negative impacts; however, the legacy effects can persist long after the plant has been removed.

The impacts of *Acacia dealbata* clearing on soil properties and native vegetation recovery remains understudied despite their significance in ecological restoration and monitoring. This comparative study determined the impacts of *A. dealbata* clearing on both soil physicochemical properties and vegetation in the Eastern Cape Province of South Africa. Soils were collected from three different clearing treatments, namely, cleared, invaded, and uninvaded, on 5 m x 5 m plots over three summer months. The plots were replicated four times for each clearing treatment, making a total of 72 sampling plots. Soils were assessed for soil pH, resistivity, P, C, N, and exchangeable cations as well as soil moisture content, penetration resistance, infiltration rate, hydraulic conductivity, and water repellency. Clearing of *A. dealbata* did not have any significant effects on most soil nutrients, however, there were variations in soil pH, resistance, and Na. Soil pH was significantly higher in the uninvaded treatments than in the cleared and invaded treatments. Soil moisture content was significantly higher in the cleared treatments than the adjacent invaded and uninvaded treatments, but this was observed in the month of December only. Soil penetration resistance and infiltration rates were significantly higher in the month of December in the cleared treatments. For all clearing treatments, no significant differences were recorded for soil hydraulic conductivity. These results on changes in soil properties following *A. dealbata* clearing are varied, with some soil properties showing decreases, an indication that removal of *A. dealbata* has the potential to shift soil properties towards a positive recovery trajectory.

This study further assessed whether the clearing of *A. dealbata* facilitates the recovery of native plant species. Vegetation surveys were conducted in the three above-mentioned treatments and plots. Results showed little recruitment of native grasses and forbs, but the persistence of *A. dealbata* seedlings in the cleared treatments. Species richness and Shannon-Wiener diversity index were significantly ($P < 0.05$) higher in the cleared and invaded treatments than the uninvaded treatments, and this was more visible for trees and shrubs. Cover for all species was significantly higher ($P < 0.05$) in the uninvaded than the cleared and invaded treatments. This study observed the recruitment of some native species in the cleared treatments that were not present in the invaded treatments. Therefore, the recruitment and establishment of some native species, mostly grasses, in the cleared treatments gives assurance that passive restoration is on a positive vegetation recovery trajectory that can lead to recovery of native vegetation after *A. dealbata* clearing. Therefore, the study concludes that investing in ecological restoration after alien plant clearing is a necessity for complete ecosystem recovery to be achieved.

Overall, the study concludes that the removal of *A. dealbata* triggers changes to some soil properties. Similarly, the study observed recruitment of some native grasses in cleared areas, an indication that alien plant clearing facilitates changes in both soil properties and vegetation. However, soil and vegetation recovery are being hampered by the regrowth of *A. dealbata* and secondary invaders that were observed in the cleared treatments. Two key recommendations of this study are (i) clearing follow-up to remove recruiting seedlings of invasive plant species and secondary invaders should be timeous and well-funded, and (ii) active restoration should be considered to speed-up soil and vegetation recovery processes.

Key words: Invasive alien plants, post-clearing, soil nutrients, soil and vegetation recovery, legacy effect, Working for Water.

Declaration

I Putuma Balintulo, hereby declare that the work outlined in this thesis was carried out at Rhodes University, Department of Environmental Science under the supervision of Dr. Sheunesu Ruwanza. This thesis has not been submitted at this university or any other university, this is my work in design and execution, and all material contained herein has been duly acknowledged.

Signature:____________________ **Date:** 07/03/2022

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The contents of this MSc thesis do not necessarily reflect the view and policies of the DFFE, Chief Directorate: NRM, nor does the mention of trade names or commercial products constitute endorsement or recommendation for use.

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Chapter 1: General introduction

1.1. Introduction

South African grassland ecosystems are under threat from invasion by woody invasive alien plants (IAP) and that is compromising the growth of native grass species which are important for grazing (Gwate et al., 2021). Little et al. (2015) reported that about 60% of the grassland biome in South Africa has been permanently transformed due to several factors, including invasion by woody alien plants, land use change, and urbanization. The accelerated encroachment and spread by IAP has resulted in massive declines in biodiversity, changes in soil nutrients, and subsequently changes in ecosystem structure and function (Ehrenfeld, 2003; Gaertner et al., 2011). Australian *Acacias* are amongst the most damaging IAP that invade South Africa and they transform ecosystems by stimulating changes in soil properties, displacement of native vegetation, and changes in hydrology (Ehrenfeld, 2003; Yelenik et al., 2004; Le Maitre et al., 2011). This ultimately compromises the ability of ecosystems to produce and deliver ecosystem services such as water provisioning as well as reducing grazing for livestock and wildlife grazing, which in turn threaten economic productivity (Kull et al., 2008; Kerr and Ruwanza, 2016; Yapi et al., 2018; Ruwanza and Tshililo, 2019). Despite these negative impacts, some communities still derive benefits from IAP such as poles for building, fencing, and fuel (Ngorima, 2016), even so, there is still consensus for them to be effectively managed and removed (Kull et al., 2011; Gwate et al., 2021). This seems to suggest that some IAP are conflict-generating species, therefore detailed socio-ecological studies are required to develop effective management strategies of such conflicting species (Zengeya et al., 2017; Novoa et al., 2018). Management of conflict-generating species is challenging, especially when the people that benefit from the species are the ones that incur the costs (Novoa et al., 2018)

1.1.1. Extent and impacts of plant invasion in South Africa

For effective control of IAP, there should be profound understanding of the spatial extent of invasion, the status of invasion, drivers of invasion that lead to changes in the functioning of ecosystems, and the social and ecological impacts of the invading species (Le Maitre et al., 2011; Ngorima, 2016; Magona et al., 2017). Having this up-to-date data is crucial to ensure that appropriate management approaches are implemented to control the IAP at all stages of introduction-naturalization-invasion continuum (Magona et al., 2017). Richardson et al. (2011) reported that 70 Australian *Acacias* had been introduced in South Africa since the 1830s, however, this number was revised to 141 *Acacias* by Magona et al. (2017), of which only 33 species are still present in the country. The species list was revised because the introduction of *Acacias* has been beneficial in terms of economic growth (Griffin at al., 2011; Richardson et al., 2011) and sustainability of livelihoods of communities (Kull et al., 2011; van Wilgen et al., 2011). However, their spread beyond formal plantations have made these species invasive and difficult to manage; moreover, the species continue to increase in geographical extent and magnitude of impacts (Wilson et al., 2011; Henderson and Wilson, 2017; Mogona et al., 2017).

Australian *Acacias* thrive in the alien range mainly because of their ability to outcompete native species in terms of accessing nutrients such as water, light, and soil nutrients (Turpie et al., 2008; Gwate et al., 2021). Additionally, *Acacias* transform ecosystems by altering the quality and quantity of resources such as soil nutrients accessible to native species (Lazzaro et al., 2014), as well as changes in water availability, and fire regimes (Yelenik et al., 2004). For example, Yelenik et al. (2004) states that the introduction of the N-fixing *Acacias* accelerates the N-fixing process, thus injecting large quantities of N into the soil, which causes declines in growth rates of some native species. Lazzaro et al. (2014) reported a decrease in species richness and diversity in areas that are invaded by *A. dealbata* because of the increased competition. Furthermore, invasion by *Acacias* suppresses the survival of soil-stored seed banks of native species, thereby leading to local extinction of native species (Richardson and van Wilgen, 2004). Changes in soil properties and vegetation composition negatively impacts ecosystem structure and function (Yelenik et al., 2004; Lazzaro et al., 2014).

Besides the ecological impacts, IAP, particularly *Acacias* also threaten hydrological processes. Invasion of hillslopes by IAP is detrimental to water availability and water flow processes because mountainous areas are important sources of runoff (Mkunyana et al., 2019). Mkunyana et al. (2019) quantified the differences in water use by *Acacia* species in riparian zones and hillslopes in the Western Cape Province of South Africa and reported that invasion by IAP results in decreased runoff which has adverse impacts on water availability downstream. Le Miatre et al. (2020) reported that the impact of IAP on surface runoff is approximately 1.44 – 2.44 billion m^3 per year. Scott-Shaw and Everson (2018) assessed water use dynamics of alien invaded riparian forests on a summer rainfall zone and revealed that invaded sites use up to 40% more water per unit areas as compared to natural sites. More losses in water availability, especially in mountainous grasslands may be due to increased severity of fires. Richardson and van Wilgen (2004) state that increased litter fall and fuel loads from IAP intensifies fire hazards and soil erosion, which in turn compromises the ability of catchment areas to store water for gradual release throughout the seasons. Increased fuel loads also influence the habits of fires leading to changes in fire regimes such as frequency, intensity, magnitude, type, and seasonality (Brooks et al., 2004). Some IAP would then take advantage of the changes in fire regimes and dominate, ultimately leading to the invaded area adopting the invasive plant's fire regime cycle (Brooks et al., 2004). Once above threshold densities, the level to which the ecosystem has been severely impacted, have been reached, IAP erode natural capital and ecosystem stability and the impacts can become permanent (Ruwanza and Shackleton, 2016; Gwate et al., 2021), thus appropriate management approaches need to be implemented to control the establishment and spread of IAP and restore cleared areas to pristine conditions.

1.1.2. Restoration after clearing; the Working for Water programme

Due to the severity of the impacts by IAP, the South African Department of Forestry, Fisheries and Environment (formerly known as Department of Environmental Affairs) implemented the Working for Water (WfW) programme, which is a nation-wide alien plant control project that is aimed at controlling the spread of IAP (van Wilgen et al., 1998; Yapi et al., 2018). The programme was established mainly for the conservation of water resources that were threatened by IAP (Yapi et al., 2018), while simultaneously creating job opportunities since it is a labour-intensive programme (van Wilgen et al., 1998). The premise of the WfW is that after clearing of IAP ecosystems will spontaneously recover to their natural state, i.e., passive restoration (Turpie et al., 2008; Kull et al., 2011; van Wilgen et al., 2012). However, several empirical studies have revealed that the clearing of IAP by the WfW has not been successful in limiting the rate of re-invasion as well as stimulating the recovery of soil properties and native vegetation in the cleared areas (van Wilgen et al., 2012; Ruwanza et al., 2013; Ndou and Ruwanza, 2016; Nsikani et al., 2017; Ruwanza and Tshililo, 2019). Therefore, there has been growing scrutiny on the recovery of ecosystems following the clearing of IAP with extensive work being done in the fynbos biome of the Western Cape Province of South Africa (Yelenik et al., 2004; Nsikani et al., 2017), however, little has been done to assess the recovery of the ecosystem in the grassland biome.

It is crucial to have insights on the recovery trajectories of soil properties and native vegetation species after the clearing of IAP because it assists in quantifying the impacts and the benefits of clearing (Ndhlovu et al., 2016; Yapi et al., 2018). Although some studies observed no native species recovery after passive restoration by WfW (Ruwanza, 2013), some studies recorded recovery of some soil properties and some native plant species on the cleared areas (Gaertner et al., 2011, Fill et al., 2017; Mangachena and Geert, 2019: Ruwanza and Tshililo, 2019). Pretorius (2008) reported the growth of some native species after the clearing of *Acacias*, on the other hand, Ndou and Ruwanza (2016) observed that changes in soil and vegetation after clearing are also affected by time since clearing. Although there has been a growing interest in assessing soil and vegetation after the removal of *Acacia* species in South Africa, there is still limited studies on investigating soil and vegetation recovery trajectories after clearing of *A. dealbata,* especially in grassland ecosystems where the species is prevalent (Ruwanza et al., 2013; Ndou and Ruwanza, 2016; Ngorima, 2016; Nsikani et al., 2017; Yapi et al., 2018; Ruwanza and Tshililo, 2019). Since previous studies have shown that IAP alter ecosystem properties (Gaertner et al., 2011; Fill et al., 2017; Nsikani et al., 2017), this means that clearing needs to be supplemented by active restoration, therefore future research needs to be done investigating possible recovery options. (Holmes et al., 2020)

1.2. Motivation for the study

Over 20 years ago, South Africa responded to the threat by IAP by launching the WfW programme, yet little is known regarding how soil properties and native vegetation species recovery after alien plant clearing (van Wilgen, 2020). Most research done on the recovery of soil properties and vegetation following *Acacia* clearing e.g., *A. mearnsii, A. longifilia* and *A. saligna* indicates that residuals and/or the legacy effect of invasion remain in the soil long after clearing and that vegetation recovery is site-specific (Ruwanza et al., 2013; Ndou and Ruwanza, 2016; Ruwanza and Tshililo, 2019). It is, therefore, important that soil and vegetation recovery studies are done in different biomes, to inform site-specific management interventions. To my knowledge, few studies have been done to assess both soil and vegetation recovery following *A. dealbata* clearing in grasslands. *Acacia dealbata* is one of the most proliferating invaders in the Eastern Cape Province of South Africa, and it is a threat to grazing ecosystems and in water sources (Yapi et al., 2018). Therefore, the motivation of this study is to investigate and compare changes in both soil properties and vegetation recovery following the clearing of *A. dealbata* in communal grassland ecosystem in the Eastern Cape Province of South Africa. Comparing cleared, invaded, and uninvaded areas will be useful in theorizing ecological restoration trajectories and inform managers on how to facilitate restoration of soil and native vegetation properties after alien plant clearing (Ruwanza and Tshililo, 2019).

1.3. Study area

The study was conducted at the Upper Tsitsana communal village, Nqanqarhu (formerly known as Maclear) in the northern parts of Eastern Cape Province of South Africa (30° 53' 19" S, 28° 26' 23" E) (Figure 1.1). The villages lie in the Tsitsa quaternary catchment which forms part of the greater Mzimvubu water management area (Gwate et al., 2021). The Tsitsa quaternary catchment is situated in the Southern Drakensberg Highland Grassland (SDHG) (Mucina and Rutherford, 2006). It spreads across the moist grasslands in the high altitudes that are associated with a high-water supply and soils with low base status (Mucina and Rutherford, 2006). Indigenous grasses that dominate the area include *Sporobolus* species, *Cynodon dactylon* and *Eragrostis* species (Gwate et al., 2021). Invasive Australian *Acacias*, particularly *A. dealbata* forms thickets and monostands on the hillslopes, abandoned fields, and along meandering rivers (Ngorima et al., 2016). The area is underlain by sandstones of the Clarens formations and sandstones, siltstones, and mudstones of the Elliot formation as well as the basalts of the Drakensberg Group with deep and fine-grained soils (Mucina and Rutherford, 2006). The area is characterized by hot summers with an average temperature of 25 ºC while winters are cold and commonly have snow in the mountains and the average temperature drops to 14 ºC (Pretorius, 2016). The Tsitsa catchment receives an annual rainfall of 850 mm but marks an average of 130 mm in January which is a peak month and is mostly thunderstorms (Pretorius, 2016).

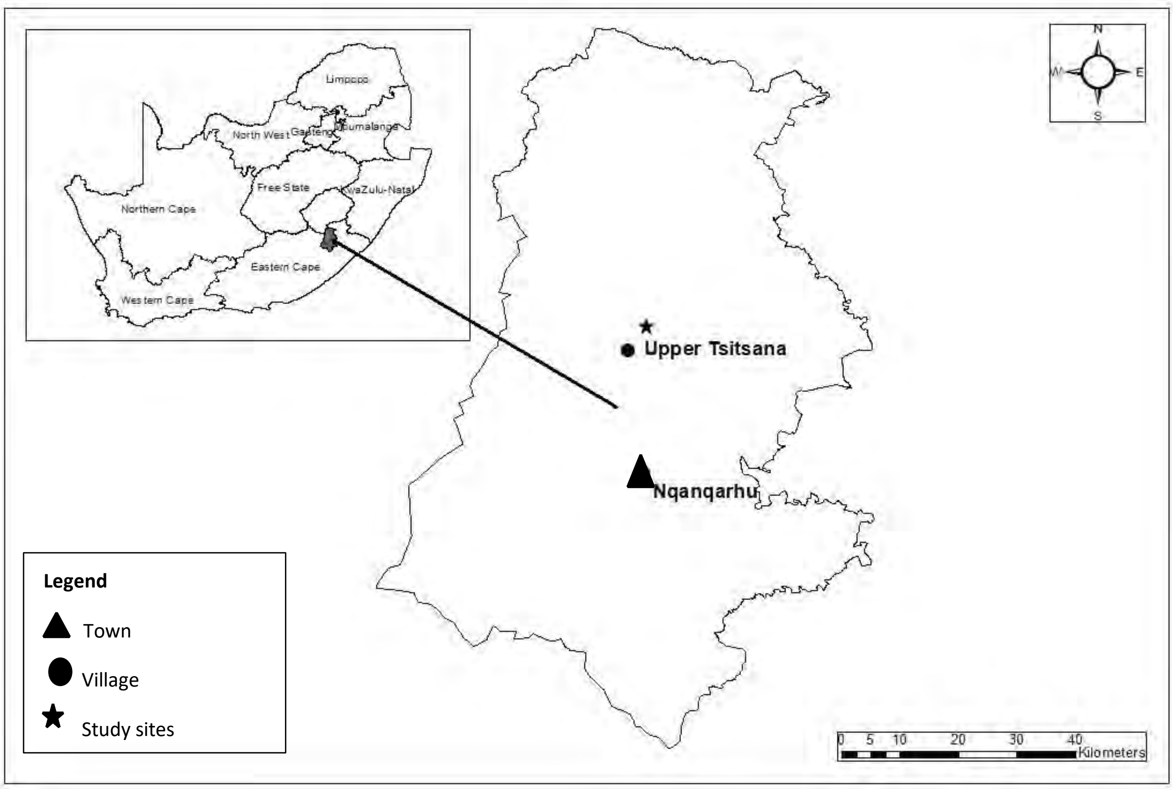

Figure 1.1 Map showing the location of the Upper Tsitsana village in Nqanqarhu town in the Eastern Cape Province of South Africa.

1.3.1. Site description and experimental design

The communal village in which the study was conducted lies in the former Transkei homelands (Palmer and Bennett, 2013). As a result, the area has been historically (and presently) overcrowded and overstocked since 1913 (Palmer and Bennett, 2013). Invasion by IAP in

South Africa can be dated back to the 19th century (van Wilgen et al., 2011), and the Tsitsa quaternary catchment has not been immune to this invasion. This study was conducted in the *A. dealbata* invaded areas of the Tsitsa quaternary catchment that were adjacent to uninvaded grassland in the areas. The alien plant mapping that was conducted by Huchzermeyer et al. (2018) formed part of the site selection process and Google Earth as well as aerial photos were used to complement the identification of cleared and invaded sites. *Acacia dealbata* invaded areas were dominated by mature stands of invasive *A. dealbata* with a percentage cover of over 75%, and these were areas not cleared by the WfW since clearing was done in sites and is still ongoing. The uninvaded areas were dominated by native grasses, forbs, and herbaceous plants with a cover of approximately 80% (based on visual cover estimates per plot that were done in this study). Uninvaded treatments were chosen based on the absence of the invasive *A. dealbata* and served as control treatment. Grasses that dominated the uninvaded sites are *Sporobolus indicus, Paspalum dilatatum, and Cynodon dactylon.* The *A. dealbata* cleared areas sites had *A. dealbata* cleared by WfW in 2016 except for one site which was cleared in 2018. The clearing involved mechanical cutting of *A. dealbata* close to the base. The felled (cut) trees were stacked on site and most of the cut stumps were harvested by the residents for fuelwood. However, large stumps of mature *A. dealbata* were left on the sites and burnt as stack piles. In this study burnt areas were avoided during soil and vegetation measurement because fires would have been an “extra” form of management and would have led to different vegetation composition to unburnt sites. The idea was to ensure that all the sites to have similar conditions so that no external factors would influence one site but not the other as that would have implications on the results.

The experimental design consisted of four sites that were approximately 5 km apart. At each site, an *A. dealbata* invaded area, a cleared area, and uninvaded area were selected (they were referred to as clearing treatments). These treatments, in each site, were approximately 100 m apart from each other. At each site, within each treatment, six 25 m^2 plots (5 m x 5 m) were laid out (marked with metal droppers) where detailed soil and vegetation surveys were done from December 2020 to February 2021. In total, 72 plots were surveyed, (four sites x three clearing treatments x six plots).

