The effects of *Chromolaena odorata* on tree growth dynamics at Buffelsdraai Landfill site

by

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Preface

The research work contained in this thesis was completed by the candidate in the School of Life Sciences, College of Agriculture, Engineering and Science, University of KwaZulu-Natal, Pietermaritzburg Campus, South Africa, under the supervision of Dr Zivanai Tsvuura and co-supervised by Dr Hloniphani Moyo. The research was financially supported by the Durban Research Action Partnership (D'RAP) and the National Research Foundation (NRF).

The contents of this work have not been submitted in any form to another university and, except where the work of others is acknowledged in the text, the results reported are due to investigations by the candidate. The chapters of this thesis are written as standalone chapters. The thesis is formatted according to South African Journal of Botany.

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Declaration 1: Plagiarism

I, < Rerani Ramaano>, declare that:

(i) The research reported in this thesis, except where otherwise indicated or acknowledged, is my original research.

(ii) This thesis has not been submitted in full or in part for any degree or examination to any other university;

(iii) This thesis does not contain other persons' data, pictures, graphs or other information, unless specifically acknowledged as being sourced from other persons;

(iv) This dissertation does not contain other persons' writing, unless specifically acknowledged as being sourced from other researchers. Where other written sources have been quoted, then:

a) Their words have been re-written but the general information attributed to them has been referenced;

b) Where their exact words have been used, their writing has been placed inside quotation marks, and referenced;

(v) Where I have used material for which publications followed, I have indicated in detail my role in the work;

(vii) This thesis does not contain text, graphics or tables copied and pasted from the Internet, unless specifically acknowledged, and the source being detailed in the dissertation and in the References sections. There are some repetition in the study area description of chapters 2 and 3 which was inevitable.

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Dedication

I would like to dedicate this work to my late father Mmbofheni Errol Ramaano who never lived to witness my success. To my late Grandmothers Mundzedzi Mudzuli Ramaano and Thinawanga Anna Mutengwe.

Abstract

Ecological restoration is a process which assists the recovery of an ecosystem which was previously disturbed or degraded. Through continuous disturbances, invasive alien plants (IAPs) are able to successfully spread and establish themselves while reducing the diversity and the abundance of native plants. The IAPs invade both human and non-human modified landscapes thus causing huge threats to biodiversity conservation and ecosystem services provision. The invasion of alien plants at Buffelsdraai Landfill Site Community Reforestation Programme is a major threat to reforestation success. Buffelsdraai is under rehabilitation through a community reforestation programme that focuses on active involvement of nearby communities in propagating seedlings and planting them on site. The study was conducted in a former sugarcane field which has been planted with diverse native tree species, but invaded by dense stands of *Chromolaena odorata*. This study aimed to 1) determine the influence of a woody shrub *C. odorata* on the growth dynamics of replanted native trees at the Buffelsdraai Landfill Site, and 2) to assess the efficacy of cutting height and frequency in mechanical control of *C. odorata*.

Over a period of 12 months, the growth responses of three native tree species (i.e., *Vachellia natalitia*, *Brachylaena discolor* and *Erythrina lysistemon*) were investigated by measuring stem length and diameter, trunk basal diameter, tree height and canopy diameter, under four treatments. The treatments consisted of: (i) a control, where no clearing of *C. odorata* or grasses was undertaken beneath the canopy of trees; (ii) clearing of *C. odorata* from underneath trees; (iii) clearing of grass from underneath trees; and (iv) clearing of both grass and *C. odorata* from trees. Measurements were undertaken on 264 trees, 88 individual trees per species, 22 individuals per species per treatment at the beginning and the end of the study. Stem, trunk and canopy length, diameter, and tree height growth data within each tree species were analysed using generalized linear models. Tree height growth comparisons among the tree species was analyzed using a non-parametric Kruskal-Wallis test since the data did not meet the assumptions of a One-way ANOVA. There were no significant differences between all four treatments across the tree species ($P > 0.05$). However, *E. lysistemon* growing where there was grass removal had significantly higher ($P = 0.033$) trunk basal diameter growth, than other treatments. Results from this study showed that the removal of *C. odorata* at the Buffelsdraai community project had no impact on the trees due to short time period of the

study. Overtime we expect that tree performamce would increase with less *C. odorata* competition. Trees performed equally well in the presence and absence of grass and *C. odorata*. Young tree saplings of pioneer species were used for this study which could be a possible influence of these results.

To investigate the impact of *C. odorata* mechanical control, 28 plots (5 m × 5 m) were established, at least 2 m apart, with one designated as the control with no cutting of the plants. The two treatment combinations were between cutting frequencies which had: (1) cutting frequency made up of single, twice and thrice cutting; and (2) stem cutting height which had 30, 100 and 200 mm cutting heights aboveground. I used 420 *C. odorata* plants (i.e. 3 cutting heights × 9 replicate plots × 15 plants + 15 control plants). Plants were first cut in November 2016, the coppice regrowth was cut on plants in 18 plots in February 2017, and a third cut was made on plants in the last nine plots in May 2017. Stump basal diameter, the number, length and diameter of resprouting shoots were measured at 3-month intervals. Mean number of resprouting shoots, mean shoot length and diameter of the resprouts, mean shoot length:diameter ratio (shoot taper function) of resprouts, shoot production (shoots mm^{-1}) and mean total shoot basal area:stump basal area ratio for all treatment combinations were analysed using the Friedman's test. The relationship between cut stem diameter and the number of resprouting shoots, resprouting shoot length, and resprouting shoot diameter were explored using correlation analysis. There was a significant difference in the number of resprouts produced across all treatment combinations ($P < 0.001$). The 30 mm cutting height produced the least number of resprouts with increasing cutting cycles compared to the 100 and 200 mm cutting height. Shoot length and diameter decreased with cutting frequency, with smaller shoots produced with continued cutting cycle. There were significant difference in the shoot length and diameter of shoots produced in the 30 mm cutting height in the last cutting cycle and those that were produced on 200 mm cutting height. Shoot taper function and shoot production showed significance ($P < 0.001$), with smaller number of shoots produced in the 30 mm cutting height over the cutting frequencies ($P < 0.001$). Total shoot basal area:stump basal area ratio did not differ significantly across the cutting heights and the cutting frequencies. There was a strong positive relationship ($r = 0.91$; $P < 0.001$) between the number of resprouts and stem diameter across all repeatedly cut stems in all cutting heights and the number of resprouts produced in each cutting height decreased over each cutting frequency. Repeated cutting of *C. odorata* at a lower cutting height of 30 mm may deplete its energy reserves, reducing the number of resprouts produced, thereby leading to death of the plants. Repeated cuttings at short-term intervals also prevents the plants from growing to reproductive maturity and seed production, leading to no seed dispersal to increase infestations.