1.4. Research objectives

This study is designed to examine soil and vegetation recovery following *A. dealbata* removal in the Tsitsa catchment, Eastern Cape Province of South Africa.

Specific objectives:

- To assess changes in soil physical and chemical properties following *A. dealbata* clearing.
- To assess changes in native vegetation diversity and cover after clearing of *A. dealbata.*

1.5. Thesis outline

This thesis is composed of five chapters. Chapter 1 gives general introduction of the study and brief description of the key concepts. Objectives of the study and the study areas are stated in this chapter. Chapter 2 is a thorough literature review on the broader perspective of IAP. Chapters 3 and 4 address the two specific research objectives of this study and are both written in the format of journal papers with an abstract, introduction, methods, results, discussion, and conclusions. Chapter 5 presents the overall thesis conclusions and recommendations. All chapters end with reference list of the journal papers cited on each chapter.

Chapter 1: This chapter covers the general introduction of the thesis, motivation for the study, the objectives of the study and a brief description of the study area.

Chapter 2: This chapter gives a detailed literature review

Chapter 3: *The effects of Acacia dealbata clearing on soil physico-chemical properties in Eastern Cape, South Africa*. This chapter assessed how soil physico-chemical properties such as soil moisture content, soil penetration resistance, soil hydraulic conductivity, infiltration rate, and soil water repellency as well as soil C, N, P, pH, and exchangeable cations were affected by the clearing of *A. dealbata*. Comparisons were done between three clearing treatments, namely, cleared, invaded and uninvaded.

Chapter 4: *Vegetation recovery following Acacia dealbata clearing; implications for ecological restoration.* This chapter investigated how species diversity and vegetation cover changed after the clearing of *A. dealbata.* Comparisons is species diversity and cover were done among the three clearing treatments as mentioned in Chapter 3. List of species that were frequently occurring is provided in Appendix 1 (although some species were not fully identified).

Chapter 5: *Conclusions and implications for restoration*. This chapter gives overall summary of the thesis, reflection on the findings from each objectives/ research question. The chapter concludes by giving recommendations to ecological restoration managers and researchers.

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Chapter 2: Literature review

2.1. The scope and classification of IAP in South Africa

South African rangelands' biomes are home to hundreds of IAP that range from about 129 species in the Nama Karoo biome to 404 species in the Savanna Biome (van Wilgen and Wilson, 2018). Invasive alien plants are plants that have been accidentally or intentionally taken from their native geographic location by humans and have established as self-replicating populations in the new habitats (Richardson and Rajmanek, 2011; Ngorima, 2016). According to Blackbum et al. (2011), Australian *Acacias*, particularly *A. mearnsii*, are fully invasive, with individuals dispersing, surviving and reproducing at multiple sites across a greater or lesser spectrum of habitats and extent of occurrence. They are often called invasive due to their detrimental socio-economic and environmental impacts (Keller et al., 2011) and alien because they are not originally from the ecosystem they invade. The Southern African Plant Invader Atlas (SAPIA) lists 773 species as invasive and 71 of these are of special importance to rangelands or grasslands (Henderson and Wilson, 2017; O'Connor and van Wilgen, 2020), and over 80 of these species are Australian *Acacias* (van Wilgen and Wilson, 2018). In 2004, about 10 million ha of land in South Africa had been invaded to some extent, and problems associated with IAP have grown ever since (Richardson et al., 2020). There is however not a set exact number of how many IAP occur in South Africa, this is because of the many definitions of what IAPs are and the fact that new introductions are still being identified. However, despite not having a comprehensive inventory of IAP for South Africa, there are 327 plant taxa listed on the national legislation, the majority being invasive plants (Richardson et al., 2020). Moreover, the National Status Report on Biological Invasions lists 379 terrestrial and fresh-water plant species as invasive (Zengeya and Wilson, 2020). Given the diverse definitions of IAP, different parts of the world classify IAP differently depending on the character traits of the plant and its impact on native ecosystems. For example, IAPs in other parts of the world are classified depending on the severity of their impacts on ecosystems, invasiveness, the potential for spread, and socio-economic and environmental values (Nel et al., 2004).

Although many IAP in South Africa are already well-established and cause substantial damage, referred to as major invaders or transformers, there are still some species that are still at the early stages of invasion, with less influence but with a potential of increased range and consequences in the next decades (Nel et al., 2004; Richardson et al., 2020). Consequently, South Africa prioritizes IAP similarly to the above-mentioned classification by other parts of the world, however, South Africa adds priority to spatial characteristics and conflicts of interest (Nel et al., 2004). According to Kumschik et al. (2012) prioritization framework, IAP are prioritised based on how species are impacting on different stakeholders. The prioritization framework gathers information on the perspective of stakeholders on IAP and incorporate that with scientific information, and catagorises the impacts and weigh them based on the data from stakeholders (Kumschick et al., 2012). This framework is useful in informing the decisions that mangers should make in terms of allocating resources for the control of IAP (Kumschick et al., 2012). Furthermore, in South Africa, IAPs are listed under three categories of invasion according to the National Environmental Management Biodiversity Act (NEMBA) (Act no 10 of 2004) (NEMBA, 2014). Category 1 is divided into two, A and B, where category 1a lists the IAPs that must be combatted and eradicated, and these plants are not allowed to be planted or traded anywhere in the country because they have a high invasive potential. Category 1b lists IAPs that must be controlled, removed and destroyed. Category 2 includes species that are potentially invasive including commercially significant species such as *Pinus, Acacia* and *Eucalyptus* species. However, for category 2 species to be cultivated, a permit must be granted (NEMBA, 2014). Lastly, category 3 includes species that should only be in designated areas or provinces. These species are regulated by activity, meaning that planting, propagation, breeding, and trade of these species outside of the prescribed areas is prohibited, however no permit will be issued for the species to exist in riparian zones (see SANBI website) (NEMBA, 2014). In South Africa, *A. dealbata* which is the study species is categorized as a category 2 invader, i.e., it may be cultivated for commercial purposed with the relevant permits, however its dire impacts mean that if it grows beyond the boundaries of the permit, it must be controlled.

2.2. Invasion by Australian *Acacias* in South Africa

In South Africa, key alien plant invaders include the Australian *Acacias*, *Pinus, Eucalyptus,* and *Lantana* which have invaded different parts of the country and have been reported to transform the ecosystem (Fill et al., 2017; Ruwanza et al., 2013, van Wilgen et al., 2020). About nine *Acacia* species are listed as major invaders in South Africa and three are listed as emerging, and all these species are affecting all regions of the country except for the arid north and humid tropical eastern lowlands (Nel et al., 2004; Kull et al., 2011). Factors such as how and why the species were introduced to South Africa and how humans use the species have a crucial role in the subsequent scale of invasion (Morris et al., 2011). In South Africa, the growth and expansion of the Australian *Acacias* was accelerated by government grants and free seeds as extensive *Acacia* woodlots were planted to promote the production of wood products, such as timber production and tanbark in a tree-poor country (Witt, 2005; Kull et al., 2011). The year 1981 records the peak in the extent of *Acacia* in the country with formal *A. mearnsii* plantations covering about 146 000 ha (Kull et al., 2011). This decreased to approximately 96

000 ha in 2011 (Kull et al. 2011). Moreover, by 2011, *A. dealbata* covered approximately 500 000 ha, invading mostly higher-level grasslands and important water catchments (Ngorima, 2016).

Initially, IAP were introduced in South Africa for fuel, sand stabilization, and ornamental reasons in the Cape region of the country (Kull et al., 2011; Le Roux et al., 2011). However, some of the *Acacia* species also contributed significantly to the economy of the country (Kull et al., 2011), through timber production. South Africa is amongst the leading producers of tannin which is derived from *A. mearnsii* (Griffin et al., 2011), as such *A. mearnsii* is estimated to have spread to over 2.5 million ha by 2001 in South Africa (De Witt et al., 2001). Moreover, many of the existing *A. mearnsii* plantations provide jobs to approximately 30 000 people in the grasslands and rural areas which have high unemployment rates (van Wilgen et al., 2011). *Acacia mearnsii* produces wood chips and pulp, making it the most profitable forestry species in Southern Africa (De Wilt et al., 2001; Kull et al., 2011). These *A. mearnsii* related products and uses contributed up to R701m to the country's economy in 2009 (van Wilgen et al., 2011). On the other hand, Australian *Acacias* are used for fuelwood, charcoal, poles, and planks by poorer communities to sustain their livelihoods (Kull et al., 2011; Ngorima, 2016). For example, Ngorima (2016) reported that *A. dealbata* is utilized by the rural communities of the Eastern Cape Province of South Africa for firewood, fencing, medicine, and for livestock fodder.

Acacia dealbata is a small-medium sized tree that is native to the south-eastern region of Australia, however, it invades other parts of the world including South Africa (Lorenzo et al., 2010; Gouws and Shackleton, 2019). *Acacia dealbata* has usually invaded areas either as distinct individuals or as dense stands of forests, especially in their invaded range (Souza-Alonso et al., 2014). According to Lorenzo et al. (2010), the invasiveness of *A. dealbata* is because of propagule pressure, introduction life history, phenotypic plasticity, and predation to disturbance. Moreover, *A. dealbata* has strong adaptive capacity to foreign environments and can reproduce vegetatively and is allelopathic (Souza-Alonso et al., 2014). *Acacia dealbata* produces large number and density of seeds that persists in the soil post clearing, i.e., they remain viable in the soil even after they have been dormant for a long time (Passos et al., 2017). The species is a successful invader because of its traits as a pioneer species, its ability to mature rapidly after germination, and because it forms a symbiotic relationship with N-fixing bacteria that allows it access and fix N more effectively (Lorenzo et al., 2010). As a result, the species alters the composition of soil communities and reduces the cover and diversity of native plant species (Souza-Alonso et al., 2014). Across the globe, particularly in South Africa, *A. dealbata* has a wide range of uses such as commercial and subsistance uses (Souza-Alonso et al., 2014). *Acacia dealbata* has long had a local utilitarian and socio-economic significance in the life of South African rural communities, particularly for fuelwood and construction materials, despite its little commercial success (Poynton, 2009; Ngorima, 2016). For instance, a study by De Neergaard et al. (2005) reported that all households in the Madlanga community of Eastern Cape use a combination of *A. dealbata* and *A. mearnsii* wood for fuelwood and construction, and almost a fifth of the households earn a cash income from selling the firewood. Furthermore, the bark products of the species can be used for medicine and as natural fertiliser, and as fodder for livestock (De Neergaard et al., 2005; Ngorima and Shackleton, 2019).

2.2.1. Impacts of IAP on water resources

Despite the economic and social benefits of IAP, recent studies reveal that the ecological cost of invasion still outweighs these benefits (Richardson et al., 2020). The effects of IAP have been recorded in many parts of the world with a range of recorded impacts including declines in native species diversity, decreases in stream flows due to increased water consumption, and changes in nutrient cycling and fire regimes (Yelenik et al., 2004; Morris et al 2011; Everson et al., 2014; Le Maitre et al., 2020).

In terms of water resources, some IAP, particularly *Acacias* and *Eucalyptus* use up more water than native species (Morris et al., 2011). For example, the above-mentioned two IAP have long tap roots and more above-ground biomass which leads to deeper water extraction, and they have a greater leaf area that is advantageous for transpiration (Moris et al., 2011). A study done by Le Maitre et al. (2016) reported that IAP reduce the mean annual runoff by approximately 1.44 billion m^3/year which is equivalent to about 2.9% of the naturalized mean annual runoff. The same study further stated that almost 34% of the water reduction was due to invasions by *Acacias* such as A. *mearnsii, A. dealbata,* and *A. decurrens* (Le Maitre et al., 2016). Le Maitre et al. (2011) also reported that, the riparian stands of *A. mearnsii* used more water as compared to adjacent drylands that are invaded by the same species. Moreover, *A. mearnsii* use of water is twice as much compared to the native vegetation growing alongside it (Scott-Shaw and Everson, 2018; Le Maitre et al., 2019). Furthermore, in river catchments, IAP have been reported to reduce water yield by approximately 5% and this is subject to increase as the rate and magnitude of invasion increase (Morris et al., 2011). The Western Cape Water Supply Scheme experiences about 38 million m^3 reduction in their yield because of IAPs (Le Maitre et al., 2020). Due to their advanced transpiration, IAP consume more water causing a decrease in river flows from invaded catchment areas (Le Maitre et al., 2011). Since there are *Acacia* invasions in the grassland biome, which is an area of great water supply, it is likely that the impacts are felt more there (Le Maitre et al., 2016), although this needs to be researched. Therefore, IAP have crucial implications for water security and decrease water availability for agriculture, industry, recreation, conservation, and domestic use (Le Maitre et al., 2011). Consequently, by 2010 South Africa was spending approximately R4 billion annually on the control of IAP and about 70% of that was directed to the reduction water resources in the grassland and fynbos biome, and this amount was estimated to increase as the extent of IAP increased (van Wilgen et al., 2011; Le Maitre et al., 2020).

Most of the water reductions by IAP, specifically Australian *Acacias*, can be attributed to their traits; rapid growth rates and ability to out-compete native plants (Moris et al., 2011). *Acacia* species' roots develop and grow longer and deeper into the ground than native species giving them access to deep soil nutrients and to deep underground water (Morris et al., 2011). For example, Everson et al. (2014) reported that between 2006 and 2013 there was an average loss of 395 mm of groundwater due to the deep-rooted IAP that source water from deep soil profiles. As a result, there was an observed drying of the soil profile in those above-mentioned years (Everson et al., 2011). The vigorous vegetative growth of IAP also supports the production of nutrient-rich seeds which contribute to their successful invasion and persistence (Morris et al., 2011; Gibson et al., 2011). During periods of water scarcity, these deep roots and fast-growing seedlings give *Acacia* an advantage to grow more than the native species (Moris et al., 2011).

2.2.2. Changes in fire regimes due to IAP

The impact of fires on *A. dealbata* in the grassland ecosystems has not been widely studied, thus remains unclear and contradictory. In many parts of South Africa, fires are perceived as important agents for the maintenance of open grasslands, and for the control of many *Acacia* species that invade the grassland ecosystems (Radford et al., 2001). Contrary, fire play a crucial role in the establishment and distribution of IAP, for example, the invasive *Melalueca quinquenervia* in Florida invades fire-cleared areas (Stocker and Hupp, 2008). In South Africa, after several fires, the invasive *Banksia ericifolia* multiplied from 100 individual species to approximately 10 000 individuals covering over 217 ha of land (Geerts et al., 2013). Furthermore, heat scarification from fires stimulates the germination and spread of the IAP, especially in the fynbos ecosystem (Dibbles et al., 2008; Le Maitre et al., 2011). A study done at the Saratonga National Historical Park in New York showed that fires increased the stem density of IAP for three years after spring burning in old fields (Dibbles et al., 2008). On the other hand, IAP plants change fire fuel properties, intensity, severity, and spatial complexity and fire regimes, subsequently creating conditions that favor IAP but are detrimental to the persistence of native species (Brooks, 2004; Setterfield et al., 2013). For example, invasive grasses increase horizontal fuel continuity, creating a fuel bed bulk density that is more likely to ignite and spread fires, thus increasing the frequency, magnitude, and spatial homogeneity (Brooks, 2004).

Woody IAP that invades grasslands change fire regimes from frequent, low severity to mixed-severity with longer fire return intervals and induce woody fuel load with high amounts of dead tissue and litter (Brooks, 2004). For example, the Tallow trees (*Triadica sebifera*) over-top surface vegetation which reduces fine grass fuels and increases coarse woody fuel that makes it harder to ignite fires (Brooks, 2004). The reduced frequency facilitates early invasion and changes of topsoil and nutrient distributions to suit the characteristics of the invading species (Brooks, 2004). Invasive alien plants take advantage of the increased nutrients, light, and reduced competition after the fires (van Wilgen, 2009; Aran et al., 2013). A study by Aran et al. (2013) showed that the seeds of *A. melanoxylon* are hard and persist in the soil and re-sprout vigorously after fires. Heat and ash from the fires also stimulate the growth of invasive *Conyza canadensis* (Aran et al., 2013). Invasive alien plants, *Acacia* in particular, have a potential for fast biomass accumulation; massive, permanent seed banks; and can fix nitrogen, thus making them flourish compared to native species (Le Miatre et al., 2011). The enhanced IAP biomass results in increased fuel loads, consequently leading to increases in the severity and intensity of fires (Le Maitre et al., 2011). These fires then kill re-sprouting native plants and change the soil structure due to burnt organic matter that holds soil particles, inducing water repellency (Holmes 2001). This leads to negative impacts on soil stability and sedimentation regulation services, ultimately increasing the chances of river and dam sedimentation rates (Le Maitre et al., 2011).

2.2.3. Impacts of IAP on grazing lands

Some IAP have invaded and are spreading to grazing lands, therefore decreasing their value to stock farmers because livestock farming on natural (uninvaded) pastures is one the most practiced land uses in grassland ecosystems (Mac Donald, 2004). Studies conducted in grasslands have reported that invasion by *Acacia* species leads to a decrease in native species diversity, herbaceous cover, and changes in soil conditions (Ndou and Ruwanza, 2016; Yapi et al., 2018). Therefore, this leads to declines in grazing capacity due to the displacement of total basal cover (Yapi et al., 2018). Both *A. mearnsii* and *A. dealbata* are amongst the IAP that invades these rangelands leading to loss of grazing area (Yapi et al., 2018; O'Connor and van Wilgen, 2020). Yapi et al. (2018) further argues that in invaded grazing lands, soil nutrients are altered to suit the properties of invading species, outcompeting herbaceous species and grass cover. Yapi et al. (2018) assessed the impacts of *Acacia* species invasion on rangeland condition and livestock carrying capacity in a South African montane grassland ecosystem, and reported negative impacts of invasion on rangelands, thereby, affecting livestock production.

2.2.4. Impacts of IAP on human livelihoods

In terms of human livelihoods and well-being of rural communities, IAP pose a threat in many ways. Firstly, woody IAP such as *A. dealbata* have been shown to invade large portions of agricultural land and transform grasslands into IAP forests or thickets (Ngorima, 2016). As a result, the carrying capacity of livestock is reduced and IAP inhibit other agricultural activities such as crop farming and causes declines in the availability of non-timber forest products (NTFP) (Ngorima, 2016). Secondly, dense forests of IAP pose security threats to local communities and can, to some extent, perpetuate criminal activity (Kanyari et al., 2021). For example, De Neergaard et al. (2005) revealed that thickets of Australian *Acacia* act as shields for thieves stealing livestock and they endanger local people's lives, especially women that collect firewood in the forests. Some IAP such as *Mikania micrantha* and *L. camara* are thorny and form thickets, thereby, making it impossible for rural communities to access local forests and this reduces the accessibility of firewood and fodder (Rai and Scarborough, 2015). Thirdly, with increasing invasion extent, the likelihood of disease outbreaks in some parts of the world as has been reported. For example, *L. camara* invasion has been associated with the occurrence of the tsetse fly that is a vector of sleeping sickness to people (Day et al., 2003; Ngorima, 2016). Finally, invasion by IAP causes cultural degradation when they invade communal commonages and rivers that are of great cultural significance. As a result, people do not want IAP to invade certain landscapes such as grazing lands, homesteads, grave sites, riverbanks, and sacred pools (Shackleton et al., 2007; Ngorima, 2016). When IAP spread into sacred cultural sites, they replace indigenous medicinal plants and other culturally significant species that are used for sacred cultural healing purposes (Ngorima, 2016). When *A. dealbata* invades these culturally and spiritually significant rivers and pools, they can reduce volume of water and stream flow, resulting in activities that require moving water such as baptism and cleansing of *abakhwetha* (*abakhwetha* are young males that are at a traditional phase of transitioning from being young boys to manhood. The traditional ceremony is also referred to the "coming of age") become negatively affected (Ngorima, 2016). This severely scars the community's attachments to these areas and results in cultural erosion due to detachment from ancestral customs (Ngorima, 2016).