Chapter One: Introduction

1.1. Background

Ecological restoration is the process of facilitating the recovery of an ecosystem which was previously disturbed, destroyed, or degraded as a result of human activities at most (Berger, 1993; Funk et al., 2008; Benayas et al., 2009). Ecological restoration is important as it creates an ecosystem that is self-sustaining or works on its own and is able to tolerate different forms of perturbation (Berger, 1993). Restoration practitioners aim at reproducing high levels of species diversity, species composition and functional group diversities in impacted areas (D'Antonio and Meyerson, 2002; Martin et al., 2005). However, restoration success may be constrained by factors such as invasive alien plants that supress and influence species composition and diversity of native plants in the ecosystem being restored (Alexander et al., 2011; Ciccarese et al., 2012). In this thesis, the growth of native trees planted in a community reforestation initiative that sought to rehabilitate the buffer zone of Buffelsdraai landfill site was investigates. The site was a sugarcane field, but now infested with invasive alien plants, with *Chromolaena odorata* being the most dominant invader. I also investigated the effectiveness of subjecting *C. odorata* to different cutting frequencies at different heights on its resprouting ability.

1.2. Restoration and rehabilitation of ecosystems

1.2.1. Restoration and rehabilitation definitions

The terms restoration and rehabilitation differ from each other by their end result as well as their processes which range amongst the degrees in which they are being applied to in terms of species composition, structure and the function of a historical ecosystem (Stanturf et al., 2014). Restoration puts more of its emphasis on the historical fidelity and also on the recovery of native species composition while rehabilitation on the other hand focuses on the functional side of the recovery and it can include species out of their native range (Chazdon et al., 2016). Restoration can conform to a certain species, it can focus on community composition or can also concentrate on the entire ecosystem or landscapes and its goals maybe be established through ecosystem services (Ehrenfeld, 2003). Rehabilitation is oriented around restoring a desired species composition, processes or structure of an already existing but degraded ecosystem (Stanturf et al., 2014). In addition, rehabilitation aims at increasing the commercial value of timber, improving biological diversity, improving soil fertility and also increasing forest functioning (Stanturf et al., 2014). The terms restoration and rehabilitation are therefore used in exchange in this thesis as this project focuses on both processes.

1.3. Ecological restoration

One of the important goals of restoration ecology is to re-establish ecosystem functioning and enrichment of biodiversity (Benayas et al., 2009), which together enhance the provision of ecosystem services (Bullock et al., 2011). Provision of ecosystem services (regulating and provisioning) and biodiversity enhancement have a positive relationship and most ecosystem services have a positive correlation with biodiversity, therefore restoring one enhances the other (Bullock et al., 2011). A lot of attention has been focused on the value of the ecosystem, the services obtainable thereof and how human beings benefit from those services (Benayas et al., 2009). For example, ecosystem services that are critical to human well-being include water quantity and quality, carbon storage, regulation of climate and maintenance of soil fertility. There is a perception that restoration ecology could assist in the provision of more ecosystem services that are critical for human well-being (Benayas et al., 2009).

Worldwide, it is estimated that about two billion hectares of tropical forests are exposed to degradation (Stanturf et al., 2014). However, scientists are starting to put more effort in restoring degraded ecosystems and ensuring that these ecosystems recover and regain functioning. A number of restoration initiatives aim at increasing plant and animal biodiversity (Benayas et al., 2009). Restoration is guided by the values of the society that is motivated or conflicted by the uncertainty of goals that aim at sustainability (Stanturf et al., 2014). These include repairing or enhancing the functioning of ecosystems, enlarging the habitat for species of concern, and enhancing biodiversity (Stanturf et al., 2014). Some restoration ecologists still consider forest communities as tightly integrated biological systems (Stanturf et al., 2014).

The actions implemented to promote ecosystem restoration in areas that were human-impacted are not similar to the original ecosystem in terms of species composition and provision of ecosystem services (Chazdon, 2008). However, biodiversity restoration is able to restore ecosystem functioning and recover many of its components to the original condition

(Chazdon, 2008). The progress of restoring an already functioning ecosystem is fully dependent on the original state of the ecosystem, the level of degradation that has occurred and also the desired outcome of that particular ecosystem (Chazdon, 2008; Ruiz-Jaen and Aide, 2005). Approaches taken for restoration must consider the abundance, spatial distribution and the quality of the remnant vegetation, which indicate potential natural regeneration (Stanturf et al., 2014).

Different kinds of disturbances, such as the invasion by alien plants, human impacts and fires degrade the functioning of ecosystems (Funk et al., 2008). With increasing human impacts on forests, there is a decline in ecosystem services which may be reversed by reforestation (a process of replanting trees) so that the condition and status of degraded forests is improved (Chazdon, 2008). In landscapes that have degraded soils, rehabilitation is achieved by planting carefully-selected native tree species (Ruiz-Jaen and Aide, 2005). This ensures that soil fertility is improved and restored, accompanied by the enhancement of biodiversity (Ruiz-Jaen and Aide, 2005). The planting of seedlings of native species is also an important feature of forest regeneration which hastens recovery of the forest and increases species richness of trees and other plants (Ruiz-Jaen and Aide, 2005). When intermediate levels of degradation occur, reforestation using native trees and supported natural regeneration can increase, and improve ecosystem services and also provide income for poor communities (Chazdon, 2008).