Other impacts that have not been quantified at a landscape scale yet include aesthetics and recreational aspects (van Wilgen et al., 2011). These have a significant impact on many sectors of society, especially the poorer communities (Kull et al., 2011). If more time passes with these impacts not being addressed or studied, ecosystem services' production and supply, and diversity of available resources will be compromised, thus, negatively affecting human well-being (Le Maitre et al., 2011). Ngorima (2016) further argue that although IAP may have undesired ecological and socio-economic impacts, rural livelihoods rely heavily on them, and complete removal may impoverish rural communities, especially where there are limited or no alternatives. Moreover, IAP may also be habitats to important animal species, for instance, the African fish eagles, which are of great cultural significance make habitats from the invasive *Eucalyptus* species (Ngorima, 2016).

There has been growing debates and conflicts pertaining the comparisons of the benefits of goods and services derived from IAP by different stakeholders as they may be positively impacting one group while negatively affecting the other (Le Maitre et al., 2011). Zengeya et al. (2017) termed these conflict-generating species and they comprised about 6% of the species that were sampled in his study, and these include *A. cyclops, A. dealbata, A. mearnsii* and *A. melanoxylon, Pinus* species, *Prosopis* species, and *Cacti* species. These conflicts arise when the species has both economic and intrinsic values and when there are uncertainties about the adverse impacts the species may have. (Zengeya et al., 2017). For instance, *A. mearnsii* injects significant economic benefits from producing wood chips and tanbark (Griffin et al., 2011; van Wilgen et al., 2020), however, those benefits may only be enjoyed by a few people. But the cost (negative impacts) of the invasion by *A. mearnsii* is borne by the larger communities (Le Maitre et al., 2011). Similarly, the *Prosopis* species is a valuable source of fodder, firewood, and shade but they have adverse social and ecological impacts ((Shackleton et al., 2014; Zengeya et al., 2017). Consequently, different organizations, in different parts of the world promote it while other countries like South Africa are spending millions of dollars to control the species (Shackleton et al., 2014; Zengeya et al., 2017). Moreover, more conflicts arise when choosing the right methods for the control of IAP (Zengeya et al., 2017). For example, biological control of Australian *Acacias* is widely accepted, however lethal biological control is still prohibited (Zengeya et al., 2017). Furthermore, some biological control agents do more harm than good. For instance, the use of cone-feeding weevil to control *P. pinaster* led to the growth of a fungus that is detrimental to commercially significant *Pinus* species (Zengeya et al., 2017).

2.2.5. Impacts of IAP on soil properties

Australian *Acacias* transform both the above and below-ground microclimates, soil communities, moisture, and nutrients (Le Maitre et al., 2011). From the early 21st century, several scholars have studied the effects of IAP on soil nutrient characteristics and soil microbial communities (Corbin and Antonio, 2004; Yelenik et al., 2004; Le Maitre et al., 2011; Ndou and Ruwanza, 2016; Ruwanza and Tshililo, 2019). Sigwela et al. (2003) stated that the most important properties to assess when studying soil are available Nitrogen (N), organic carbon (C), phosphorus (P), effective infiltration, and soils pH, and in the above-mentioned studies at least one or more of these soil properties were reported. Studying the recovery of soil properties after a disturbance is essential because soils provide a matrix where plant roots provide support and forage for nutrients and water, this means that soils are essential for biodiversity recovery (Lileskov et al., 2009). Yelenik et al. (2004) studied the effect of clear-cutting stands on soil N cycling and soil microclimates and reported that *Acacia* generated large amounts of litter, thus injecting more N to the soil. Ndou and Ruwanza (2016) reported that soil nutrients were significantly lower in the cleared sites than in the invaded sites. Ruwanza and Tshililo (2019) reported significant changes in soil infiltration rate and hydraulic conductivity following the clearing of *Eucalyptus* in Limpopo Province of South Africa. Some studies have reported that *A. mearnsii*. *A. saligna,* and *A. longifolia* increase soil organic matter and litter layer in lowland fynbos ecosystems (Le Maitre et al., 2011). Invasion by N-fixing *Acacia* promotes the domination of herbaceous species after there has been clearing of the *Acacia* (Le Maitre et al., 2011). This high concentration of N facilitates the growth of nitrophilous species and *Acacias*, and dense forests of these species hinder native species restoration (Le Maitre et al., 2011).

Invasive alien plants have a significant effect on the delivery of soil related ecosystem services as they are sensitive and largely affected by changes in the biotic communities (Lilleskov et al., 2009). They do this by changing the nutrient and water cycle, decomposition rates and soil carbon storage, and soil fertility amongst other important soil properties (Lilleskov et al., 2009). Broadly explained, the introduction of invasive ecosystem engineers such as earthworms, ants, and termites completely change structural and chemical characteristics of the soil and in turn other species dependent on the soil (Lilleskov et al., 2009). For example, invasion by earthworms (which are ecosystem engineers) mixes soils' organic horizons and cause problems such as siltation of water sources and threatens herbaceous plants (Lilleskov et al., 2009). Termites change soil properties through bioturbation rates of soils and carbon cycle (Lilleskov et al., 2009).

2.3. Alien plant control in South Africa: Working for Water programme

Numerous attempts have been implemented in South Africa to control the growth and expansion of the IAP since the 19th century (van Wilgen et al., 2020). Of interest then, was the control of certain species such as *Xanthium spinosum* (Bur weed) (van Wilgen et al., 2020). More experiments on the control of IAP gained momentum in 1913 and the success of these experiments led to this method being implemented on a wider range of IAP (Zimmermann et al., 2004; van Wilgen et al., 2020). At the time, only the invasive cactus species *Opuntia monacantha* (drooping prickly pear) was targeted for control, however, biological control has moved to large trees and water weeds (Zimmermann et al., 2004). South Africa adopted its first use of biological control after IAP were confirmed as a threat to the conservation of indigenous vegetation in the fynbos biome in the Western Cape Province of South Africa (Zimeermann, et al., 2004; van Wilgen et al., 2020). Although this was first implemented in designated areas, as the extent of invasion and the magnitude of the impacts of IAP increased, the control was then extended to other areas beyond proclaimed areas (Le Maitre et al., 2011; van Wilgen et al., 2020). This method was also implemented for the control of IAP in Kruger National Park, South Africa (Foxcroft and Freitag-Ronaldson, 2007). The late 19th century marks the growth in the momentum of studies on invasion by IAP in South Africa with participation in programs such as the Scientific Committee on Problems of the Environment (SCOPE) projects (van Wilgen et al., 2020) and the establishment of the Forestry and Agricultural Biotechnology Institute (FABI) (van Wilgen et al., 2020).

In 1995, the Working for Water (WfW) programme was launched and administered by the Department of Water Affairs and Forestry before being transferred to the Department of Forestry, Fisheries, and the Environment (formerly known as the Department of Environmental Affairs). The programme's primary goal is increasing water supply by controlling IAP invasion in water catchment areas as well as to create employment though the Public Works Programme (MacDonald, 2004). The WfW programme adopted a comprehensive approach of both chemical and mechanical clearing of the IAP in South Africa (van Wilgen, 2012). The post-apartheid South African government established and funded these labor-intensive attempts to both clear IAP and to create jobs to curb unemployment and poverty (MacDonald, 2004; Fill et al., 2017). Developments in the WfW project included an interdisciplinary approach that addresses the ecology, economics, management, and social impacts of biological invasions (Macdonald, 2004). The WfW programme also aims to improve the integrity of natural resources through stopping invasion by new species, detecting and effectively responding to emerging species, clearing already established IAP, and managing the impacts of the established IAP (Fabricus et al., 2016). The desired state envisioned by the WfW is ideally one where it would be possible to maintain the cleared sites in an uninvaded state at low cost in perpetuity (Fill et al., 2017).

2.4. Why do IAP persist after clearing?

van Wilgen (2012) argues that invasion by IAP continues to be one of the pressing issues threatening many biomes in South Africa although efforts such as clearing have been done. Regardless of the government efforts to clear IAP, several invasion studies show mixed results with regards to the recovery of soil and vegetation following the clearing of invasive plants (Ndou and Ruwanza, 2016; Yapi et al., 2018). As a result, there is still uncertainty on whether the areas targeted by WfW for removal of IAP are of top priority and whether the clearing methods used were the most effective because there has been little success in efforts made to decrease the rate of invasion, and in areas that were cleared, the rate of re-invasion is high (van Wilgen, 2012). It is clear that some areas still need post-clearance restoration actions to facilitate native vegetation recovery (Pretorius et al., 2008).

There is an assumption that after clearing of the IAP, there will be a spontaneous natural recovery of vegetation, i.e., passive restoration, (Ndou and Ruwanza, 2016). This assumption stems from the agricultural perspective that the desired vegetation grows after the pest is removed (Nsikani et al., 2019). However, Wittenberg and Cock (2005) and Ruwanza et al. (2013) argue that the mere clearance of the invasive species after it has had its impacts on an ecosystem does not guarantee that the ecosystem will recover to the original state in just a few years. In fact, a study by Ruwanza et al. (2013) showed no recruitment of native vegetation after passive restoration but rather more cover by invasive herbs and graminoids were reported along the Berg River where *Eucalyptus* was removed. This is because of factors such as secondary invasion, resource alteration, the legacy effect, and in some cases where biotic and abiotic thresholds have been crossed due to the dense stands of IAP being present for many years (Ruwanza et al., 2013).

Firstly, most invasive Australian *Acacias* accumulate persistent soil seed banks in many ecosystems (Le Maitre et al., 2011). *Acacia dealbata, A. saligna* and *A. longifolia,* amongst other IAP, re-sprout after being cleared (Le Miatre et al., 2011). Also, clearing of *Acacia* followed by burning in the fynbos biome leads to dense recruitment of the species, especially because fires stimulate *Acacia* seedbank from the soil (Le Maitre et al., 2011). Furthermore, to reduce the density of IAP, the "Fell and Remove" clearing treatment is suggested by (Blanchard and Holmes, 2008) since it allows native vegetation to colonize from the soil seed bank, thus if the soil seed bank is present (Le Maitre et al., 2011). However, Ruwanza et al. (2013) showed that alien herbs such as *Solanum nigrum, Rumex crispus* and *Lactuca serriola* dominated after the felling and removal of *E. camaldulensis*.

Secondly, after clearing there are areas that are further degraded through reinvasion by the same species or a different invasive species (secondary invaders) (Ndou and Ruwanza, 2016; Nsikani et al., 2019; Geert et al., 2022). Secondary invasion, in this context, can be described as the process when species that thrive in disturbed environments grow where IAP have been cleared (LeMaitre et al., 2011). Nsikani et al. (2019) describe this as the increase in the abundance of the non-targeted IAP. A study by Fill et al. (2017) showed that there was a rapid secondary invasion by IAP and weedy grasses in an *Acacia* cleared site. Secondary invaders flourish because the clearing of the primary invader allows them to recruit from soil seed banks accumulated before the invasion (Nsikani et al., 2019). Clearing of N-fixing *Acacias* leaves excess resources in the soil, such as N and sunlight, which secondary invaders consume to facilitate their growth (Nsikani et al., 2019). Methods used to clear IAP may also facilitate secondary invasions. For example, using broadleaf herbicides favors the proliferation of secondary invader grasses (Nsikani et al., 2019). Following the clearing of *A. saligna* in the fynbos biome, 32 secondary invaders inhibited the area (Nsikani et al., 2019). This was due to several factors including fires; as low-intensity fires improve the germination rate of the invading species' seeds (Nsikani et al., 2019).

Lastly, the legacy effect is also prominent in areas that have been invaded for a long period and it hampers recovery of native vegetation in an area (Le Maitre et al., 2011). The legacy effect is the "long-lasting changes in ecosystem structure such as increased soil nutrient levels that persist following the removal of the invasive species" (Le Maitre et al., 2011). Even after clearance, soil processes and properties such as soil organic matter and nutrient stocks, nutrient cycling rates, soil moisture content, pH, and cation distribution are altered (Corbin and D'Antonio, 2004). For example, the invasion of woody N-fixing species into habitats that previously lacked woody N fixers, such as *Acacia*, leads to substantial increases in soil N pools (Corbin and D'Antonio, 2004) and it may take many years for soil nutrients and processes to bounce back to their original state (Fill et al., 2017). If this legacy effect prevails in the soil, it hinders native vegetation restoration efforts by facilitating re-invasion by the same or other IAP and by preventing recovery of native vegetation (Marchante et al., 2009). Active management strategies would then be needed as the secondary dominance by invasive grasses have significant impacts on ecosystem dynamics (Fill et al., 2017). Management efforts would also then be cognizant of the factors that drive the establishment of secondary invaders and should explicitly address the fundamental drivers and their effects (Le Maitre et al., 2011).

2.5. Ecological restoration

2.5.1. Soil and vegetation recovery

Studies have shown that there are complex interactions between vegetation composition, soil microbes, and nutrient cycling (Corbin and D'Antonio, 2004). In the context of ecological restoration, considering these interactions is essential because the effects of IAP invasion and clearing may be very challenging and may affect the rate and path of the succession of vegetation after clearing (Knops et al., 2002; Corbin and D'Antonio, 2004). Therefore, when studying IAP and recovery of areas cleared of IAP, it is important to take into consideration the duration of invasion so as to have a clear understanding of the impacts and extent of invasion over time (Marchente et al., 2009). Soil and vegetation recovery after clearing of the IAP depends on soil types, traits of the IAP that invades, infestation age and density, and treatments used for clearing the invasive plant (Pretorius et al., 2008).

Some invasive alien plants store up seed banks that persist after their clearing, thus complicating the long-term restoration of the cleared areas because these seed banks become sources of re-invasion (Pretorius et al., 2008). For example, when IAP such as the Kikuyu (*Pennisetum clandestinum*) are cleared and burnt, the regenetation of native species gets encroached by *P. clandestinum*, leading to delayed native vegetation recovery (Reinecke et al., 2008). In another study, Blachard and Holmes (2008) reported that eight years after *A. mearnsii* and *E. cladocalyx* clearing and follow-up treatment, few native species that were sown recruited because seedbanks of the above-mentioned invasive species persisted since they are long-lived and were stimulated by fires. Even after fires (eight years later) *A. mearnsii* dominated and individuals continued to recruit from soil seed banks (Blanchard and Holmes, 2008; Pretorius et al., 2008). Therefore, to account for re-invasions, there must be a focus on investigating the impacts of re-invasion and secondary invasion on soil and native vegetation recovery, and how both re-invasion and secondary invaders can be suppressed (Blanchard and Holmes, 2008).

2.5.2. Successful recovery of soil properties and native vegetation

Passive restoration is reported to be insufficient in terms of soil and vegetation recovery, however, it has a potential to work in the short-term, e.g., ten years after *Acacia* clearing using the 'Fell and remove' method Blanchard and Holmes (2008) reported some native vegetation recovery. Other methods to investigate ecological restoration efficacy include recovery after clear felling and herbicide application on stumps, thinning the slash to allow native vegetation to grow between stacks, and, in conducive ecosystems, burn the stacks or leave them to rot (Blanchard et al. 2008). Sowing a mixture of native seeds and alien grasses suppresses the growth of *A. mearnsii, E. cladocalyx* and *Pteridium aquilinum* recruits (Pretorius et al., 2008). Fill et al. (2017) conducted a study on the recovery of the Western Cape endemic vegetation following *Acacia* removal and reported that the clearing of IAP significantly reduced alien plant cover, allowing remnant native species to recover. Planting a mixture of indigenous seeds after clearing improves biodiversity and abundance of the indigenous vegetation although not fully (Pretorius et al., 2008). Pretorius et al. (2008) showed that sowing treatments suppress the recruitment of *A. mearnsii* seeds over time. This positive response of the indigenous species gives hope that given enough time and necessary intervention methods (active restoration), areas cleared of IAP will move towards the desired states like undisturbed or uninvaded communities (Fill et al., 2017).

Restoration ecology provides an intimate understanding of ecosystems, succession, and the different methods that can be implemented to enhance soil and native vegetation recovery after alien plant clearing, and ultimately the recovery of ecosystem functions, processes, and ultimately biodiversity restoration (Dobson et al., 1997). However, to successfully restore an area, there should be a thorough understanding of landscape ecology with achievable restoration goals. Cairns (1989) provided a model of potential management pathways for the restoration of disturbed areas. The model describes that from a current state of disturbance, an ecosystem may stay stable or further degradation may occur. To prevent the latter, active recovery efforts must be implemented to facilitate soil recovery and native species revegetation (Cairns, 1989). In this context, restoration is defined as the "re-establishment of the structure, functions, and natural diversity of an area that has been altered from its natural state" (Pess et al., 2003). Ecological restoration creates natural ecosystems that are functional and creates interactions among ecosystems (Pess et al., 2003; Rosenfield and Muller, 2019). Furthermore, vegetation restoration increases carbon fixation, biodiversity, and rural development (Pausas et al., 2014). An effective restoration technique is one that reduces the amount of soil loss from ecosystems after IAP clearing, thus increasing the soil volume and microsites that promote the establishment and growth of native plants in the cleared areas (Kimiti et al. 2011).

Ecological infrastructure is an important component to take into consideration when attempting to restore land that has undergone various forms of disturbances such as clearing of IAP, especially in a socio-ecological system (Fabricus et al., 2016). There is a range of benefits ecosystems get from improved ecological infrastructure, from enhancing soil fertility for agricultural activities and protecting people and the environment from natural disasters such as floods, fires, wind, and drought (Fabricus et al., 2016). Furthermore, ecological infrastructure is fundamental to humanity through the delivery of provisioning, regulating and cultural ecosystem services (Fabricus et al., 2016). In the case of the Tsitsa catchment, robust and fertile soil with the ability to resist erosion and provide a basis for agricultural production is amongst the most valuable and crucial ecological infrastructure. Accelerated degradation of the ecological infrastructure, due to invasion by *A. dealbata* in the Tsitsa catchment poses a huge threat to the adaptive capacity of people and ecosystems (Fabricus et al., 2016). To avoid and mitigate such threats, there needs to be an investment in projects that aim at increasing the natural vegetation cover, enhance soil organic contents, and protect riverbanks and riparian zone and in some areas restore wetlands (Fabricus et al., 2016).

Therefore, active soil and native vegetation recovery measures are a necessity and should be implemented after the clearing of IAP including removal of secondary invaders and planting native grasses that would assist in recovering soil properties (Fill et al., 2017). This means that, for successful recovery of soil properties, there should be a clear understanding of how vital soil properties such as primary nutrients (N, C, P, and exchangeable cations) and soil physical properties (soil pH, infiltration rate, penetration resistance, hydraulic conductivity, moisture content) are changed by invasion. To attain this information, comparison of invaded, cleared, and uninvaded sites is advised. Thorough information on how the soil properties are impacted will inform which active restoration approach should be implemented to suppress invasion and facilitate native vegetation recovery. Restoring ecological infrastructure will recover human-induced disturbances to biodiversity and the dynamics of the ecosystem. This entails reconstructing the cleared areas to a desirable state through re-establishing species assemblages, structure, and functions of the ecosystem (Richardson et al., 2007). This would of course need critical assessment of which invasive species to prioritize in terms of controlling and which areas should be of top priority to work on (van Wilgen, 2012). There should also be clear and feasible goals set and progress should be monitored within a framework of adaptive management (van Wilgen, 2012). It is also crucial to assess how contextual factors such as land ownership patterns and land practices have played a role in the area and restoration efforts that have been done there, meaning long-term monitoring is fundamental (Fabricus et al., 2016). Absolute ecological restoration of an area to its original state can be challenging to achieve at a landscape scale due to costs and land-use conflicts and at catchment scale due to large alterations on hydrological and fluvial processes (Richardson et al., 2007). Therefore, an informed understanding of the temporal and spatial dynamics of the catchment landscape is crucial (Richardson et al., 2007).

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Chapter 3: The effects of *Acacia dealbata* clearing on soil physico-chemical properties in the Eastern Cape Province, South Africa.

Abstract

Invasion by alien plant species in South Africa continues to compromise the stability of ecosystems by causing declines in biodiversity, altering soil nutrients and processes, and subsequently transforming ecosystem functionality. The control of invasive alien plants has been widely implemented across the globe to minimize their negative impacts; however, the impacts can persist long after the invasive plant has been removed. The impacts of *Acacia. dealbata* clearing on soil properties remains understudied despite the significant role soils play in ecological restoration. This comparative study determined the impacts of *A. dealbata* clearing on soil physico-chemical properties in the Eastern Cape Province, South Africa. Soils were collected from three different clearing treatments, namely, cleared, invaded, and uninvaded, on six 5 x 5 m plots over three summer months. The plots were replicated four times for each clearing treatment, for a total of 72 sampling plots. Soils were assessed for soil pH, resistivity, P, C, N, and exchangeable cations as well as soil moisture content, penetration resistance, infiltration rate, hydraulic conductivity, and water repellency. Clearing of *A. dealbata* did not have any significant effect on soil nutrients, however there were variation in soil pH, resistance, and sodium (Na). Soil pH was significantly higher in the uninvaded treatments than the cleared and invaded treatments. Soil moisture content was significantly higher in the cleared treatments than the adjacent invaded and uninvaded treatments in December than in January and February. Soil penetration resistance and infiltration rates were significantly higher in December, whereas infiltration rates were significantly higher in December in the cleared treatments. No significant differences were recorded for soil hydraulic conductivity. *Acacia dealbata* clearing triggers varied soil responses with some properties showing decreases, while shows increases. The increase in soil moisture content and infiltration rates, as well as Na are an indication that clearing *A. dealbata* has the potential to shift soil properties towards a recovery trajectory. However, for clearing to be successful clearing methods such as fell and remove should be practised and be followed up by active restoration approaches such as active seeding.

Key words: Invasive alien plants, post-clearing, soil nutrients, soil recovery, legacy effect.

3.1. Introduction

Invasive alien plants (IAP), particularly *Acacias,* have been shown to transform both the above and below-ground microclimates, soil communities, moisture and nutrients (Le Maitre et al., 2011; Ndou and Ruwanza, 2016). Invasive alien plants are non-native plants species that severely compromise the integrity and functioning of the ecosystems they invade, and they are viewed as the second greatest drivers of global change (Bonanno, 2016; Ruwanza and Shackleton, 2016). In South Africa, key alien plant invaders include the Australian *Acacias*, *Pinus, Eucalyptus,* and *Lantana* which have invaded different parts of the country (Ruwanza et al., 2013; Fill et al., 2017; van Wilgen et al., 2020). In the past two decades, several studies have shown the negative effects of IAP on soil nutrient characteristics and soil microbial communities (Corbin and Antonio, 2004; Ruwanza et al., 2013, Ndou and Ruwanza, 2016; Ruwanza and Tshililo, 2019). For example, Ndou and Ruwanza (2016) reported that the *Acacia* invasion and its subsequent clearing results in varied changes in soil properties. Although IAP have undesired impacts, some communities are using them to their advantage, e.g., for fuel wood and building material (Ngorima, 2016), thus making them conflicting species (Zengeya et al., 2017). However ecological impacts seem to outweigh these benefits; therefore, the species are still set to be controlled in many parts of the world, including South Africa where a national IAP clearing programme has been implemented (Kull et al., 2011; Gwate et al., 2021). Nevertheless, clearing alone has been proved to be insufficient to induce soil recovery because most IAP have been shown to cause long-lasting soil legacy effect (van Wilgen, 2012; Nsikani et al., 2018).

There has not been an adequate description of the ability of autogenic recovery of soil properties after the clearing of IAP in South Africa (Ndou and Ruwanza, 2016; Nsikani et al., 2018). Successful recovery of ecosystems is largely influenced by the capacity of both biotic and abiotic factors to enhance recovery, and as far as ecological restoration is concerned, preinvaded areas have a potential to self-regenerate or can recover when aided through biotic and abiotic interventions (Le Maitre et al., 2011; Holmes et al., 2020; Gwate et al., 2021). Studies have concluded that ecological restoration is elusive, however, there is an appreciation of potential soil properties that may be used as a prognosis of ecosystem recovery after clearing IAP (Costantini et al., 2016; Gwate et al., 2021). For example, previous studies have shown that the recovery of soil fertility (particularly soil N, C and P) has the potential to kick-start vegetation recovery (Ndou and Ruwanza, 2016; Nsikani et al., 2018). On the other hand, other studies have suggested that for soil recovery to take place after IAP clearing, soil legacy effect linked to N_2 fixing plants will have to be managed first through active restoration (Ndou and

Ruwanza, 2016). Therefore, an intimate understanding of soil physico-chemical properties to inform soil recovery interventions after IAP removal is needed. Soils are very essential for life and biodiversity because they provide a matrix where plant roots provide support and forage for nutrients and water (Lilleskov et al., 2009). However, the threat from invasion by IAP and the changes they inflict compromise the ability of the soils to provide these supporting benefits.

Australian *Acacias* are famous for their N-fixing abilities but are also known for their transforming abilities as they can alter ecosystems by using and adding significant quantities of resources, causing erosion, stimulating or suppressing fires, and accumulating large amounts of litter (Richardson and van Wilgen, 2004; Vardein et al., 2012; Ruwanza and Shackleton, 2016). For example, Le Maître et al. (2011) reported that *A. saligna* and *A. longifolia* increase soil organic matter and litter layer in Portuguese coastal systems and lowland fynbos biome in South Africa. This ultimately results in increased soil moisture after the above-mentioned *Acacias* invaded the areas (Le Maitre et al., 2011). As N-fixing species, *Acacias* influence the inputs of soil C, N, and microbial processes, resulting in altered ecosystems (Le Maitre et al., 2011; Ruwanza and Shackleton, 2016). They have been shown to cause shifts from low to high N-cycling regimes in the fynbos biome of South Africa (Yelenik et al., 2004; Le Maitre et al., 2011). Invasion by N-fixing *Acacia* promotes the domination of herbaceous species after the clearing of the *Acacia* (Le Maitre et al., 2011). This high concentration of N facilitates the growth of nitrophilous species and recruiting *Acacias*, and dense stands or patches of these species hinder restoration (Le Maitre et al., 2011).

Acacia dealbata is a successful invader and ecosystem transformer in many parts of the world including Europe, Africa, and America and has been shown to strongly alter soil and plant communities (Lazzaro et al., 2014). The plant was introduced in South Africa between 1880 and 1890 for different reasons but mostly for economic benefit, such as forestry, and ornamental purposes, but the species now invades several parts of the country (Hirsch et al., 2017). Due to its versatile and adaptive traits, *A. dealbata* can significantly change soil properties, almost permanently, hence it is called a transformer species (Richardson et al., 2000; Lazzaro et al., 2014). The spread of *A. dealbata* poses threats to the conservation of areas that are essential hotspots of biodiversity and endemism (Mittermier et al., 2004; Lazzaro et al., 2014).

Invasion by IAP degrades the natural environment, tempers ecosystem processes and stability and subsequently compromises the ability of ecosystems to produce and deliver ecosystem services (Gaertner et al., 2009; Kerr and Ruwanza, 2016). The impacts IAP exert stretch beyond the physical environment; they also directly and indirectly lead to significant dents in economic productivity and have adverse impacts on human livelihoods (Richardson and van Wilgen, 2004; Kerr and Ruwanza, 2016; Ruwanza and Shackleton 2016; Ngorima, 2016). Moreover, IAP are of great concern in the grassland biome because they consume water resources more than native species (Gwate et al., 2021). For example, Le Maitre et al. (2016) reported that mean annual runoff decreased by approximately 1.44 billion m^3/year which is equivalent to 2.9% of the naturalized mean annual runoff. Moreover, reductions of water by wattles such as *A. mearnsii, A. dealbata* and *A. decurrens* amount to about 34% of the water reductions in South African catchments (Le Maitre et al., 2016). Most of these *Acacia* invasions are found in the grassland biome hence these impacts are felt more there (Le Maitre et al., 2016). Between 2006 and 2013, an average of 395 mm of groundwater was lost to the deep-rooted IAP that source water from deep soil profiles leading to the drying of the soil profile and subsequent changes to soil properties (Everson et al., 2011).

South Africa has not been immune to the invasion by *Acacias*, in fact, it is declared a global hotspot for wattle introductions and tree invasion (Magona et al., 2018). Australian *Acacias* marked their peak in invading South African ecosystems about three decades ago and to date, about nine *Acacia* species are listed as major invaders in South Africa, affecting all regions except for the arid north and humid tropical eastern lowlands (Nel et al., 2004; Kull et al., 2011). In the fynbos biome, a low-nutrient ecosystem, *Acacia* invasion altered the natural N-cycling process from a slow to a faster process (Yelenik et al., 2004). Recent studies have shown that the N-fixing legacy effect linked to *Acacia* invasion in the fynbos biome persist for several years after clearing (Yelenik et al., 2004; Nsikani et al., 2018). Gouws and Shackleton (2019) reported that *A. dealbata* displaces native species through its ability to change soil properties. Musil and Midgley (1990) reported that the impact of *Acacia* infestation on soil properties is greater that the impacts caused by fires on ecosystems. *Acacia* infestation resulted in some soil chemical properties being twice as much compared to natural sites (Musil and Midgley (1990). Van Der Waal (2009) reported an increase in soil chemical properties such as N, P, C, Zn, Mn, and Ca as well as in soil water repellency 11 years after clearing of *Acacia*.

Thus, there has been a growing vigilance by ecologists on the various ways in which invasion by IAP impact soil properties in different ecosystems across different regions of South Africa (Yelenik et al., 2004; Kerr and Ruwanza, 2016; Yapi et al., 2018; Ruwanza and Tshililo, 2019; Gwate et al., 2021). These studies reveal a range of significant community level impacts in many parts of the country that are consistent with findings from other parts of the world

(Richardson et al., 2000). Invasive alien plants are known to amplify nutrient mineralization and shifts nutrient regimes due to their faster nutrient uptake abilities (Ehrenfeld, 2003; CastroDiez et al., 2012; Kerr and Ruwanza, 2016). They also modify soil microclimates and microbial communities, shift species composition, and alter the quality and quantity of litter (Yelenik et al., 2004; Castro-Diez et al., 2012; Ruwanza and Shackleton, 2016). Changes associated with litter fall tend to exert more intense changes to soil properties, as such litter decomposition is regarded as a crucial link between below-ground and above-ground processes (Ruwanza and Shackleton, 2016). These changes in soil properties subsequently lead to changes in plant structure and composition, usually promoting mono stands of the invading species and reducing plant richness and diversity (Ruwanza and Shackleton, 2016).

Numerous attempts have been implemented in South Africa to control the growth and expansion of the IAP since the 19th century (van Wilgen et al., 2020). In 1995, the Working for Water (WfW) programme was launched with a primary goal of increasing water supply by controlling IAP invasion in catchments (MacDonald, 2004). The programme operated under the assumption of passive restoration, which assumed that after clearing, soil properties and native vegetation would recover naturally after the invasive species has been cleared (Elser et al., 2008; Ndou and Ruwanza, 2016). The goal of the WfW is to improve the integrity of natural resources through stopping invasion by new species, detecting, and effectively responding to emerging species, and managing the impacts of the established IAP (Fabricus et al., 2016). However, recent reviews of the programme have shown that WfW has failed to implement active restoration measures that would facilitate the recovery of soil and native vegetation (van Wilgen and Wannenburgh, 2016). Therefore, despite the time and money invested in research and the control of IAP in South Africa, several post-clearance empirical studies have revealed the persistence of the impacts induced by invasion on soil properties (Ruwanza et al., 2013; Kerr and Ruwanza, 2016; Ndou and Ruwanza, 2016; Ruwanza and Shackleton, 2016; Nsikani et al., 2017; Ruwanza and Tshililo, 2019; Gwate et al., 2021). These above-cited studies quantified the effects IAP such as *Acacias, Eucalyptus,* and *L. camara* clearing on soil properties and have reported mixed results such as decreases in soil nutrients such as P, N, and C (Kerr and Ruwanza, 2016; Ndou and Ruwanza, 2016; Ruwanza and Tshililo, 2019; Gwate et al., 2021) and increase in soil moisture content and water repellency (Ruwanza and Tshililo, 2019). On the other hand, Ndou and Ruwanza (2016) recorded declines in soil moisture content and water repellency after *A. longifolia* removal, however this was several years after plant clearing.

Although these above-mentioned studies on IAP invasion and clearing indicate that residuals of invasion remain in the soil long after clearing, limited studies have been done to assess below-ground properties and processes after the clearing of *A. dealbata*, a dominant invader in the rural Eastern Cape. To my understanding, no study has been done to quantify soil properties post *A. dealbata* clearing in South Africa. Lazzaro et al. (2014) studied changes in soil after invasion by *A. dealbata* in Italy and their results were consistent with those of the above-cited studies in South Africa, although these are for different *Acacias*. Lazzaro et al. (2014) further echoed that *A. delabata* removal increases acidification, shifts bacterial and fungal communities, shifts plant communities, and the long-term alteration of soil nutrients.

Once modified by IAP, soil properties may take many years to be successfully repaired to preinvasion state (Ndou and Ruwanza, 2016; Nsikani et al., 2017; Fill et al., 2018). Invasive alien plants such as *A. longifolia* have high litter production which alters soil properties such as C and N pools after over 20 years of invasion (Marchante et al., 2009). Therefore, even after removal of the IAP, soil C and N pools may remain unchanged because of the legacy effect which is linked to invasion extent and duration (Marchante et al., 2009). Soil legacy effect which happens when impacts of the invading species change soil chemistry and microbial communities and these effects remain in the soil for years after the species has been removed (Marchante et al., 2009). This delays the recovery of soil processes and may lead to further invasion by secondary invaders (Stricker et al., 2015). The establishment of secondary invaders further leads to the persistence of the soil nutrients that suppress the recovery of the desired soil properties. The magnitude and intensity of the impacts of *Acacias* on both below and above-ground communities necessitates more quantitative studies to assess the long-term effects of *A. dealbata* invasion on soil recovery following its clearing.

Most comparative studies looked at the differences between the invaded, cleared, and uninvaded areas (Ruwanza et al., 2013; Ndou and Ruwanza, 2016; Ruwanza and Tshililo (2019). However, these studies often fall short in evaluating causation since there may be several other factors influencing the results e.g., the pre-invasion ecosystem (Ruwanza and Tshililo, 2019). According to Stricker et al. (2015) and Ruwanza and Tshililo (2019) comparing soil properties before and after invasion would give more insight into how restoration efforts should be, however, this is difficult given that most invasions happened several years ago. In this field based comparative study, I assessed soil physico-chemical properties between three clearing treatments, namely, cleared, invaded, and uninvaded, following the clearing of *A. dealbata*. The specific research question is, how do soil physico-chemical properties change after the clearing of *A. dealbata*? Comparing these treatments will be useful because they allow theorizing ecological restoration trajectories and inform managers on how to facilitate restoration of soil properties after alien plant clearing (Ruwanza and Tshililo, 2019).

3.2. Methods

3.2.1. Study area

See section 1.3 in Chapter 1 for the study area details.

3.2.2. Experimental design

The experimental design consisted of four sites that were approximately 5 km apart to allow for independence between sites. At each site, an *A. dealbata* invaded area, a cleared area, and uninvaded area were selected (they were referred to as clearing treatments). These treatments, in each site, were approximately 100 m apart from each other. Within each treatment, at each site, six 25 m^2 plots (5 m x 5 m) were laid out where detailed soil measurements were done monthly from December 2020 to February 2021. In total, 72 plots were surveyed, (four sites x three treatments x six plots). The *A. dealbata* invaded sites were dominated by mature stands of the plant with a percentage cover of over 75%, and these were areas not cleared by the WfW clearing programme since clearing was done in patches. The uninvaded sites were dominated by native grasses, and herbaceous plants with a cover of approximately 80%. The uninvaded sites served as the control sites. Grasses that dominated the uninvaded sites are *Sporobolus indicus, Paspalum dilatatum,* and *Cynodon dactylon.* The *A. dealbata* cleared areas sites had *A. dealbata* cleared by WfW in 2016 except for one site which was cleared in 2018. The clearing involved mechanical cutting of *A. dealbata* close to the base. The felled (cut) trees were stacked on site and most of the cut stumps were harvested by the residents for fuelwood. The remaining big stumps were burnt on site and all burnt patches were avoided in this study.

3.2.3. Soil physico-chemical properties sampling

Soil surveys aimed at measuring soil physico-chemical properties were done once in each month over a three-month period, i.e., December 2020, as well as January, and February 2021. Soil physical properties were measured monthly for the above-mentioned months; however, soil chemical properties were measured once due to financial limitations and the assumption is that no variations in chemical properties will be observed within one season. On each of the above-mentioned plots, five samples of soil were collected from the four corners and at the centre, at the depth of 10 cm. Only the first 10 cm depth was collected because it has the primary nutrients that are needed to support plant growth (e.g. grasses). Financial constrains also did not allow for the collection of soil in deeper level. Firstly, litter was removed by hand and then the soil cores were collected using a hand trowel. A 10 cm^2 steel square was used to mark the area where soil samples were taken. The soil samples from each plot were bulked for physico-chemical property analysis. To analyse the following soil properties pH, P, N, C, K, Na, Ca and Mg, samples were taken for chemical analysis at the commercial laboratory, BemLab (PTY) LTD in Somerset West in the Western Cape Province of South Africa. Assessing these primary nutrients is essential because they are needed in significant quantities to support plant growth, and soil P, C and N are indices of soil fertility (Ndou and Ruwanza, 2016). Furthermore, Costantini et al. (2016) states that these properties are critical indicators of soil recovery following disturbance. To avoid any loss of soil nutrients during transportation, the samples were stored in airtight containers.

Soil pH was measured in 1:5 soil-KCl extracts, whereas exchangeable cations of K, Na, Ca, and Mg were extracted in a 1:10 ammonium acetate solution using the centrifuge procedure and analysed by atomic absorption spectrometry. Phosphorus was measured using a Bray-2 extract method and total N was analysed by complete combustion using a EuroVector Euro EA Elemental Analyser. Total C was measured using a modified Walker-Black method (Chan et al., 2001).

Gravimetric soil moisture was assessed by weighing the wet soil cores, after which they were dried in an oven at 105 $^{\circ}$C for 72 hours and then reweighed again to record water content. Soil penetration resistance level was measured in the field approximately 30 cm from the centre where the soil samples have been collected. These were measured by piercing a pocket penetrometer into the soil, where the metal ring was pushed up to mark the resistance value in kg/c $^{-2}$ (Leung and Meyer, 2003). Four measurements were taken at each plot in the field and were averaged to represent soil penetration resistance per plot.

Infiltration rate in the soil was measured in the field on a flat surface using a mini-disk infiltrometer. The infiltrometer is an acrylic tube with two chambers, semipermeable plastic disk, a suction tube inside, and a rubber stopper (Latorre et al., 2013). For this study, both chambers of the infiltrometer were filled with clean water and the suction rate was set at 2.0 cm, which is ideal for clay loam soils (Zhang, 1998). On a flat surface, about 30 cm from the centre of each plot, litter was removed by hand and the infiltrometer was placed. Readings were recorded from the drop of water level in the lower chamber in mL at 30 seconds intervals for five minutes. Cumulative infiltration rate over time were obtained following methods by Zhang (1998). Hydraulic conductivity was calculated using the infiltration data using the van Genuchten-Zhang method (Zhang, 1998).

Water repellence was tested using the Water Droplet Penetration Time (WDPT) method which measures the time it takes for repellence to persists on a porous surface (Doerr and Thomas, 2000). Soil samples were sieved and air-dried for seven days in controlled laboratory conditions in petri dishes and measured for WDPT. Testing for WDPT was done by placing five droplets of distilled water using a hypodermic syringe on the soil surface and record how long it takes for the water to penetrate the soil. An average time of penetration was used to represent the WDPT for each soil sample. For this study, soil samples were classified based of the repellency classes according to Bisdom et al. (1993) (see Table 3.1).