Forest restoration in both developing and developed countries is carried out by local communities with the involvement of non-governmental organisations and the state through national programs (Chazdon, 2008). Local knowledge of the restoration process, establishment of different species of economic and ecological concern and the inclusion of different rehabilitation methods with local development programms are important components of restoration success (Chazdon, 2008). The effects of restoration on a particular ecosystem differs with spatial scale of restoration, symbiotic effects of biotic and abiotic stresses from factors such as invasion by alien species, changes in plant-animal relationships and climate change. A wide range of forest restoration has direct relevance in influencing the composition and the structure of future forests, fauna and landscapes (Chazdon, 2008).

1.4. Restoration and global climate change

The anticipated global climate change effects suggest that the need for forest restoration in the future will be much greater given the increasing intensity of invasion by flora and the amount of land invaded (Stanturf et al. 2014). Due to the anticipated increasing effects of global climate change, the change is rapid now than before and it is important to understand how human activities are influencing this change with regards to biodiversity and the functioning of the ecosystem (Tylianakis et al., 2008; Vilà et al., 2011). Alien plant invasion of terrestrial ecosystems is one of the drivers caused by this change given that some of these species successfully become abundant and influence biodiversity (Vilà et al., 2011). Alien invasive plants decrease native plant diversity and this is due to their competitive ability that may be enhanced by elevated carbon dioxide levels in the atmosphere (Bradley et al., 2009). In contrast, climate change does not always influence plant invasion but can also reduce the competitiveness of invasive plants given that the environmental conditions are not favourable (Bradley et al., 2009).

1.5. Invasive alien plants

Invasive alien plants (IAPs) are non-native plant species that have the ability to successfully spread and establish outside of their native range (Daehler, 2003). These plants may reduce the diversity and the abundance of native plants in an ecosystem, and they are capable of threatening the persistence of the native plants through resource competition (Reid et al., 2009). In addition, IAPs pose a huge threat to biodiversity conservation (Rejmánek et al., 2005; te Beest et al., 2012) and the provision of ecosystem services (van Wilgen et al., 2008), while also invading both human and non-human modified landscapes (te Beest et al., 2012). The IAPs are one of the major causes of plants extinction soon after changes in land use, and are regarded as one of the most important global threats to biodiversity, amongst climate change and human population growth (van Wilgen et al., 2008). Plant invasions are responsible for changing the structure and composition of plant communities and also disturb the stability of ecosystems while also threatening economic productivity of forests (Witkowski and Wilson, 2001).

Once IAPs invade an area, they cause accelerated increases in the soil nitrate pools which drastically increases seed germination (D’Antonio and Meyerson, 2002). As such, these weeds have persistent seed banks whose size is often much larger than those of native species (D’Antonio and Meyerson, 2002). Successful control of IAPs enables natural ecosystems to recover to their natural states and allows those ecosystems to function accordingly (Reid et al., 2009). In order for an ecosystem to recover from invasion by alien plants, management of IAPs is one of the most important tools that is available for use (Reid et al., 2009).

1.5.1. Invasive alien plants in South Africa

Natural ecosystems in South Africa as well as those from other parts of the world are vulnerable to threats from invasive plants (Nel et al., 2004). The scale of invasion by alien plants in South Africa is huge, and about 10 million hectares of land have already been invaded (Nel et al., 2004).

A large number of IAPs is already well-established across South Africa (e.g. *Chromolaena odorata*, *Lantana camara*, *Solanum* spp.), while there are some that are at early establishment stages (Witkowski and Wilson, 2001). In addition, a number of these species have just been introduced and are at a phase of rapid population growth (Nel et al. 2004). Problems arising from the proliferation of IAPS include biodiversity loss, reduced water supply, and native plants extinction, all of which are increasing at a rapid rate (Nel et al. 2004). A comparative approach between native and non-native plants species has often been used in studies that seek to understand the causes of the rapid and major success of these plants (Daehler, 2003). Approaches focus on pairing alien invasive with native plants to determine the success of an invader over the native plants (Daehler, 2003). Due to limited resources, it is important that most choices made on invasion management must be focussed on controlling IAPs.

1.5.2. Chromolaena odorata

Chromolaena odorata (L.) R.M King & H. Robinson (Asteraceae) is known for its large native and non-native distribution across the world (Witkowski and Wilson, 2001). In its native range, the species is largely distributed in the southern part of United States of America to the northern part of Argentina (Witkowski and Wilson, 2001). The species invaded the non-native range of Asia in the 19^{th} century (Zachariades et al., 1999; 2016). This plant has become one of the most problematic invasive plants in South Africa. Due to its rapid growth, fast regeneration and production of numerous seeds which are easily dispersed, *C. odorata* is difficult to control (Zachariades et al., 2016). *Chromolaena odorata* highly impacts disturbed land, biodiversity as well as grazing and crop lands (Zachariades et al., 2016).

Chromolaena odorata was introduced in South Africa in 1940 through the Durban harbour (Van Gils et al. 2004). It is the most common invasive alien plant species in KwaZulu-Natal, South Africa (Witkowski and Wilson, 2001). It forms dense impenetrable monospecific stands (Goodall and Erasmus, 1996) of up to 2 m in height, but can also reach 6 m in forests as a climber (McFadyen and Skarratt, 1996). The success of *C. odorata* is facilitated by its production of tens of thousands of light weight seeds which are mainly wind-dispersed (Witkowski and Wilson, 2001). A single *C. odorata* plant is capable of producing 800 000 seeds, 30% of which germinable (Witkowski and Wilson, 2001). *Chromolaena odorata* belongs to a category 1(b) invasive species under the National Environmental Management: Biodiversity Act No.10 of 2004 (Dew et al., 2017). This implies that *C. odorata* can be legally removed from private and public lands with immediate effect (van Gils et al., 2004).