Table 3.1. Classification of soil water repellency based on the Water Droplet Penetration Time (WDPT)method.

Classification	The Water Droplet Penetration Time (s)
Non-repellent	< 5
Slightly water repellent	6 – 60
Strongly water repellent	61 – 600
Severely water repellent	601 – 3 600
Extremely water repellent	> 3600

3.2.4. Statistical analysis

The data from all the plots in all treatments were averaged to avoid pseudo-replication. Soil chemical measurements among the three treatments were analysed using one-way ANOVA in Statistica version 13.4 since data was collected once. Effects of clearing on soil properties such as soil moisture, soil penetration resistance, infiltration rates, and hydraulic conductivity were analysed using the two-way ANOVA since data was collected over three months. To determine the differences between clearing methods, Tukey's HSD unequal *n* test, at $P < 0.05$, was used. Soil water repellency was analysed using the Chi-squared test based on WDPT categories.

3.3. Results

3.3.1. Effects of clearing on soil chemical properties

Approximately 67% of the soils in the study areas were sandy, 30% were loamy and only 3% were clay soils. The effect of *A. dealbata* invasion and clearing on soil chemical nutrients did not show much variation since most chemical properties did not show statistical difference. Although the soils in all the treatments were acidic, the soil pH was significantly higher (p = 0.003) in the uninvaded treatments (mean = 4.18 ± 0.11) than the cleared (mean =3.80 ± 0.08) and invaded treatments (mean = 3.8 ± 0.04) (Table 3.2). Soil resistance also indicated a significant difference (p = 0.009) among the treatments with the uninvaded treatments having the higher (mean = 2473.33 ± 266.6) resistance than the cleared (mean= 1612.50 ± 173.73) and invaded (mean =1729.17 ± 143.68) treatments (Table 3.2). Total soil N, total C, P, H, and all the exchangeable cations (K, Na, Ca, and Mg) showed no significant differences among the treatments (p > 0.05) (Table 3.2).

Table 3.2. Comparison of soil physical and chemical properties between the cleared, invaded, and uninvaded treatments. Data are means ± SE. Columns with different letter superscripts are significantly different.

	Clearing treatments			**One-way ANOVA**	
Soil properties	**Cleared**	**Invaded**	**Uninvaded**	***F* - values**	***P* - values**
pH (KCL)	3.80 ± 0.08^{b}	3.80 ± 0.04^{b}	4.18 ± 0.11^{a}	6.869	0.003
Resistance (ohm)	1612.50 ± 173.73^{b}	1729.17 ± 143.68^{b}	2473.33 ± 266.83^{a}	5.361	0.009
H+ (cmol/Kg)	3.02 ± 0.42^{a}	2.97 ± 0.22^{a}	2.02 ± 0.48^{a}	2.089	0.14
P Bray II (mg/kg)	4.80 ± 0.98^{a}	2.73 ± 0.26^{a}	5.21 ± 1.59^{a}	1.495	0.239
C (%)	3.14 ± 0.37^{a}	3.32 ± 0.22^{a}	2.71 ± 0.56^{a}	0.586	0.562
N (%)	0.17 ± 0.02^{a}	0.21 ± 0.27^{a}	0.15 ± 0.03^{a}	1.346	0.274
Exchangeable cations					
Ca (cmol/kg)	2.86 ± 0.74^{a}	1.86 ± 0.25^{a}	2.88 ± 0.22^{a}	1.57	0.23
Mg (cmol/kg)	1.41 ± 0.48^{a}	0.87 ± 0.11^{a}	1.36 ± 0.15^{a}	1.016	0.373
K (cmol/kg)	0.30 ± 0.04^{a}	0.42 ± 0.06^{a}	0.29 ± 0.04^{a}	1.936	0.16
Na (cmol/kg)	0.10 ± 0.12^{a}	0.13 ± 0.01^{a}	0.14 ± 0.01^{a}	2.55	0.093

3.3.2. Effects of clearing on soil moisture content and soil penetration resistance

The gravimetric soil moisture content showed significant variations among the clearing treatments and among the months (p < 0.001) (Figure 3.1a). These differences among the clearing treatments were observed in December and February, but not in January. In December, gravimetric soil moisture was significantly higher in the cleared treatments (mean = 70.17 ± 24.33) than in the invaded (mean = 10.60 ± 1.62) and uninvaded treatments (mean = 11.64 ± 1.98) (Figure 3.1a). In February, the uninvaded treatments had significantly lower (mean: 19.45 ± 1.31) than the invaded (mean = 28.80 ± 5.44) treatments (Figure 3.1a). Comparison among the months showed that December had significantly lower (p < 0.05) soil moisture compared to January and February, except for the cleared treatment in December (Figure 3.1a). Soil penetration resistance levels were significantly higher (p < 0.001) in December than January and February (Figure 3.1b). Comparisons among the clearing treatments showed that all the treatments in December had significantly higher (p < 0.05) soil penetration resistance than in January and February.

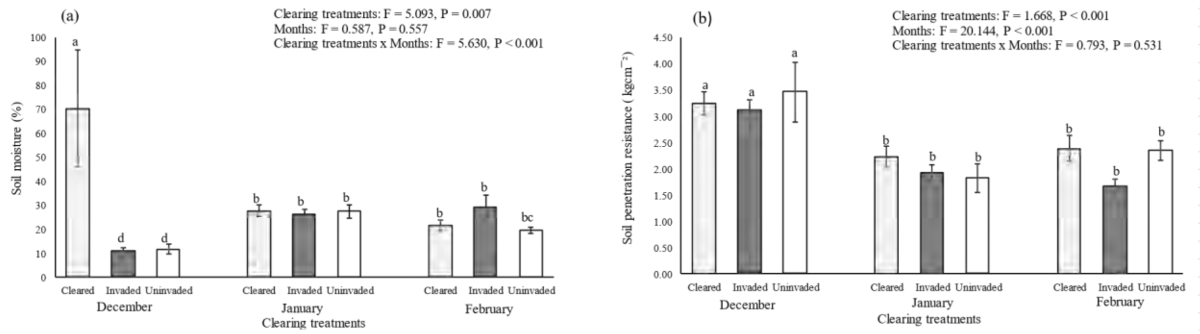

Figure 3.1. Results showing (a) gravimetric soil moisture content expressed in percentage and (b) soil penetration resistance expressed in kgcm^{-2}. Bars represent mean ± SE. ANOVA results are shown. Bars with different letter superscripts are significantly different ($p < 0.05$).

3.3.3. Effects of clearing on soil infiltration rate and hydraulic conductivity

The infiltration rates varied significantly among the clearing treatments ($p < 0.05$) and among the months ($p < 0.05$), but cross comparison among the clearing treatments and the months showed no significant difference ($p = 0.65$) (Table 3.2). The above-mentioned significant differences on infiltration rates were observed in all three months with December recording the highest infiltration rates than the other months for all treatments. Infiltration rates were significantly higher in cleared treatments than invaded and uninvaded treatments. Soil hydraulic conductivity showed no significant ($p > 0.05$) difference among the clearing treatments. However, comparisons among the months showed significant ($p = 0.006$) differences (Figure 3.2). Cross comparison among the clearing treatments and the months also showed no significant difference ($p = 0.006$) (Figure 3.2).

Table 3.3. Comparisons of average soil infiltration rate between the cleared, invaded and uninvaded clearing treatments over a three-month period. Data are means ± SE. Factorial ANOVA results are shown (* = significantly different).

	Clearing treatments			**Factorial ANOVA**		
Months	**Cleared**	**Invaded**	**Uninvaded**		***F*- values**	***P*- values**
December	12.16 ± 1.41	7.04 ± 0.99	7.50 ± 2.16	Clearing treatments	17.293	0.000*
January	4.25 ± 1.01	2.54 ± 0.60	2.25 ± 0.55	Months	7.319	0.001*
February	6.88 ± 1.67	1.75 ± 0.37	2.13 ± 0.71	Clearing treatments x Months	0.612	0.654

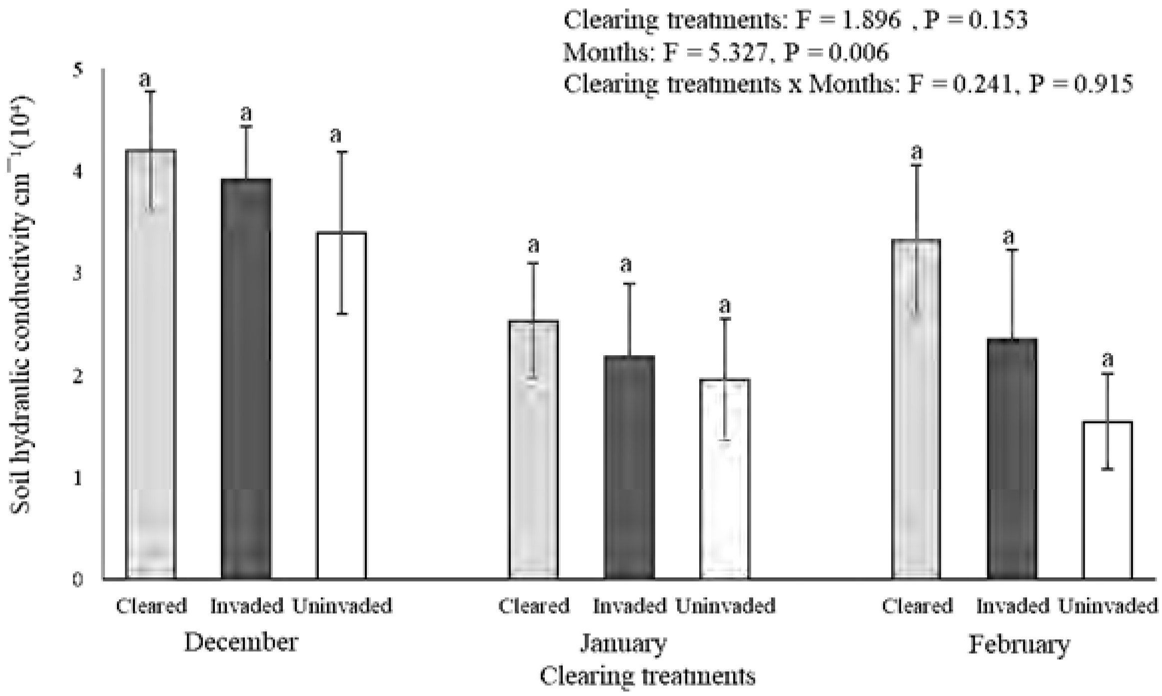

Figure 3.2. Soil hydraulic conductivity levels in the soil samples taken from the cleared, invaded and uninvaded treatments in three different months. Bars represent mean ± SE. ANOVA results are shown. Bars with different letter superscripts are significantly different ($p < 0.05$).

3.3.4. Effects of clearing on soil water repellency

The chi-squared test of WDPT classes showed no significant ($p = 1.00$) differences among the three clearing treatments and the months (Figure 4). In all the treatments, all the months had over 70% of non-repellent soils and the remaining soils were slightly repellent. In December, the uninvaded treatments had the least repellent soils with 98% soils being non-repellent (Figure 3.3).

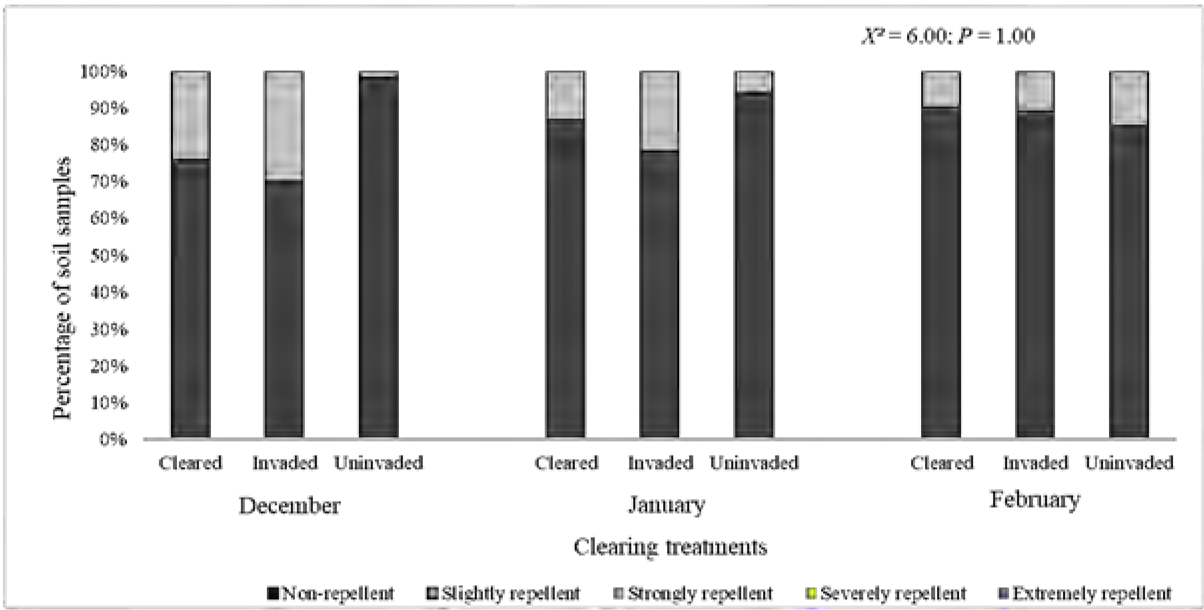

Figure 3.3. Distribution of soil water repellency classes (based on Water Droplet Penetration Time method) in soil samples taken from the cleared, invaded and uninvaded treatments. Chi-squared results are shown.

3.4. Discussion

This study sought to assess the impacts of *A. dealbata* clearing on soil physico-chemical properties. Results from this study support previous studies which investigated the impacts of *Acacia* clearing on soil properties (Marchante et al., 2009; Ndou and Ruwanza, 2016; Ruwanza and Tshililo, 2019; Gwate et al. 2021), which suggest that the clearing of *Acacias* triggers varied changes in soil properties such as decreases in some soil nutrients and increases in some soil physical properties. The clearing of *A. dealbata* increased soil physical properties such as soil moisture content, soil penetration resistance, soil infiltration rate, and soil hydraulic conductivity. On the other hand, this study showed no significant changes in soil chemical properties, although there are decreasing trends observed. Soil pH remained the same in cleared and invaded treatments, but this was significantly lower compared to the uninvaded treatments. These results concur with findings by Ndou and Ruwanza (2016) and Marchante et al. (2009) who also recorded decreases in some soil nutrients after *Acacia* clearing.

3.4.1. Effects of clearing on soil nutrients/ chemical properties

All the treatments had acidic soils (soil pH less than 7), however, the uninvaded treatments had significantly higher pH levels than the cleared and invaded treatments. These findings are consistent with previous studies which indicated that *Acacia* invaded sites had more acidic soils than uninvaded and cleared sites (Lorenzo et al., 2010; Lazzaro et al., 2014; Kerr and Ruwanza, 2016; Gwate et al., 2021). The soils in the study areas are derived from sandstones, which have poor nutrient retention capacity, thus the observed acidity (Lewis and Illgner, 1998; Kerr and Ruwanza, 2016). Soil acidity can also be attributed to podsolization which is linked to release of organic matter and ions during soil formation (Starfinger et al., 2003; Kerr and Ruwanza, 2016). Contrary, Nsikani et al. (2017) recorded significantly lower pH (more acidic) in uninvaded treatments compared to cleared and invaded treatments. The above-mentioned study, Nsikani et al. (2017) assessed the period it takes for soil legacy effects to persist after the clearing of *A. saligna* by comparing invaded, cleared, and non-invaded sites. According to Nsikani et al. (2017) the decomposition of large amounts of litter that *Acacias* add to the soils may induce an increase in soil pH.

Kerr and Ruwanza (2016) state that in soils with pH less than 5.5, soluble P is likely to form insoluble compounds with Al and Fe, which cannot be accessed by plants, therefore, leaving high concentrations of P in soils. This view contradicts with my results because the uninvaded treatments had higher (although not statistically significant) soil P concentrations than cleared and invaded treatments which would be expected to have more soil P due to them being more acidic. Furthermore, Lazzaro et al. (2014) stated that a decrease in soil pH due to invasion, leads to a decrease in soil P content. The contradiction between these two statements means that, pH alone, cannot fully explain the changes in soil P and potentially other soil properties. Hence, it is vital to be cognizant of other local factors such as site soil properties, and variations in soils and geology when analysing soil changes following alien plant clearing (Gwate et al., 2021). Consistent with Gaertner et al. (2011) and Nsikani et al. (2017), my study observed no significant differences in P availability between the different clearing treatments. However, it is worth noting that slight increase in soil P were observed in the cleared treatments than the invaded treatments. Nsikani et al. (2017) linked the increase in soil P content after clearing to the amounts of organic matter left behind by *Acacia* after clearing because during decomposition, P is released into the soil. Secondary invaders and native weedy species may also increase soil P concentrations in the soil (Nsikani et al., 2017). Indeed, there were secondary invaders and herbaceous plants recorded in the cleared treatments (see results in Chapter 4) which may have influenced the soil P concentration.

According to Gwate et al. (2021), Ca and Mg are promoted more in soil with low pH levels, hence it made sense for the cleared treatments to have higher content of these two soil properties, although there were no significant differences in comparison to other treatments. On the other hand, although there were no statistically significant differences for total C, total N, K, and Na, these properties were lower after clearing and these findings were consistent with previous studies (Nzila et al., 2004; Lazzaro et al., 2014; Kerr and Ruwanza, 2016; Ndou and Ruwanza, 2016; Gwate et al. 2021). Like my findings, O'Connell et al. (2004) and Goncalves et al. (2007) also did not get significantly different results but still documented a decrease in soil C after the clearing of *Eucalyptus* and removal of its residue from site. Similarly, Nsikani et al. (2017) also recorded no significant difference in N between the invaded and uninvaded sites. However, on one of their sites, Nsikani et al. (2017) observed significant declines in N after clearing. The decrease is soil C and N indicate a general decrease in soil fertility because these are primary nutrients indicating soil fertility (Berthrong et al, 2009; Kerr and Ruwanza, 2016). Factors that explain these nutrient changes or increase the fertility of the soils are usually litter availability and the quality and rate of decomposition (Kerr and Ruwanza, 2016). The above statement was proven to be true by the findings of this study, which revealed that the cleared treatments had less litter cover than the invaded treatments.

Being N-fixing species, it makes sense for *Acacia* litter to have higher total N concentrations in invaded areas (Corbin and D'Antonio, 2004, Marchante et al., 2009; Lazzaro et al., 2014). Moreover, *Acacia* species grow at faster rates (Ehrenfeld, 2003; Lazzaro et al., 2014), meaning they deposit dead leaves more frequently than native species. This litter and other detritus material under *A. dealbata* canopies could be used to justify the high levels of C and N in the invaded areas (Lazzaro, et al., 2014). The magnitude of the *Acacia* impact on N is dependent on the leaf chemistry of the *Acacia* species in question (Nsikani et al., 2017). The decline in soil fertility after clearing could also be caused by erosion and displacement, leaching, clearing intensity such as off-site debarking, and management harvesting such as burning or clearing residues removal from site (Gomez-Rey et al., 2008; Reitz, 2010; Titshall et al., 2013).

3.4.2. Effects of clearing on soil moisture

This study recorded no significant differences in soil moisture content among the months, except for December, in which, surprisingly, the cleared treatments showed significantly higher soil moisture content than the invaded and uninvaded treatments. These findings were consistent with those of Yapi et al. (2018), where they found that sites densely invaded by *Acacia* had soil moisture content reduced by approximately 5%. During the sampling period of this study, there were heavy rainfalls which may have influenced the results, leading to high moisture content in cleared sites. Cleared sites in this case would have more moisture content because there is more surface runoff compared to invaded and uninvaded sites that have soil moisture absorbed by vegetation. Moreover, December also recorded significantly lower moisture content in the invaded and uninvaded treatments compared to January and February. Recent studies on the impact of *Acacia* removal on soil properties indicate that the clearing of *Acacia* leads to decreased soil moisture content (Ndou and Ruwanza, 2016; Ruwanza and Tshililo, 2019). Both these studies looked at the recovery of soil properties following the clearing of *Acacia*. Ruwanza et al. (2013a) and Kerr and Ruwanza (2016) also recorded a decrease in soil moisture content after the clearing of the invasive *Eucalyptus*. High moisture content in the invaded treatments is supported by the argument that high tree density promotes the retention of water by soils, meaning soils in invaded areas will have higher water holding capacity than soils in cleared areas (Wang et al., 2013; Schoonover and Crim, 2015). *Acacias* have roots that penetrate deeper into the soil, therefore have an advantage of accessing water from the water table (Orwa et al., 2009). This allows the plant to transport water from the wet soil profiles to drier soil profiles, a process called hydraulic redistribution (Leffler et al., 2005; Brooksbank et al., 2011; Ruwanza and Tshililo, 2019).

3.4.3. Effects of clearing on soil penetration resistance

Like soil moisture content, the observed significant differences in soil penetration resistance were recorded only in the month of December. There were no significant differences among the clearing treatments in January and February, however, December recorded higher soil penetration resistance in all the treatments than the other months. Even so, soils in cleared treatments had generally higher soil penetration resistance than soils in invaded treatments, meaning that clearing of *A. delabata* increased the compaction of the soil. In his study, Ruwanza (2020) found that cleared soils were more compact than uninvaded soils. Ruwanza et al. (2015) links these findings to the fact that *Eucalyptus* leaves produced allelopathic compounds that react with soil moisture leading to compact soil in cleared treatments. Worthy of note here, is that soil penetration resistance was higher in treatments that recorded significantly lower soil moisture (in December). Observed changes in moisture in January and February could explain the decrease in soil resistance in those months. Ruwanza and Tshililo (2019) also observed higher soil penetration resistance in cleared sites which had significantly lower soil moisture. This inverse relationship between soil moisture and soil penetration resistance was also reported by Ruwanza and Shackleton (2016).

Studies that looked at the impacts of mechanical harvesting on soil properties also echo the argument that penetration resistance can be influenced by soil water content; thus, it varies seasonally, with soils becoming less compact (less resistant) in periods of high moisture content (Smith, 2003; Rietz, 2010; Titshall et al., 2013). According to Smith (2003), there are limitations in using soil water content to ascertain factors influencing soil penetration resistance due to localized variations in soil water content. Thus, looking at combinations of different factors such as, soil type, bulk density, management approaches and organic matter is crucial because these play a pivotal role in influencing soil penetration resistance (Smith, 2003; Rietz, 2010; Titshall et al., 2013; Ruwanza, 2020). For instance, sandy and loamy soils reduce the available soil water capacity, therefore, tend to be more compact (Rietz, 2010; Titshall et al., 2013). Indeed, soils in the study area were mostly sandy and loamy, hence they had higher penetration resistance (compaction). Furthermore, sandy soils have higher density than clay and silt soils which have more pore spaces; thus, sandy soils will be more compact (Smith, 2003). Clearing methods such as mechanical clearing triggers soil compaction, thus soil penetration resistance (Smith, 2003; Rietz, 2010; Titshall et al., 2013). During mechanical clearing, the machinery weight and movement increases compaction by decreasing the effectiveness of ground pressure (Titshall et al., 2013).

3.4.4. Effects of clearing on infiltration rate and hydraulic conductivity

The results of this study showed significant differences in average infiltration rates among the clearing treatments and the months. Significantly higher infiltration rates were observed in the cleared treatments compared to the invaded and uninvaded treatments. This study indicated no significant differences in hydraulic conductivity among the clearing treatments, but indicated significant differences among the months, with December having the highest hydraulic conductivity. Although there was no significant difference among the treatments, the cleared treatments had generally higher hydraulic conductivity and the uninvaded sites had the least hydraulic conductivity. These were consistent with a study by Ruwanza and Tshililo (2019) whose study assessed the recovery of soil properties after *A. mearnsii* removal. Ruwanza and Shackleton (2016), yielded similar results of recording a significant difference in one month only (out of three months) when they assessed the differences in soil properties between invaded and natural sites in *L. camara* invaded sites. In that month, the invaded sites had significantly higher hydraulic conductivity than the natural sites. They attributed their findings to the presence of biotic factors such as earthworms in the soils (Ruwanza and Shackleton, 2016). This study did not measure any presence or absence of living organisms in the soil; however, it draws from the argument that the presence of earthworms causes changes soil structure, water movement and soil nutrients (Fischer et al., 2014; Ruwanza and Shackleton, 2016). Earthworm activity is influenced by soil moisture and temperature, with wetter soils having more activity than drier soils (Ruwanza and Shackleton, 2016). When earthworms burrow, they cause an increase in soil infiltration and hydraulic conductivity. Indeed, cleared treatments had higher soil moisture in December, which could have induced more earthworm burrowing, hence the higher hydraulic conductivity in December. Contrary, Ruwanza (2020), indicated that uninvaded sites had significantly higher hydraulic conductivity than cleared sites.

3.4.5. Effects of clearing on soil water repellency

Results from this study showed no significant differences in soil water repellency among the clearing treatments and the months. A large percentage of the sampled soils were non-repellent, and this could be attributed to soil moisture content. Ruwanza and Thsililo (2019) state that soils with low moisture content are likely to be more repellent. Because this study was conducted during the rainy season, this could have influenced the soils to be less repellent. Another thing worthy of note is that the cleared treatments are intermediate between the invaded and the uninvaded treatments. According to Ruwanza (2017), invasion by *Acacia* causes soils to be repellent, therefore there clearing of the species can be expected to make soils less repellent (Ruwanza and Tshililo, 2019). This statement suggests that the abovementioned intermediate trend in soil repellency could mean that the clearing of *A. dealbata* results in soils becoming less repellent. Furthermore, it is argued that sandy soils tend to be more repellent (Ruwanza et al., 2013; Ruwanza and Shackleton, 2016). Indeed, the soils in the sites were sandy and loamy, and that could explain the small proportion of slightly repellent soils observed in this study. Furthermore, repellency of soils can be induced by the heat and drying that are linked to evapotranspiration (Ruwanza and Shackleton, 2016). This argument could also support the proportion of soils that were slightly repellent since the study was done during summer months when temperatures vary between days due to rain.

3.5. Conclusion and implications for restoration

The magnitude of the effect of IAP on soil properties differ depending on the species in question, the site dynamics, and the soil properties of the study site. This study showed that the effects of *A. dealbata* on soil properties vary and can be long-lasting even after the species has been cleared because there were properties that showed no significant differences between the cleared and the invaded treatments. The sites samples in this study were cleared in 2016, therefore, after 4 years of clearing, some soil properties were similar to the invaded treatments, e.g., soil N, P, C, water repellecy, and hydraulic conductivity. Furthermore, the study was done as a once-off study, so there is no data on different years after clearing. Future research needs to assess and compare trends and changes in soil properties over different years after clearing of Acacias. Several other studies have documented persistence of altered soil properties years following the removal of IAP and linked the findings to soil legacy effect (Marchante et al., 2009; van Der Waal, 2009; Nsikani et al., 2017). The persistence of these legacy effects years after clearing insinuates that the passive restoration efforts by WfW are not sufficient to stimulate soil recovery. Furthermore, the sites in this study did not have consistent follow-up monitoring after the initial clearing, therefore, re-invasion by *A. dealbata* might be influencing the persistence of the reported soil properties (see Chapter 4 which confirms *A. dealbata* re-invasion). This study also recorded significant increases in some soil properties while also recorded significant declines in some. Despite the mixed results from different studies cited in this chapter, these studies also present hope that soil properties can still be recovered after clearing. Nsikani et al. (2017) state that soil pH can be recovered because two of their sites were showing trends of moving towards uninvaded levels. However, this study proved that the legacy effect of *A. dealbata* can persist even four years after clearing because the pH in the cleared treatments remained the same as in invaded treatments. Some soil properties, such as the soil water repellency, compaction, soil moisture, hydraulic conductivity and infiltration showed positive trends of leaning towards similar conditions as the uninvaded treatments after clearing. This means that given enough time and with sufficient investment in active restoration, full recovery of soils can be achieved.

From an intervention standpoint, results of this study seem to suggest the following. Firstly, since some soil nutrients did not change significantly after four years of clearing, clearing of IAP needs to be supplemented with active restoration to stimulate soil nutrient recovery after clearing. This can be achieved through soil ploughing or fertilizer addition in the cleared areas to supplement the needed soil nutrients and/or to induce an increase in soil P, C and cations. However, the above-mentioned needs further studies. Secondly, the need for proper implementation of follow-up clearing treatments to effectively control *A. dealbata* in cleared sites and possibly other secondary invaders that might be influencing soil property changes, is emphasized. Thirdly, there should be clear understanding of landscape ecology with achievable and feasible active restoration goals set and progress should be monitored within a framework of adaptive management (van Wilgen, 2012). Restoration ecology provides this understanding of systems, succession and the different methods that can be used for rehabilitation of soils, ecosystem functions, and ultimately restore biodiversity (Dobson et al., 1997). Further research needs to be done to investigate how below-ground communities and micro-climates are impacted by invasion because below-ground organisms such as earthworms and termites play a significant role in soil properties. This would supplement the information from changes in soil chemical properties.

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Chapter 4: Vegetation recovery following *Acacia dealbata* clearing: Implications for ecological restoration

Abstract

Invasive Australian plants such as *Acacia dealbata* have detrimental effects on biodiversity through their ability to alter ecosystem structure and functioning, thus threatening ecosystem service delivery and ultimately human well-being. In response to alien plant invasion, several control mechanisms such as mechanical clearing have been implemented across the globe. However, studies have examined native plants recovery following alien plant clearing and previous studies have reported that clearing programmes fall short in terms of promoting native vegetation recovery in the cleared sites. This study assessed whether the clearing of *A. dealbata* leads to the recovery of native vegetation in the Tsitsa catchment, Eastern Cape Province of South Africa. Vegetation surveys were done in three clearing treatments; namely, cleared, invaded, and uninvaded, and each treatment had six plots which were replicated four times. Results showed little recruitment of native grasses and forbs, but persistence of *A. dealbata* seedlings in the cleared treatments. Species richness and Shannon-Wiener diversity index were significantly ($p < 0.05$) higher in the cleared and invaded treatments than the uninvaded treatments, and this was more visible for trees and shrubs. Cover for all species was significantly higher in the uninvaded than the cleared and invaded treatments. This study observed the recruitment of some native plant species in the cleared treatments that were not present in the invaded treatments. Therefore, concludes the recruitment and establishment of some native species, mostly grasses, in the cleared treatments gives assurance that passive restoration is in a positive vegetation trajectory that can lead to recovery of native vegetation after *A. dealbata* clearing. Furthermore, the study recommends that clearing must be supplemented with active restoration to fast-track vegetation recovery, and that investing in ecological restoration after alien plant clearing is a necessity for complete ecosystem recovery.

Key words:

Invasive alien plants, alien plant control, passive restoration, vegetation recovery, plant species diversity.

4.1. Introduction

Globally, there has been a growing interest in biological invasion studies, especially invasion by woody trees, due to the escalating impacts of invasions on biodiversity and ecosystem function (Rouget et al., 2004; Lukey and Hill, 2020; van Wilgen et al., 2020). Invasion by invasive alien plants (IAP) continue to be one of the pressing environmental issues threatening many biomes in South Africa, although efforts such as plant clearing have been intensified since 1995 (van Wilgen, 2012). Invasive alien plants are defined as plants that are not native to the ecosystem under consideration, whose introduction causes or is likely to cause environmental, socio-economic harm, or harm to human health (Richardson et al., 2011). Invasive alien plants create self-sustaining populations and produce large amounts of reproductive offspring, and thus have potential to spread over large areas of land (Richardson et al., 2000). The compromised structure, functioning and biodiversity of ecosystems due to invasion by IAP further affect the ability of the ecosystems to produce and deliver ecosystem services (Le Maitre et al., 2011; van Wilgen, 2012). Furthermore, IAP pose a potential threat to human health and to the economy (Will and Hulme, 2002). For example, recent studies have suggested that invasion by IAP costs South Africa an estimated amount of over R15 billion per annum (van Wilgen et al., 2020). Although, they were initially introduced purposefully for different reasons (Mac Donald et al., 2004; Kull et al., 2011; Le Roux et al., 2011; van Wilgen et al., 2020), most IAP, such as *Acacia dealbata* in South Africa began spreading beyond designated areas to areas set aside for conservation and water production (De Witt et al., 2001). They are also known to invade human disturbed and transformed areas such as old fields, roadsides, and catchment areas (Kull et al., 2011; van Wilgen et al., 2020), thus causing the failure of such areas to recover unaided.

In 2004, about 10 million ha of land in South Africa had been invaded by IAP, and the socioeconomic problems associated with IAP have been growing ever since (Nel et al., 2014). One of the main IAP in South Africa is *A. dealbata*, commonly known as silver wattle. *Acacia dealbata* is listed amongst the most successful invaders in South Africa because of its competitive ability due to its morphological characteristics (Morris et al., 2011). The plant grows to about five to ten meters and is green throughout the year. Although the plant is found in most parts of the country, *A dealbata* is most problematic in KwaZulu Natal, Eastern Cape, Free State and Mpumalanga. Its native range is South-eastern Australia and Tasmania (Morris et al., 2011; van Wilgen et al., 2020). In South Africa, the plant is listed as a category 2 invader, meaning a permit is required to grow or possess *A. delabata*. Like most invasive *Acacias* in South Africa, *A. dealbata* contributes significantly to the economy of the country as they are used to produce tannin, wood chips and pulp (Kull et al., 2011; van Wilgen et al., 2011). The contribution of *Acacia* species in South African economy is valued at billions of Rands through plantation production (De Wilt et al., 2001; Kull et al., 2011; van Wilgen et al., 2011). Besides the above, *A. dealbata* contributes to human livelihoods through provision of firewood, medicine, livestock fodder, and building material (Ngorima and Shackleton, 2019).

Despite these benefits, the effects of IAP on ecosystems have been recorded in many parts of the world, and these include decline in native species diversity, decreases in stream flows due to increased water consumption, changes in soil nutrient cycling, and changes in fire regimes (Yelenik et al., 2004; Morris et al., 2011; Everson et al., 2014; Le Maitre et al., 2019). For example, recent studies have shown that *A. mearnsii* and *A. dealbata* are amongst IAPs that invade rangelands and alter vegetation structure and soil chemical properties in leading to loss of grazing areas (Yapi et al., 2018; Ngorima and Shackleton, 2019; O'Connor and van Wilgen, 2020). Large portions of agricultural lands are invaded by *A. dealbata* that replace grasslands, thus reducing the carrying capacity of livestock and inhibit other agricultural products and causes declines in the availability of non-timber forest products (NTFPs) (Ngorima, 2016).

In response to the intensifying impacts of IAPs in South Africa, the Working for Water (WfW) programme was launched in 1995 with a goal of increasing water supply by controlling IAP invasion in river catchment areas (MacDonald, 2004). The WfW programme adopted a comprehensive approach of both chemical and mechanical control of the IAP in South Africa (van Wilgen, 2012). However, due to mixed results received from biological invasion and restoration studies, it is still not clearly understood whether the areas targeted for clearing by WfW are of top priority and little or no impact has been made to decrease the rate of invasion (van Wilgen, 2012). *Acacia dealbata,* amongst other IAP, has been observed to persist in areas where it was cleared (Huchzermeyer et al., 2018a). Persistence of these IAP despite clearing, is a cause for concern as their impacts will continue being felt. Previous studies have suggested that passive restoration (i.e., clearing alone) should be supplemented by follow-up active restoration techniques and timeous monitoring after IAP clearing (Ruwanza et al*.*, 2013a).

Few studies have examined how native plant species recover after clearing by WfW. Blanchard and Holmes (2008) examined vegetation recovery after *A. mearnsii* removal and concluded that little vegetation recovery is happening. Several factors have been shown to limit vegetation recovery, e.g., lack of native plants soil seedbanks, secondary invasion, and external environmental factors like hot summers that's suppress seedling growth (Ruwanza et al., 2013; Nsikani et al., 2020). Galatowitsch and Richardson (2005) examined vegetation recovery after alien clearing in riparian systems and reported that active restoration is required if clearing is to yield successful results. Ndou and Ruwanza (2016) showed positive vegetation recovery trajectory after *A. longifolia* removal in Grahamstown, this is likely to show that 10 years after initial clearing is done, vegetation recovery is possible. On the other hand, vegetation recovery after IAP clearing can be facilitated by the presence of remnant native species, seed dispersal by birds, wind, and wild animals, as well as proximity to intact vegetation (Ruwanza et al., 2013a). Indeed, Fill et al. (2017) assessed the short-term recovery of vegetation after invasive alien plant clearing and reported an increase in native species cover, but also, an increase in alien weedy grass cover. The increased cover by re-invading IAP means that restoration goals should focus more on implementing active management (Fill et al., 2017). Therefore, there is a need to evaluate vegetation recovery following IAP clearing to inform WfW clearing as well as to develop effective restoration strategies.

The aim of this field-based study was to examine vegetation recovery following *A. delabata* clearing in the Tsitsa Catchment, Eastern Cape Province of South Africa. To achieve this, this chapter answered the two questions; (1) Does the clearing of *A. dealbata* influence recovery of native vegetation diversity recovery? (2) Does clearing of *A. dealbata* facilitate recovery of vegetation structure and composition? Results from this study aim to inform ecologists, land managers, policy makers and local community stakeholders on effects of *A. dealbata* clearing on vegetation diversity and composition. Insights from this study will guide future passive restoration approaches, particularly the work done by WfW which is based on unaided recovery once the IAP is removed.

4.2. Methods

4.2.1. Study area

See section 1.3 in Chapter 1 for the study area details.

4.2.2. Experimental design

The experimental design consisted of four sites that were approximately 5 km apart to allow independence between sites. At each site, an *A. dealbata* invaded area, a cleared area, and an uninvaded area were selected (they were referred to as clearing treatments). These treatments, in each site, were approximately 100 m apart from each other. Within each treatment, at each site, six 25 m² (5 m x 5 m) plots were laid out where detailed vegetation surveys were done in January 2021. In total, 72 plots were surveyed, (four sites x three treatments x six plots). See detailed in section 1.3 and 3.2.2 regarding the characteristics of the treatments

4.2.3. Vegetation surveys

Vegetation surveys were done in January 2021, thus during the rainy and plant flowering season to maximize plant identification. To measure species diversity in all the clearing treatments, count of all the trees, shrubs, forbs, and grasses present in the plots were conducted. Species were grouped into growth forms, namely, trees, shrubs, forbs, and grasses according to their morphology. Grouping species based on morphology allowed for the assessment of vegetation changes following alien plant clearing (Dorrepaal et al., 2005). Cover of individual plant species were visually estimated (as suggested by Ruwanza, 2020) to the nearest 5% or the 1% when species occupied less than 5% cover. A sample of each of the species present in the plots, even those identifiable in the field, were collected for identification at the Selmar Schonland Herbarium at the Albany Museum at Rhodes University.

4.2.4. Statistical analysis

The effects of different clearing treatments on species diversity were assessed using ShannonWiener index, species richness, and species evenness which were calculated per plot. After testing for normality and homogeneity of variances (for which data was normally distributed), the measured variables (species diversity and cover) were compared using One-Way ANOVA. For significantly different ANOVA. Tukey's test was done to determine differences among individual clearing treatments at $p < 0.05$. All the above variables were analyzed using Statistica version 13.4. Correspondence Analysis (CA) was used to evaluate species composition change based on clearing treatments and it was based on presence-absence data. Correspondence Analysis was conducted using CANOCO version 5.