1.6. Effects of alien plant invasion

There is evidence that invasive alien plants negatively affect species composition and richness of native plants (Gerber et al., 2008; Levine et al., 2003). For example, Martin (1999) reported lower sapling density and seedling richness in areas that were invaded than uninvaded areas. Similarly, Pyšek and Pyšek (1995) found that there was a reduction in the species diversity on areas that were invaded by *Heracleum mantegazzianum* compared to uninvaded areas in the Czech Republic (Levin et al., 2003).

Invasive plants are known to decrease ecosystem productivity while also altering the rate of soil nutrient cycling thereby negatively affecting the provision of ecosystem services (Ehrenfeld, 2003; Vilà et al., 2011; Gaertner et al., 2016). Invasive alien plants have the ability to dominate and have high species abundance in an ecosystem, and also affect the performance of resident species populations (Vilà and Weiner, 2004). The invader will therefore have both direct and indirect effects on the functioning of the ecosystem and influence plant community structure (Levine et al., 2003). In this study, I examine the influence of an invasive plant *Chromolaena odorata* and herbaceous plants particularly native grasses on native tree growth in a community reforestation programme at a Buffelsdraai landfill site in the Verulam area of KwaZulu-Natal, South Africa.

Buffelsdraai Landfill Site is under rehabilitation by a reforestation programme that emphasises active involvement of nearby communities in propagating seedlings and planting them on site (Douwes et al., 2015). The programme collaborated with the Wildlands Conservation Trust as a site-implementing agent. The main aim of the Buffelsdraai reforestation project was to reduce elevated greenhouse gas emissions emanating from the 2010 FIFA World Cup hosted in the city of Durban (Douwes et al., 2015). This study focuse on the buffer zone of the landfill, which is an open former sugarcane field planted with seedlings of diverse tree species. Previously, the site was part of a sugarcane plantation, which has since been invaded by *C. odorata* after the abandonment of a sugarcane farming.

1.7. Costs of invasive alien plant control

Costs on IAPs control in South Africa are funded by the state. The annual cost associated with invasion control on Working for Water operation increased from 25 million Rands in the year 1995/96 to over 400 million Rands in 2003/04 financial year (Kettenring and Adams, 2011). It has been estimated that different organisations have spent close to 350 million Rands over the past 20 years on IAPs control in the Kruger National Park alone (Marais and Wannenburgh, 2008; van Wilgen et al., 2017). A total of R102 million Rands has been spent on research alone for biological control agents on invasive species of *Chromolaena, Pinus, Acacia* and *Opuntia* genera amongst others in South Africa in 2008 (De Lange and van Wilgen, 2010). However, the designated funds quickly run out (La Maitre et al., 1996). This decline in funds is not a result of authorities failing to deliver but through funds competition from other projects that need and target government funding. The disadvantage faced with declining funds is the reduction of water supply to agricultural sectors, industries and cities and it is important to evaluate and address if such problems can be afforded. For example, the city of Cape Town, South Africa, receives some of its water from the catchment area which is 250 000 ha in size, this could imply that about 87.0 million m^3 of water can be lost to invasion annually if not controlled (Le Maitre et al., 1996). As a result, the city started IAPs management programme in 2008 which had an annual budget of about R1.1 million on a semi-skilled team (Gaertner et al., 2016).

In South Africa, programmes such as the Working for Water aim at creating employment and alleviating poverty for poor people, therefore, they provided substantial and additional funds of about R30 million in 2014 towards IAPs control (Gaertner et al., 2016). Hill and Coetzee (2017) made an observation that manual clearing of invasive weeds can be successful and at the same time labour intensive. However, it is one of the control methods that create jobs for the unemployed on controlling terrestrial weeds, while it is ineffective on water weeds (van Wilgen et al., 2017). In areas like the Kruger National Park, the use of mechanical control on aquatic weeds is totally unsuitable because of *Crocodylus acutus* and *Hippopotamus amphibius* (van Wilgen et al., 2017). Although management of alien-plant invaded ecosystems is of vital importance, it comes with considerable costs: it is labour-intensive, is time-consuming, and has short time results (Reid et al., 2009; te Beest et al. 2012). For example, about R825 million Rands was spent on IAPs control initiatives of the working for water programme between 1995 and 2000 (Reid et al., 2009).

1.8. Management of *Chromolaena odorata*

1.8.1. Biological control of Chromolaena odorata

Biological control is the use of introduced natural enemies or predators to control the spread of invasive plants and pests (Zachariades et al., 1999). Biological control programmes for *C. odorata* were introduced in South Africa in 1988 (Zachariades et al., 2016). This was introduced because it was considered a quicker and sustainable way to control *C. odorata*. Biological control was considered cost effective and required less labour compared to chemical and mechanical control (Cock and Holloway, 1982). One of the biological control agents for *C. odorata* which showed success was the leaf-feeding moth *Pareuchaetes pseudoinsulata* (Lepidoptera: Arctiidae). This insect was established in Sri Lanka in the 1970s by the Commonwealth Institute of Biological Control in collaboration with the Nigerian Institute for Oil Palm Research (Zachariades et al., 1999; Zachariades et al., 2016). During the late 1980s, the moth *Pareuchaetes pseudoinsulata* was tested on *C. odorata* by the Agricultural Research Council’s Plant Protection Research Institute at Cedara in KwaZulu-Natal, South Africa. The agent was then released in Durban on sites which were dominated by the *C. odorata* but did not establish successfully due to egg predation (Zachariades et al., 2016). Other biocontrol agents such as *Pareuchaetes aurata* and *Pareuchaetes insulata* (moth species) were introduced to control *C. odorata*, but again these were unsuccessful (Zachariades et al., 2016). Current efforts on biocontrol focus on finding a more effective biocontrol agent that will successfully reduce *C. odorata* populations.