4.3. Results

4.3.1. Effects of *A. delabata* clearing on species diversity

A total of 20 trees and shrubs, 42 forbs, and 13 grasses were recorded for all the treatments, therefore, for a total of 75 species (Appendix 1). There was a significant difference among the clearing treatments for diversity indices, namely Shannon-Wiener index, and species richness (Table 4.1). Shannon-Wiener index of diversity was significantly ($p < 0.05$) higher in the invaded and cleared treatments than the uninvaded sites. The invaded treatments (mean = 2.03 ± 0.30) had the highest Shannon-Wiener index, followed by the cleared (mean = 1.89 ± 0.44) and uninvaded (mean = 1.75± 0.36) treatments. The species richness also showed statistically significant ($p < 0.05$) differences amongst all three treatments. The invaded (mean = 12.58 ± 3.09) and cleared (mean = 11.88 ± 2.01) treatments had highest means compared to the uninvaded (mean = 9.54 ± 2.54) treatments (Table 4.1). In contrast, species evenness, showed no significant ($p > 0.05$) differences amongst the three clearing treatments.

Species richness for different growth forms varied amongst the three clearing treatments (Table 4.1). In terms of trees and shrubs, there was a significant ($p < 0.05$) difference in species richness amongst the three clearing treatments (Table 4.1). The invaded (mean = 5.42 ± 2.32) and cleared (mean = 3.04 ± 1.73) treatments had the highest trees and shrubs richness than the uninvaded (mean =0.47 ± 1.01) treatments (Tablet 4.1). Forbs showed significantly ($p = < 0.05$) higher richness in uninvaded (mean = 5.75 ± 1.70) and cleared (mean = 4.75 ± 1.42) as compared to the invaded (mean = 4.33 ± 1.76) treatments. In contrast, richness for grasses was significantly higher in cleared (mean =4.21 ± 1.32) and uninvaded (mean = 3.50 ± 1.10) than in invaded (mean = 3.09 ± 1.28) treatments (Table 4.1).

Table 4.1. Species richness, Shannon-Wiener index and evenness based on species abundance counts from different clearing treatments (cleared, invaded and uninvaded). Data are mean ± se and One-Way ANOVA results are shown (*$p < 0.05$). Means with different superscript letters are significantly different.

	Cleared	Invaded	Uninvaded	One-Way ANOVA (F values
Species richness	11.88 ± 2.01^{a}	12.58 ± 3.09^{a}	9.541 ± 2.54^{b}	658.89*
Shannon-Wiener index	1.89 ± 0.44^{ab}	2.03 ± 0.30^{a}	1.75 ± 0.36^{b}	1870.73*
Evenness	0.76 ± 0.08^{a}	0.81 ± 0.12^{a}	0.79 ± 0.57^{a}	658.89
Species richness per growth form				
Trees and shrubs	3.04 ± 1.73^{ab}	5.42 ± 2.32^{a}	0.47 ± 1.01^{b}	81.198*
Forbs	4.75 ± 1.42^{ab}	4.33 ± 1.76^{b}	5.75 ± 1.70a	658.89*
Grasses	4.21 ± 1.32a	3.09 ± 1.28^{b}	3.50 ± 1.10^{ab}	602.12*

4.3.2. Effects of *A. dealbata* clearing on vegetation cover

Vegetation cover for all species was significantly ($p < 0.05$) different amongst the three clearing treatments. The uninvaded treatments (mean = 24.93 ± 1.71) had higher vegetation cover than the invaded (mean = 11.31 ± 1.44) and the cleared (mean = 7.42 ± 0.86) (Figure 4.1a). Cover for trees and shrubs was significantly higher in the invaded (mean = 19.98 ± 2.99) than the cleared (mean = 2.38 ± 0.55) and uninvaded (mean = 3.20 ± 1.23) treatments (Figure 4.1b). Uninvaded (mean = 24.00 ± 2.49) treatments had a higher cover for forbs as compared to the cleared (mean = 7.00 ± 1.15) and invaded (mean = 5.00 ± 1.06) treatments (Figure 4.1c).

Forbs dominated the uninvaded treatments with an average cover of approximately 24% while they were down to 7% and 5% in the cleared and invaded treatments respectively (Figure 4.1c). Similarly, cover for grasses was significantly higher in the uninvaded (mean = 29.87 ± 2.28) treatments as compared to the cleared (mean = 11.45 ± 2.00) and invaded (mean = 4.74 ± 1.42) treatments (Figure 4.1d).

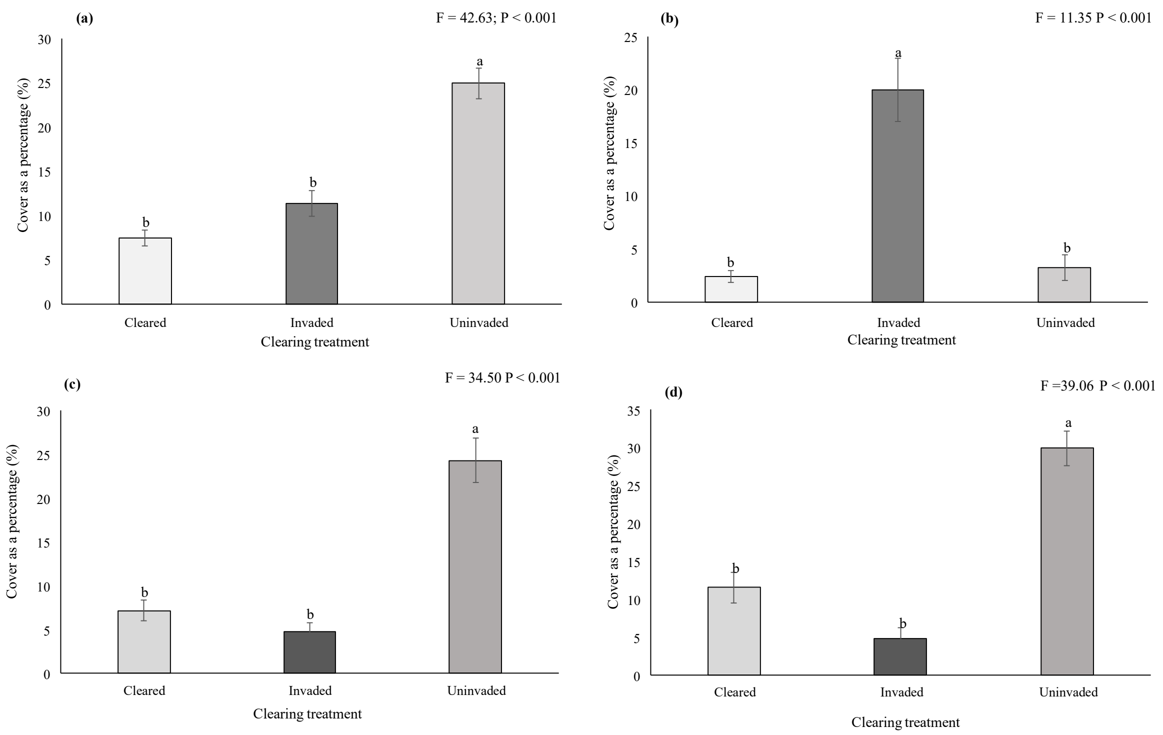

Figure 4.1. Species percentage cover comparisons between there clearing treatments of cleared, invaded, and uninvaded for (a) all species, (b) trees and shrubs, (c) forbs and (d) grasses. Bars indicate mean ± SE and bars with different superscript letters are significantly different at $p < 0.05$. One-Way ANOVA results are shown.

4.3.3. Effects of *A. dealbata* clearing on species composition.

Of all the 75 plant species, forbs constituted over half of the recorded plant species (Appendix 1). Seventeen species were found in all the treatments, these include two shrubs of *Bidens pilosa* and *Cheilanthens viridis,* nine forbs, namely *Helianthus longifolius, H. petiolare, Hypochaeris glabra, Oxalis sp, Plantago lancealata, Senecio sp, Taraxacum laevigatum,* and two unidentified forbs, and six grasses *of S. indicus, S. compositus, C. dactylon, P. dilatatum, A. stricta,* and a *Bulbostyis sp.* Although these species were present across all treatments, forbs had the highest occupancy frequency in the uninvaded treatments while grasses had high occupancy frequency in the cleared treatments. Native grass species of *S. indicus, C. dactylon,* and *P. dilatatum* had high occupancy in the cleared treatments. Of all the trees and shrubs, recruiting *A. dealbata* had the highest species occupancy frequency in the cleared treatments, while *H. petiolare* had the highest occupancy frequency for forbs (Appendix 1).

Clearing of *A. dealbata* had a significant influence in species composition (Figure 4.2). Correspondence Analysis bi-plots of species for all identified species (species with names i.e., positively identified species) indicate some clear species separation (Figure 4.2a). Most grasses such as *C. dactylon, Bulbostylis sp*., and *Agrostis gigantea* assembled more in the cleared than invaded and uninvaded treatments (Figure 4.2a and d). As expected, trees, such as *A. dealbata* were mostly associated with invaded and cleared treatments (Figure 4.2 b), whereas *Leucosiden sericea* and *Biden pilosa* dominated uninvaded treatments (Figure 4.3c). Grasses and forbs that commonly appear in the uninvaded treatments were also present in the cleared treatments (Figure 4.2c and d). Eigenvalues for the first two axes accounted for a total of 8% for all species, 51% for trees and shrubs, 20% for forbs, and 33% for grasses.

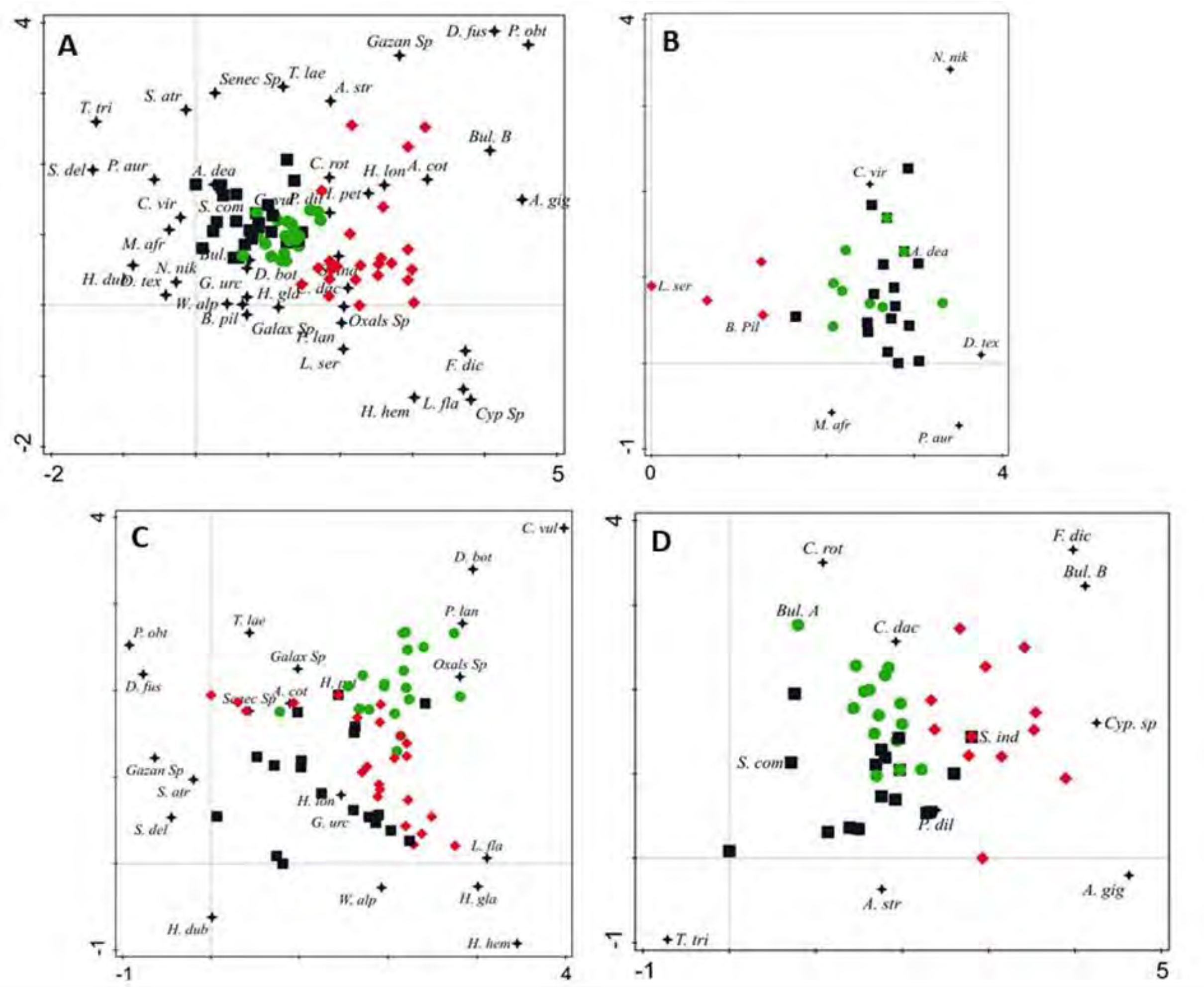

Figure 4.2. Correspondence Analysis (CA) bi-plots of species from different treatments (cleared (green dots)), invaded (black squares), uninvaded (red diamond)) for (a) all plants, (b) trees and shrubs, (c) forbs, and (d) grasses. The first four letters represent the full names of the species as listed in Appendix 1.

4.4. Discussion

The effects of IAP invasion and control have been extensively studied in South Africa and several of those studies indicated that IAP significantly alter the species richness, plant cover, abundance, and composition of native species (Ruwanza et al., 2013a; Tererai et al., 2013, Kerr and Ruwanza, 2016; Yapi et al., 2018). Given their unique competitive traits, such as the ability to uptake water and nutrients from deep soils (Richardson et al., 2007), Australian *Acacias* have the capacity to alter the ecological functioning and structure of native ecosystems (Marchente et al., 2003; Holmes et al., 2005; Le Maitre et al., 2011; van Wilgen, 2012). Results from this study have shown that *A. dealbata* invasion negatively affects plant diversity and cover, however, clearing of the species facilitates recovery although the recovery trajectory is hindered by regeneration of *A. dealbata* seedlings.

4.4.1 Effects of invasion and clearing on vegetation diversity and cover

Results from this study concur with the findings of previous studies by Blanchard and Holmes (2008), Ndou and Ruwanza (2016) and Kerr and Ruwanza (2016) which indicated that the clearing of the IAP does affect the structure and diversity of other native species. Ndou and Ruwanza (2016) reported that vegetation diversity and cover was more in sites that were cleared of *A. longifolia* more than 10 years ago than sites that were cleared less than three years ago. Kerr and Ruwanza (2016) reported a positive vegetation recovery trajectory in sites that were cleared of *E. grandis* in the Eastern Cape Province of South Africa. Van Wilgen et al. (2010) states that the ecological integrity of grasslands is compromised by invasion by IAP, thus negatively impacting on their species composition, cover, and functions. This is largely because many IAP, including *A. dealbata,* transform natural ecosystems and the quality of the resources of the ecosystems (Holmes et al., 2005). In this study, total species richness and diversity for grasses and forbs were low in invaded treatments as compared to uninvaded treatments. Results showed that, vegetation diversity and species richness in the cleared treatments were similar to the one in the invaded treatments, however, these cleared treatments had higher species richness than the uninvaded treatments. This concurs with findings by Ruwanza et al. (2013a); due to re-invasion by secondary invaders, mainly grasses and alien herbaceous vegetation, species richness and diversity may be similar in cleared and uninvaded areas. Vegetation in the invaded treatments was dominated by large *A. dealbata* trees which formed closed canopy, but some shrubs and understory grasses, forbs and herbaceous plants were present. The presence of the understory vegetation contributed to the high species richness in the invaded treatments (Ruwanza, 2020). Furthermore, recruitment and establishment of these native understory grasses and herbaceous species suggest that *A. dealbata* invasion does not result in total removal of native species (Ruwanza, 2020). Wang et al. (2011) showed that native species that can compete for resources such as water and soil nutrients are at a better advantage of recruiting underneath the invading species. However, the increased canopy cover by *A. dealbata* hinders light penetration to the native vegetation, thus reducing the cover of the grasses and herbaceous species (Tererai et al., 2013; Kerr and Ruwanza, 2016). According to Yapi et al. (2018), IAP alter soil nutrients to benefit the invading plant, making them outcompete herbaceous species and grass cover. Observations in this study indicate that richness for grasses and forbs was low in the invaded treatments. These observations are supported by Reynolds and Cooper (2010) who argued that canopy shading is related to the decreased grasses and forbs richness in invaded treatments.

The uninvaded treatments were dominated by native grasses such as *S. indicus, P. dilatatum* and *C. dactylon* as well as forbs such as *Galax urceolata.* Dominant native species are argued to have some similar traits to invasive species and therefore may have similar effects as invasive species on species that are less competitive (Davis et al., 2010; Tererai et al., 2013). Furthermore, overtime, the dominant and competitive grass species and forbs out compete other species, thus leading to a decline in species richness and diversity (Richardson et al., 2007; Tererai et al., 2013). This argument supports the findings from this study which indicated that the species richness and diversity was lower in uninvaded treatments due to the dominance of few native species.

4.4.2. Vegetation recovery in cleared sites

The empty niche hypothesis postulates that IAP take advantage of empty niches (in this case, recently cleared areas) and can establish, persist, and invade these areas by accessing and utilizing resources that are not consumed by native species (Elton, 1958; Morris et al., 2011). This hypothesis supports some of the findings of this study, where *A. dealbata* seedlings continued to persist in the cleared treatments. A key factor leading to this re-invasion is the ability of the invading species to better acquire limiting resources and their ability to utilize these resources more efficiently than native species, particularly in cleared treatments (Funk and Vitousek, 2007; Morris et al., 2011). There was vegetation recovery on the cleared treatments although it was a mixture of native and recruiting alien plants. Richardson et al. (2000), Ruwanza et al. (2013b), Ndou and Ruwanza (2016), and Nsikani et al. (2019) reported the growth of secondary invaders, mostly alien herbaceous plants, and grasses in the cleared treatments. Ruwanza et al. (2013a) and Nsikani et al. (2019) found that the clearing of IAP does not always yield favorable results of promoting native vegetation recovery, however, it may facilitate the proliferation of secondary alien species. Although the proliferation of these secondary invaders has been studied in the fynbos biome (Galatowitsch and Richardson, 2005; Nsikani et al., 2019), dominance of these secondary invaders has hardly been reported in the grassland communities.

Globally, *Acacias* are amongst the most crucial prolific IAP (Morris et al., 2011). As such, disturbed, low-competition, and nutrient-depleted ecosystems are more vulnerable to invasion by these IAP (Kull et al., 2011). This could be used to support the findings of this study which indicated the re-invasion of the cleared treatments by the same *A. dealbata* and by other herbaceous plants such *Dysphania botrys* and *Helichrysum petiolare.* However, results of this study cannot necessarily be used to infer causation given that monitoring prior to *A. dealbata* invasion was hardly done, nevertheless, results of this study draw on a study by Marchante et al. (2011) and Nsikani et al. (2019) which argue that the presence of viable soil seed banks of the cleared IAP may facilitate the regrowth of the cleared plant. In their study, Pretorius et al. (2008), also revealed that after years of clearing, seedlings of *Acacia* species persisted on the cleared site even though native species were sown; this emphasizes that seed banks of invasive *Acacias* are long-lived. After clearing, the cleared material is usually left on site before burning (in cases where burning is conducted), and these large stumps left standing on site (Fell and Stack) have potential to be a source of seeds for the re-invasion (Le Maitre et al., 2011). Therefore, for clearing to yield positive vegetation recovery, it is advised to adopt the Fell and Remove method because it facilitates the removal of alien plant seeds after clearing thus increasing chances of native vegetation to colonize (Le Maitre et al., 2011). Fell and remove will create space for active restoration measures such as seeding to be done, as well as reduce the amount of N that is fixed into the soil by Acacia litter (Nsikani et al., 2019).

Furthermore, the competitive traits of recruiting *A. dealbata* seedlings and other secondary invaders allow these to outcompete other native species for accessing the excess and unused soil nutrients left after the clearing of the IAP (Morris et al., 2011; Kerr and Ruwanza, 2016; Nsikani et al., 2019). *Acacia* species are referred to as transformer species because they change soil properties to conditions that promote the establishment of secondary alien species and some pioneer indigenous species (Holmes et al., 2005). After clearing, the soil is left with unused nutrients, therefore fast-growing plants take this advantage to quickly inhabit the area and use the excess resources (Yelenik et al., 2004; Kerr and Ruwanza, 2016; Ndou and Ruwanza, 2016). Moreover, *Acacia* species are N-fixing meaning they enrich the soil, therefore creating a conducive environment for alien herbaceous plants to dominate the cleared area (Marchante et al., 2001; Corbin and D'Antonio, 2004; Ruwanza et al., 2013a). The fluctuating resource hypothesis by Davis et al. (2000) states that when resources fluctuate and become temporarily available due to excess inputs or reduced use, invasion by IAP is accelerated in such ecosystems, in this case, invasion by recruiting *A. dealbata* seedlings. Marchante et al. (2011) and Fill et al. (2017) echoed this argument stating that after removal of invasive *Acacia* species, it takes several years for soil nutrients to recover to pre-invasion conditions. Ruwanza et al. (2013a), however, argues that the recruiting of the herbaceous plants in the cleared treatments have the possibility of hindering the establishment of native vegetation. Indeed, a study by Ruwanza et al. (2013a) showed that other graminoids recruited in the cleared sites, therefore native plants could not compete for soil nutrients. The recruitment of these plants, although they were mostly in the juvenile stage, could be the reason the cleared treatments had higher species richness.

Moreover, IAP often thrive in areas that do not have their natural suite of plant feeding insects and pathogens, this could also explain why *A. dealbata* seedlings continue to grow in the cleared treatments (Zimmermann et al., 2004). The absence of these pressures, adapting to new conditions, and the ability to produce many seeds, means that IAP can out-compete native vegetation once they have recruited in cleared sites (Zimmermann et al., 2004). Re-invasion by *A. dealbata* and secondary invasion by other alien forbs and grasses hinders the recovery of native vegetation (Ruwanza et al., 2013a; Kerr and Ruwanza, 2016). Huchzermeyer et al. (2018b) stated that rapid transformation from an invaded state with canopy cover to a completely cleared state with full exposure from sunlight is not favorable for the establishment of grasses, particularly in these grasslands.

After the clearing of the *A dealbata* in the study sites, there was little follow-up treatment and/or active restoration measures in place by the WfW programme. Therefore, considering the persistent nature of *A. dealbata* seed banks, this lack of frequent and timeous follow-up and active restoration measures could be the reason why *A. dealbata* is re-growing faster in the cleared treatments (Fourie, 2008; Kerr and Ruwanza, 2016). Holmes et al. (2005) state that IAP thrive in disturbed areas, and they alter the establishment conditions. This often leads to unsuitable germination and recruitment conditions for native vegetation as was observed in this study which showed little native vegetation recovery in the cleared sites (Holmes et al., 2005). In fact, following invasion and clearing of an invasive plant, conditions of the cleared area are often not optimal for the establishments of many species, as such recovery of native vegetation is such areas may have to be facilitated by active restoration (Holmes et al., 2005).

On the contrary, the presence of some native vegetation in the cleared treatments is a good thing, as these few recovering native species in cleared treatments have the potential to act as centers of vegetation recovery. Similar findings of low native vegetation recovery in areas cleared of AIP were recorded by Blanchard and Holmes (2008), while Reinecke et al. (2008) and Fill et al. (2017) in their studies recorded higher recovery of native vegetation. This native vegetation recovery can be influenced by, but not limited to, improved soil nutrient recovery after clearing, native soil seed banks, and seed germination (Ruwanza et al., 2013a; Kerr and Ruwanza, 2016). This means that if there is abundance of native seed banks and propagules from surrounding ecosystems, recovery of native vegetation can be achieved (Ruwanza et al., 2013; Ndou and Ruwanza, 2016). Furthermore, the presence of remnant indigenous vegetation promotes seeds dispersal and subsequently, the colonization and establishment by native species (Ruwanza et al., 2013a). The native plants that were present in the cleared site were grasses of *S. indicus* and *P. dilatatum*, forbs of *D. botrys* and *H. pertiolare*, and the shrub species *Myrsine Africana*. These species were present/dominant in the adjacent uninvaded treatments; therefore, it is safe to argue that these uninvaded treatments were sources of seed dispersal to the cleared treatments (Gauriguata and Ostertag, 2001; Ruwanza et al, 2013a). In future, the above-mentioned recruiting native species in cleared treatments have potential to serve as recovery patches (Gauriguata and Ostertag, 2001; Ruwanza et al, 2013a). The low number of native vegetation recruitment can be attributed to seed dormancy (Holmes et al., 2005; Kerr and Ruwanza, 2016). This positive response of the indigenous species gives hope that given enough time and necessary intervention methods, areas cleared of IAP will move towards the desired states, similar to undisturbed communities (Fill et al., 2017). According to Pretorious et al. (2008), planting a mixture of native seeds after clearing improves biodiversity and abundance of the native species. In fact, findings from their study indicated that sowing treatments suppress the recruitment of *A. mearnsii* seeds over time (Pretorious et al., 2008).

4.5. Conclusion and restoration implications

This study echoes what is eloquently stated by Wittenberg and Cock (2005) and Ruwanza et al. (2013a); that the mere clearance of an IAP (passive restoration) is not sufficient to promote spontaneous recovery of native vegetation. The persistence of *A. dealbata* in the cleared treatments is a consequence of a range of factors including the legacy effect, resource alteration, secondary invasion, and the competitive traits of the invading species (Marchante et al., 2011; Ruwanza et al., 2013a). The persistence of *A. dealbata* and invasion by other alien vegetation, if not controlled on time, will lead to the accelerated degradation of cleared sites in the Tsitsa catchment and this will continue to pose a huge threat to the adaptive capacity of people and ecosystems (Fabricus et al., 2016). Therefore, to avoid and mitigate such threats, there needs to be an investment in projects that aim at increasing the indigenous vegetation cover, and enhance soil organic contents (Fabricus et al., 2016). Kimiti et al. (2017) and Fill et al. (2017) advise that this can be done by implementing active restoration techniques that will reduce revegetation by secondary invaders and to some extent the amount of soil loss after clearing and promote the creation of microsites that promote the establishment and growth of native plants. Drawing from studies that showed a positive response of some native vegetation recovery, it can be argued that with the right intervention methods (e.g., native species seeding), continuous and timeous monitoring, and enough time to facilitate soil recovery, ecosystems stand a chance to recover to desired states after a disturbance, i.e., after clearing of IAP (Pretorius et al., 2008; Fill et al., 2017).

Restoration ecology, particularly active seeding, provides pathways of how this ‘new’ or desired state can be achieved (Dobson et al., 1997). Cairns (1989), in his model of potential management pathways for the restoration of disturbed areas states that, from the current state, an ecosystem may stay stable or further degradation may occur. To prevent the latter, active recovery efforts must be implemented and facilitate vegetation recovery (Cairns, 1989; Kimiti et al., 2017). In this context, restoration is defined as the “re-establishment of the structure, functions, and natural diversity of an area that has been altered from its natural state” (Pess et al., 2003). Lastly, more robust research on the clearing methods should be done to assess which methods or mixture of methods can completely remove an IAP as well as suppress re-invasion by the same species or by secondary invaders (Blanchard and Holmes, 2008; Blanchard et al., 2008; Pretorius et al., 2008). Huchzermeyer et al. (2018b) provided a list of clearing methods that can be used to achieve a gradual increase in the cover of native grasses, namely the strip clearing method, bark stripping method, and leaving standing stumps of the invasive plant.

More research is needed to examine the restoration efficacy of the above-mentioned methods.

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Chapter 5: Synthesis, recommendations, and conclusions

5.1. Introduction

Invasive alien plants (IAP) are the second greatest threat, after human impacts, to the environment and biodiversity (Peichar and Mooney, 2010). Therefore, controlling IAP has been widely implemented across the globe to minimize the impacts. In South Africa, the national clearing programme, namely Working for Water (WfW) adopted a passive restoration approach under the premise that after clearing of the IAP, ecosystems will spontaneously recover to their natural state (Le Miatre et al., 2020). However, it has not been successful in limiting the rate of invasion and promoting soil and indigenous species recovery after clearing and the impacts persist long after the plant has been removed (Marchante et al., 2009; Ruwanza et al., 2013a; Nsikani et al., 2019). Considering the limitations of the WfW alien plant clearing programme, it is vital to implement measures that will facilitate recovery of the cleared areas to pristine conditions. To achieve that, there needs to be an intimate understanding of below and above-ground processes after clearing and an assessment of the responses in vegetation diversity and composition after the removal of the invasive plant. This study sought to bridge that gap by investigating, 1) how do soil properties change after the clearing of *A. dealbata*, and 2) how does vegetation diversity and composition change after clearing *A. dealbata.* To do this, I assessed recovery trajectories of soil properties and native vegetation after clearing of *A. dealbata* as this will help to quantify the impacts and benefits of *A. dealbata* clearing.

5.2. Summary of results

Objective 1: To assess changes in soil physical and chemical properties following *A. dealbata* clearing.

Literature revealed that IAP persist after clearing because of soil legacy effect, which is the long-lasting changes in soil properties and is prominent in areas that have been invaded for a long period (Marchante et al., 2009; Nsikani et al., 2017). Invasive alien plants also have persistent soil seed banks that re-sprout after clearing, therefore making them flourish after being removed (Le Maitre et al., 2011). Several studies have shown that IAP thrive because of their ability to alter the quality and quantity of resources such as soil nutrients, sun light, and water. For example, Yelenik et al. (2004) state that *Acacias* inject large quantities of N to the soil, changing the N-cycling processes and that is detrimental to the survival of indigenous species. They also suppress the survival of soil seed banks of native species; this leads to local extinction of native species.

This study showed that the effects of *A. dealbata* invasion and clearing on soil properties vary and can be long-lasting even after the species has been cleared. This is because some soil properties showed no significant differences between the cleared and the invaded treatments. For example, soil pH remained the same in the cleared treatments as in the invaded treatments, and similar results were observed in previous studies (Lorenzo et al., 2010; Lazzaro et al., 2014; Kerr and Ruwanza, 2016; Gwate et al., 2021). This emphasizes the persistence of altered soil properties years following the removal of IAP and several studies linked these findings to soil legacy effect (Marchante et al., 2009; van Der Waal, 2009; Nsikani et al., 2017). Regardless of this persistence in soil acidity after clearing, Nsikani et al. (2017) shed some hope for recovery of soil pH because they reported trends of moving towards pH levels that are like the levels of the uninvaded sites. This study reported no significant differences in soil chemical properties such as soil P, C, N, Ca, Mg, and Na, and these concur with previous studies (Nzila et al., 2004; Lazzaro et al., 2014; Kerr and Ruwanza, 2016; and Ndou and Ruwanza, 2016; Gwate et al. 2021). Since these soil properties did not show significant differences, restoration approaches should invest more on soil properties such as, resistivity, soil moisture, and infiltration rates since these are highly impacted. However, that does not mean other properties should not be actively restored at all. Cleared treatments had less litter fall compared to the invaded treatments because *A. dealbata* deposits large amounts of litter, thus the invaded sites had more soil N that they derived from the *Acacia* leaves (Corbin and D'Antonio, 2004, Marchante et al., 2009; Lazzaro et al., 2014). Moreover, soils underneath *Acacias* have faster decomposition rates than soils underneath native species and that feeds on the C and N levels in the invaded areas (Ehrenfeld, 2003; Lazzaro et al., 2014, Kerr and Ruwanza, 2016). Results showed that the soils in the area were mostly non-repellent and that could be associated with the observed soil moisture in the area. The non-repellent soils could also have led to the increased infiltration capacity of soils in the cleared treatments (Doerr et al., 2000). Therefore, the study concluded that with soil becoming less repellent and with more infiltration, that is an indication of a positive trend towards recovery of soil physical properties to conditions similar to uninvaded treatments after clearing.

Objective 2: To assess changes in native vegetation diversity and composition following the clearing of *A. dealbata.*

Consistent with findings by Kerr and Ruwanza (2016) and Ndou and Ruwanza (2016), this study found that clearing of *A. dealbata* triggers changes in vegetation diversity and composition. Indeed, this study reported the persistence of *Acacia* seedlings in the cleared treatments. These results were consistent with the findings by Pretorius (2008). The persistence of *Acacia* seedlings is largely because *Acacias* are prolific and that their morphological characteristics allow them the advantage to better access limiting resources and utilising these resources more effectively than native species (Funk and Vitousek, 2007; Morris et al., 2011). There are usually excess nutrients in the soil after clearing IAP which fast-growing secondary invaders take advantage of and quickly recruit in the cleared area (Yelenik et al., 2004; Kerr and Ruwanza, 2016; Ndou and Ruwanza, 2016). The empty niche hypothesis emphasizes that IAP will colonize successfully in "empty" niches, which means that they would recruit in the cleared areas and utilize resources that are not used by recruiting native species (Elton, 1958; Morris et al., 2011). The occurrence of the *Acacia* seedlings can be also attributed to their ability to produce viable soil seedbanks which allow them to recruit after clearing, especially when there is little or no follow up treatment and no active seeding of native species (Marchante et al., 2011; Nsikani et al., 2019).

Despite the persistence of *Acacia* seedlings and other alien herbaceous species (secondary invaders) in the cleared areas, this study recorded recruitment of some native species in the cleared sites which is an indication of a positive recovery trajectory, and this result is consistent with results reported by Blanchard and Holmes (2008); Reinecke et al. (2008) and Fill et al. (2017). Native species that recruited in the cleared treatments were species that dominated the adjacent uninvaded treatments. This means that the adjacent uninvaded treatments could potentially act as recovery sources/nuclei and provide the cleared treatments with some native seeds through seed dispersal (Gauriguata and Ostertag, 2001; Ruwanza et al, 2013a).

5.3. Conclusion and recommendations

This study concludes that the clearing of *A. dealbata* is showing successes in terms of reducing cover by the alien plant. However, recovery of soil properties and native plant species is still very limited, and this is due to several reasons such as lack of native soil seed banks, varied changes in soil properties, insufficient follow-up control treatments which results in *Acacia* reinvasion, and proliferation of secondary invaders. The recovery of some soil physico-chemical properties observed from this study may have provided a base support for native vegetation species to recover in the cleared treatments. However, for complete or successful ecosystem recovery, there must be sufficient supply of native soil seed banks and propagules that would aid the recruitment of native species. The study further noted that the adjacent uninvaded treatments have the capacity to act as recovery patches and provide the needed sources of native seeds that can germinate in the cleared treatments (Blanchard and Holmes, 2008; Ruwanza et al., 2013b). To strengthen the potential of the soil and vegetation recovery in cleared sites, an effective clearing method should be used to ensure that there is a reduced soil damage from the clearing process (considering that heavy machinery during clearing induces soil water repellency). In this study, mechanical clearing and fell and stack methods were used to clear *A. dealbata*. Blanchard and Holmes (2008) proposed that clearing programmes adopt the Fell and Remove clearing method because removing the IAP stumps from the sites will suppress their seeds and prevent re-invasion, thus granting native species an opportunity to recover. Blanchard and Holmes (2008) further added that thinning should be implemented in certain areas as this will allow native species to recruit between the stacks. Ruwanza et al. (2013b) suggested that a four-stage thinning process is the most effective in facilitating the recovery of ecosystem structure and composition. Although this method has mostly been done in riparian systems, it can still be implemented in the grasslands too. Clearing alone has been proven to not yield successful ecosystems recovery, so these clearing methods must be aided with long-term monitoring and continuous active restoration measures (Holmes et al., 2020). Further studies must be done to provide an understanding of how IAP impact on soil microbial communities and how that plays a role in the transformation of soil properties and subsequently on vegetation. There should be investment on long-term ecosystem recovery after IAP clearing and an appreciation of active restoration ecology techniques as tools to achieve this because it provides ecosystem-based solutions. Studies should look at how other factors such as land use and climate change are influencing the rate of recovery of ecosystems after clearing invasive alien plants. This study recommends the following:

1. The clearing of IAP should be supplemented by measures to recover soils and these measures can include soil manipulation measures (e.g., N reduction techniques) aimed to reducing soil nutrient legacy effect caused by the invading *A. dealbata.* After four years of clearing, N in the cleared sites is still similar to those in invaded sites. This means active measure need to be implemented to alter N concentration to the levels that are suitable for native grasses growth.
2. Clearing should be followed by a detailed soil and vegetation restoration initiative that is effective. Given the cost implications associated with active restoration, simple techniques such as re-seeding and brush packing can be implemented on the cleared sites. However, future studies need to test the efficacy of these simple active restoration techniques.
3. The observed *Acacia* re-invasion and the proliferation of herbaceous cover seem to suggest that the WfW programme needs to come up with an effective alien clearing

follow-up programme. It is high time WfW develops a timeous and effective follow-up programme if IAP clearing is to yield effective results.

4. Lastly, although clearing using the fell and stack burning method does reduce alien plant biomass, WfW should start considering use of an effective clearing methods that also promote soil and native vegetation recovery. Several methods e.g., the fell and removal and thinning (selective logging) have been suggested, however, more research needs to be done to investigate their efficacy. These methods need to be examined and potentially used in future IAP clearing programmes.

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Appendix 1. List of 75 frequently occurring plant species present in three clearing treatments. (*) indicates that the species was present and is based on calculated species occupancy frequencies categorized as (1-25%), **(26 – 50%), *** (51 – 75%) and **** (76 – 100%). (-) indicates that the species was no present in that treatment.

Species name	Cleared	Invaded	Uninvaded
Trees and shrubs			
Acacia dealbata	****	****	-
Bidens pilosa	**	*	*
Cheilanthens viridis	**	***	*
Diospyros texana	**	**	-
Leucosidea sericea	-	*	*
Myrsine africana	*	**	-
Nemosenecio nikoensis	*	*	-
Plumbago auriculata	-	**	-
Rhus longipes	-	-	*
Species AA	-	*	-
Species AB	*	*	-
Species AC	*	-	-
Species AE	*	*	-
Species AG	-	-	*
Species C	-	*	-
Species K	-	*	-
Species M	-	*	-
Species N	*	*	-
Species W	-	**	-
Species Z	-	*	-
Forbs			
Anthemis cotula	*	-	*
Cirsium vulgare	*	-	-
Dactylorhiza fuschii	-	-	*
Dysphania botrys	*	-	-
Galactia sp	-	-	*
Galax sp	*	*	-
Galax urceolata	-	***	**
Gazania sp	-	*	*
Helianthus longifolius	**	**	***
Helichrysum petiolare	****	**	****
Heteranthea dubia	-	*	-
Hypochaeris glabra	*	**	*
Hypoxis hemerocallidea	-	-	*
Lobelia flaccida	-	-	*
Oxalis sp	***	*	**
Plantago lancealata	*	*	*

Pseudognaphalium obtusifolium	-	-	*
Senecio atratus	*	*	-
Senecio deltoideus	-	-	-
Senecio sp	*	*	*
Taraxacum laevigatum	*	*	*
Woodsia alpina	-	*	*
Species A	-	*	-
Species AD	*	-	-
Species AH	-	-	*
Species B	****	***	****
Species D	*	*	-
Species E	*	-	*
Species F	*	-	*
Species G	*	-	-
Species H	-	*	-
Species I	*	-	*
Species J	*	-	-
Species L	*	*	*
Species O	-	-	*
Species R	-	-	*
Species S	-	*	*
Species U	-	*	-
Species V	-	*	-
Species X	-	*	-
Species Y	-	*	*
Species T	-	-	*
Grasses			
Sporobolus indicus			
Themeda triandra	****	****	****
Cynodon dactylon	-	*	-
Paspalum dilatatum	***	*	**
Sporobolus compositus	***	***	****
Aristida stricta	***	***	*
Cyperus rotundus	*	**	*
Bulbostylis sp	*	-	*
Cyperus sp	****	*	*
Fimbristylis dicotoma	-	-	**
Agrostis gigantea	-	-	*
Bulbostylis B	-	-	*
Species P	-	-	*
	-	-	*