1.8.2. Chemical control of Chromolaena odorata

Chemical control using herbicides has been successful on individual plants than large populations and this is more effective when the herbicide is applied on freshly cut stems (Goodall and Erasmus, 1996). However, due to the species’ shallow root system it is encouraged to facilitate mechanical procedures (Goodall and Erasmus, 1996; Zachariades et al., 2016). Chemical control is useful and very effective for most IAPs (Hellman et al., 2008). However, the use of herbicides to control *C. odorata* is not a common practice. Limited use could be a result of unselective effects of the herbicides on neighbouring non-target plants as well as environmental concerns (Erasmus, 1988). A disadvantage for most herbicides is that they need to be registered for approval before being used and their control prescriptions are expensive while sometimes being site specific (Goodall and Erasmus, 1996). Even so, the use of a number of herbicides has been restricted in the past due to areas being unsuitable for their use (Erasmus, 1988). Other restrictions on the use of herbicides are related to height differences among target plants, some herbicides may be more effective on seedlings than adult plants (Erasmus, 1988).

1.8.3. Mechanical control of C. odorata in South Africa

Mechanical control of IAPs has been the most used and most effective technique in South Africa, even though it is only a short term solution (Hellmann et al., 2008). This control measure, like any other, has its downfall such that its usefulness and effectiveness may differ by location (Hellmann et al., 2008). Mechanical clearing of IAPs mainly uses hand-tools such as bush knives, hand hoes (ploughing), picks and mattocks, chaining and bulldozing, sometimes tractor-drawn mowers and motorized brush-cutters are utilised (Paynter and Flanagant, 2004; Hellmann et al., 2008). A major advantage of mechanical control is that it prevents flowering and seed production if the cutting or removal was undertaken during the most appropriate time of the year in the end of the wet season, beginning of the dry season.

In South Africa the Working for Water programme has used up to 171.8 million Rands eradicating *C. odorata* alone from the year 1998 until 2008 (Dew et al., 2017). The control of *C. odorata* in South Africa has been ongoing since 2004 at the Hluhluwe-iMfolozi Park, KwaZulu-Natal, by the large scale control programme. The used control programme was adapted from the Working for Water programme and used their basic methods. After a period of 10 years, the Working for Water programme employed a considerable number of labourer from the surrounding communities and managed to successfully decrease the species infestation to a maintenance level of 5% at a cost of about 103 million Rands (Dew et al., 2017).

1.9. Tree-grass competition in savannas

Savanna ecosystems cover about one-fifth of the earth's land surface and are characterized by a continuous layer of grasses and scattered trees (Scholes and Archer, 1997; Ludwig et al. 2001; Ludwig at al. 2004; Sankaran et al., 2004; Riginos, 2009). The causes of the co-existence of grasses and trees in savanna biomes is one of the most prevailing topics in terrestrial ecology (Ludwig et al., 2004; Riginos, 2009). Savanna ecosystems act as distinctive habitat for a lot of large herbivores which depend on the mixed tree-grass system for survival (Riginos, 2009).

Tree-grass competition is defined as the ability of one plant form to have an effect on the other form through the availability and quality of shared resources causing harm to the next individual (McConnaughay and Bazzaz, 1992). The elements and mechanisms that allow for trees and grasses to coexist, as well as the factors that determine their relative proportions across different savannas are still unknown (Sankaran et al., 2004). This is because most studies have solely focused on the effect that savanna trees have on the functioning of the grass layer and the species growing underneath the trees and not on open grasslands (Ludwig et al., 2004). The effect of trees on grasses can either be positive, neutral or negative based on certain characteristics of grasses and tree growth forms, and most importantly the resource requirements for the plants (Scholes and Archer, 1997).

The distribution and relative abundance of woody plants and grasses in savanna affects important aspects of ecosystem functioning, such as the changes in carbon, nitrogen and nutrients cycling, and resource storage and hydrology (Riginos, 2009). In addition, the relative abundance and the distribution of trees and grasses are determined by the dynamic interactions of abiotic factors such as climate, geology, topography and fire (Scholes and Archer, 1997; Riginos, 2009). In savannas, the presence or absence of woody vegetation can change the composition and productivity of grasses and also alter the spatial distribution of grasses (Scholes and Archer, 1997).

Trees and grasses compete for below ground resources such as nutrients and water, and in doing so are able to decrease soil fertility and soil moisture (Ludwig et al. 2001). Furthermore, if there is any change in the availability of resources, trees may influence the productivity of plants growing under them (Ludwig et al., 2004). Grasses that grow underneath the tree may have access to nutrients resources via litter fall and they may also benefit from water use since temperatures are lower under tree canopies, allowing grasses growing underneath the tree canopy may experience reduced light availability from shading (Scholes and Archer, 1997; Ludwig et al., 2001). Grass species composition away and underneath the trees is more distinct in areas where there is lower rainfall than in areas of higher rainfall. This implies that environmental gradients are strong in areas where the effect of radiant energy and root competition have a greater effect on species interactions (Scholes and Archer, 1997).

The uneven distribution of trees in savannas often influences grasses growing outside and underneath the tree canopy (Whitecross et al., 2017). Determining what makes the plants growing under trees change their productivity as well as species composition is not easy since trees control the light, water and nutrient availability of these plants (Ludwig et al., 2004). There are differences in growth rate, leaf area expansion and accumulation of biomass in grasses that receive low light intensity underneath the tree canopy compared to those growing away from the canopy (Whitecross et al., 2017). These differences are more common in high rainfall areas than low rainfall. Grass to grass competition may be more severe between grasses that grow outside tree canopies, particularly because of resource limitation (Belsky, 1994; Whitecross et al., 2017).

Riginos (2009) reported that most studies and competition-based models distinctively show niche partitioning mechanisms, where grasses and trees refrain from competition especially competition for water. However, these models predict that bigger trees are better competitors to grass and also grasses invoke minimal impact on the survival and growth of larger trees (Scholes and Archer, 1997; Riginos, 2009). A few studies however reported cases in which grass competition suppresses tree growth although in such cases the trees are mostly in their seedling establishment stage (Honu and Dang, 2000; Riginos, 2009; Flory and Clay, 2009). Experimental studies suggest that the rooting of trees and grasses may overlap considerably, indicating that large trees may also compete for limited resources such as soil moisture with grass (Smit and Rethman, 2000). The co-existence of trees and grasses is complex; this is because a large number of studies have reported that larger trees suppress grasses while some studies reported that grasses can only suppress trees on their seedling stages (Belsky, 1994; Ludwig et al., 2004; Riginos, 2009).

1.10. Resprouting and reseeding strategies in plants

Savannas are characterised and shaped by disturbances such as herbivory, fires, which are the most common type of disturbance in savannas. Frost, drought, and anthropogenic activities are some of the disturbances that damage or kill above ground plant parts (Lawes and Clarke, 2011). As a result of these disturbances, some plants die, and others undergo resprouting or regeneration (Bond and Midgley, 2001; Lawes and Clarke, 2011). Resprouting is the ability of plants to regrow after any kind of disturbance through the renewal of lost parts using stored resources (Cruz et al., 2003; Nzunda et al., 2008). Plant regrowth is facilitated by stored resources in roots or shoots depending on the plant species and type of ecosystem in which they occur (Cruz et al., 2003; Nzunda et al., 2008).

In most ecosystems, disturbance and resource availability are considered to be the defining factors that stimulate the growth of plants together with their growth organs (Paula and Pausas, 2011. Resprouting plants invest energy and fitness by distributing their resources to structures (e.g. below-ground organs such as roots) that improve chances of survival following a disturbance event (Knox and Clarke, 2005; Paula and Pausas, 2011). The time at which a disturbance occurs is important because perennial plants have non-structural carbohydrates stored for spring regrowth other than for respouting after disturbance (Klimešová and Klimeš, 2003). The time at which a disturbance occurs largely influences the growth capacity of the resprouts due to the amount of resources stored since they determine the initial stages of regeneration (Klimešová and Klimeš, 2003). As such, plants have differences in the amount of the bud bank and stores of non-structural carbohydrates throughout the year, with maximum and minimum storage in the dry and wet seasons, respectively (Klimešová and Klimeš, 2003; Knox and Clarke, 2005). The degree to which different plants resprout is dependent on the age and size of the plants, ecosystem type as well as the intensity of the damage during a disturbance (Bond and Midgley, 2001; Bond and Midgley, 2003).

Plants with resprouting abilities occur mostly in areas of frequent disturbance, as well as intermediate disturbance, compared to those plants which are seed producers (Klimešová and Klimeš, 2003; Nzunda and Lawes, 2011). This is because resprouters persist by resprouting and produce fewer seeds while reseeders (seed producers) are weak resprouters (Klimešová and Klimeš, 2003; Nzunda and Lawes, 2011). Furthermore, resprouters tend to be deep-rooted while non-resprouters are mostly shallow rooted and for this reason, resprouters have lower shoot: root ratios than non-resprouters (Bell, 2001; Paula and Pausas, 2006, 2011). The main reason for the difference may be that resprouters allocate and store more resources to roots for regrowth (Paula and Pausas, 2006). Since resprouters tend to be deep-rooted, they can access ground water and so tend to be drought-tolerant (Paula and Pausas, 2006). Non-resprouters have specialized above-ground organs that allows for storing resources to enhance survival in dry seasons (Paula and Pausas, 2006, 2011). However, literature on how plants respond towards the timing of disturbance and mechanisms is limited (Cruz et al., 2003).

1.11. Recovery from disturbance-induced damage

The mechanisms plants use to deal with disturbance are vital from both ecological and evolutionary perspectives (Bond and Midgley, 2003). Plant species that have resprouting abilities can occupy the same area for hundreds to thousands of years with no changes in their population dynamics (Bond and Midgley, 2003). The population of a particular resprouting species will only be minimized during a disturbance if turnover of that population is reduced as a result of more species introduction (Bond and Midgley, 2001).

There are a number of variables that are involved in plant regrowth and productivity after disturbance (Meiss et al., 2008). The condition of leaf area or plant tissues that are active for photosynthesis after physical damage is one of the important variables that define plant productivity (Meiss et al., 2008). For a plant to regrow after physical damage, new leaf area must be established quickly enough to restore shoot: root functional equilibrium (Meiss et al., 2008). The process, however, might be slow due to the reduction or absence of photosynthesis and also the size and morphology of the plant (Meiss et al., 2008). The availability of nitrogen also influences the recovery of plants. If there is an absence or reduction of photosynthesis in plants, plants will need sufficient reserves of non-structural carbohydrate resources to be mobilized from the roots (Meiss et al., 2008). However, these carbohydrates will be exhausted if the plant is defoliated repeatedly (Meiss, 2008). In addition, external factors such as light, water, competition, temperature, and pathogens may affect overall plant performance (Cruz et al., 2003).

1.12. Invasion of the buffer zone of Buffelsdraai Landfill Site

The invasion of alien plants in the site of the Buffelsdraai Community Reforestation Programme is a major threat to reforestation success. A number of IAPs that invaded

Buffelsdraai include woody shrubs such as *C. odorata* and *L. camara*, and trees such as *Eucalyptus spp*., *Melia azedarach* and *Solanum mauritianum*. However, *C. odorata* is the most dominant invasive plant on the site (Mugwedi et al. 2017).

Control of woody invasive plants in Buffelsdraai is done by brush-cutting the stem at 50 mm above the ground once a year (Mugwediet al., 2017). However, *C. odorata* is capable of resprouting after cutting), which is a problem for the eThekwini municipality in the management of Buffelsdraai landfill site. Buffelsdraai landfill site is highly invaded by *C. odorata,* therefore, there is a need to effectively manage and control this weed as it is posing problems for the Community Reforestation Programme through competition with native planted trees (Mugwedi et al. 2017). To address the problem of *C. odorata,* the aims and objectives indicated below were generated.

1.13. Aims and Objectives

Aims

This study sought to examine the influence of an alien invasive shrub *Chromolaena odorata* and grasses on native tree growth at Buffelsdraai Landfill Site and also to assess the effect of repeated cutting on *C. odorata*'s ability to resprout and persist.

The objectives of this study are:

1. To determine whether *C. odorata* and grass competitively suppress tree growth (*Erythrina lysistemon, Brachylaena discolour* and *Vachellia natalitia)*; and

2. To determine the effectiveness and the impact of subjecting *C. odorata* to different cutting frequencies at different heights.

1.14. Outline of thesis

This thesis contains four chapters. The first is introductory, and contains the background of the study. I also provide an overview of restoration of ecosystems in the context of global climate change and alien plant invasion. Also, the management of alien invasive plants is discussed. Additionaly, I briefly describe tree-grass interactions in savannas and end with the aims and objectives of the study.

Chapter 2 investigated whether the invasive alien shrub *Chromolaena odorata* and grass species competitively supress native tree growth and if this effect varies with tree species, which is relevant for understanding the constrains to restoration.

Chapter 3 investigated mechanical control of *C. odorata.* Here, I established whether repeated cutting of the shrub significantly interfere with its ability to resprout from the stem and whether repeated cutting influences survivorship of the plants.

Chapter 4 is a concluding and recommendations chapter which integrates the findings of chapters 2 and 3.

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Chapter 2: The influence of understory plants on the growth of established tree saplings in a community restoration initiative

2.1. Abstract

Ecosystem restoration seeks to create an ecosystem that is self-sustaining or works independently on its own and that is able to tolerate perturbation. Active restoration involving planting trees may be constrained by competition from alien and native plants such as grasses. The aim of this study was to examine the influence of the invasive plant, *Chromolaena odorata,* and native grasses on the growth of native trees planted as an ecosystem restoration initiative at the Buffelsdraai Landfill Site in KwaZulu-Natal, South Africa. Two hundred and sixty-four (264) trees of three species (*Erythrina lysistemon, Brachylaena discolour* and *Vachellia natalitia*) of 1-3 m in height were selected for this study. Eighty-eight trees of each species were subjected to four treatments: (i) control, where no clearing of other plants was undertaken around the planted trees, (ii) removal of *C. odorata* around the trees, (iii) removal of grass around the planted trees, and (iv) removal of both grass and *C. odorata* around the planted trees. The following tree measurements were done on selected species; stem length and diameter, trunk basal diameter, tree height and canopy diameter, under four treatments over one year. Trunk basal diameter, canopy diameter, and tree height growth of each tree species were analysed using generalised linear models after twelve (12) months of growth. Tree height growth comparisons among treatments were analysed using a non-parametric Kruskal-Wallis test. A correlation test was performed to measure the relationship between the mean stem length and mean stem diameter for each species. There were no significant difference between all four treatments across the tree species ($P > 0.05$), but there was a significant difference between trunk basal diameter growth of *E. lysistemon* growing with cleared grass and *C. odorata* and of that growing with *C. odorata* ($P = 0.033$). Results from this study showed that the removal of *C. odorata* at the Buffelsdraai community project did not have an effect on planted trees, however it reduced the competition for nutrients, light and space for native plants as well as improved survival for planted trees. *Chromolaena odorata* had no effect on the growth dynamics of trees since they performed well in the presence and absence of grass and the invasive *C. odorata*. In addition the sampling duration of this study was short which did not give trees enough time to show growth. Sub-adult pioneer species were used for this study which could be a possible influence for these results and this is because pioneer species grow faster and perform well

under disturbed sites such that they may not be affected by the presence of understory plants near them.

Keywords: Alien invasive plants, *Chromolaena odorata*, competition, native trees, grasses, growth rate.

2.2. Introduction

Ecological restoration explores the processes and activities that assist the recovery of a previously damaged or destroyed ecosystem back to its healthy state (Alexander et al., 2011; Suding et al., 2015; McDonald et al., 2016; Chazdon et al., 2017;). This can be achieved through the clearing of invasive weeds and reintroduction of species populations, i.e., transplanting seedlings of native species or through seeding (Funk and Vitousek, 2007; Brudvig et al., 2011; Suding et al., 2015; Reid et al., 2018; Caughlin et al., 2019). Seedling transplantation works as an immediate approach to restore ecosystem structure and biodiversity in disturbed ecosystems (Caughlin et al., 2019; Reid et al., 2018). This is in comparison with restoration through sowing seeds in an ecosystem, which facilitates species richness, especially in areas where dispersal is constrained and establishment opportunities are reduced (Brudvig et al., 2011; Caughlin et al., 2019). The goals of ecological restoration may be linked to alleviating the effects of climate change, increase ecosystem stability and species biodiversity to a self-sustaining state in the shortest possible time, while also implementing benefits to human beings and natural systems (Alexander et al., 2011; Jones et al., 2018; Reid et al., 2018).

Restoring a degraded ecosystem has many challenges that delay its progress, and these include invasion by alien plants, low rainfall, uncontrolled fires and herbivory (D'Antonio and Meyerson, 2002; Chazdon et al., 2017; Crouzeilles et al., 2017). Invasive alien plants hinder the growth of native plant seedlings and tree saplings by competing with them for limited resources such as sunlight, nutrients and water (Prior et al., 2018; Kunstler et al., 2011). Uncontrolled fires, unsustainable harvesting of trees for firewood and overgrazing by livestock and wildlife are also some of the major challenges that slow down ecological restoration (Ciccarese et al., 2012; Crouzeilles et al., 2017). Failure to control such pressures leads to unsuccessful restoration and a damaged ecosystem, which results in continued loss of biodiversity and habitats, poor soil nutrient status and poor water quality (Alexander et al., 2011; Ciccarese et al., 2012). Restoration of degraded and damaged ecosystems is an expensive practice, especially in sites where the area has been previously managed poorly (D'Antonio and Meyerson, 2002). This leads to decreased contents of soil organic carbon, soil nutrients and an increase in soil compaction, which may lead to severe degradation (D'Antonio and Meyerson, 2002; Treuer et al., 2017).

While acquiring a healthy and self-sustaining ecosystem, there are positive (e.g. resource partitioning) and negative (e.g. resource competition) plant interactions that are vital for balancing the composition and the dynamics of plant populations (D'Antonio and Meyerson, 2002). Plant-plant competition is defined as the ability of a plant to have an effect on the other through the availability and quality of shared resources causing harm to the next individual (McConnaughay and Bazzaz, 1992; Looney et al., 2018). Resource competition is one of the important aspects that governs the interaction between plant populations (Kunstler et al., 2011). Therefore, resource competition is important in determining plant community structure, especially when considering that different plant species, at different growth stages, require particular amounts of resources (Kunstler et al., 2011; Xu et al., 2018).

Trees and grasses compete for below and above ground resources such as nutrients, water, and sunlight (Cabal and Rubenstein, 2018). In so doing, the competition for resources enables a decrease in nutrient and soil moisture availability, leading to decreasing growth and establishment of other plants (Cabal and Rubenstein, 2018). For example, *Vachellia tortilis* had a significant effect on the growth and production of grasses under its canopy compared to grass growth outside its canopy (Ludwig et al., 2001).

Compared to native plants, invasive alien plants are known to be better competitors for resources and establish themselves more successfully, while reducing the abundance of native plants (Nkambule et al., 2017; Bradleyet al., 2009; Reid et al., 2009; van Wilgen et al., 2007). The competitiveness and success of invasive plants is dependent on the environment and the biotic factors of the invaded community (Bradley et al., 2009). For example, a native of eastern Canada, hardwood tree, *Acer saccharum*, showed significantly less mycorrhizae growth, slower biomass accumulation and lower germination rates when grown on soils previously invaded by *Alliaria petiolate*, compared to trees grown on soils not previously invaded (Stinson et al., 2006). Also, the invasive shrub *Lonicera maackii* was shown to increase mortality of native tree saplings, reporting that it affects natural regeneration of secondary forests in the USA (Gorchov and Trisel, 2003).

Grasses compete for resources with trees (Sankaran et al., 2004; Bond, 2008) while trees are also known to compete with grasses (e.g. Scholes and Archer, 1997; Ludwig et al., 2001; Riginos et al., 2009). However, some studies and competition-based models predict that bigger adult trees are better competitors to grass and grass pose a minimal impact on the growth and survival of bigger trees (Honu and Dang, 2000; Riginos, 2009). There is a lot of complexity on the co-existence of trees and grasses in savanna ecosystems, this is because several studies have reported that larger trees suppress grasses while a few other studies reported that grass can only suppress trees at seedling stages (Belsky, 1994; Honu and Dang, 2000; Ludwig et al., 2001; Riginos, 2009).

There is insufficient information as to why some plant species are invaders and what makes them successful invaders (Seabloom et al., 2003; Funk and Vitousek, 2007; Meijninger and Jarmain, 2014). Plant invaders may be superior and have a competitive advantage over native plants because they possess particular traits such as deep tap roots that increase access to soil nutrients and water or because they are able to reproduce and grow quicker than native plants (Seabloom et al., 2003). Improved understanding of invasion mechanisms is important as it may contribute towards strategies for invasive plant management and control, as well as identifying potential invasive plants and finding ways to control them to restore biodiversity (Seabloom et al., 2003). This study aimed to investigate the influence of an invasive plant, *Chromolaena odorata,* and native grasses on native tree growth at the Buffelsdraai Landfill Site. I predicted that the invasive plant *C. odorata* will out-compete the native tree species and that grasses and native trees will be less competitive to each other. I explored whether *C. odorata* and grasses competitively suppress native tree establishment. I used a restoration site established by a peri-urban community located near the Buffelsdraai Landfill Site in the eThekwini Municipality as the focus of this study.

2.3. Methods

2.3.1. Study area

The study was carried out at the Buffelsdraai Landfill Site (appendix 3.1), an active landfill of 116 ha with a buffer zone of 757 ha, located in the Verulam area of the greater Durban Municipal Area in KwaZulu-Natal, South Africa (29.6450°S, 30.9780°E). The area receives approximately 766 mm rainfall per annum, as measured at Verulam. Most rainfall is received during the summer season (October to May). The mean annual maximum temperature ranges from 22ºC during the winter season (April to September) to 27 ºC in summer (Von Maltitz et al., 2003). The geology of the area is dominated by Dwyka Tillite while soils are highly variable and range from deep, well-drained red Hutton forms to shallow, poorly-drained Glenrosa forms (Marneweck and McCulloch, 2014; but see also Appendix 3.2 for details of soil physical and chemical properties at Buffelsdraai). The topography of the area is composed of steep slopes dissected by a large stream, the Black Mhlalasini, which flows through the northern section of the site (MacFarlane et al., 2011). The vegetation is broadly described as KwaZulu-Natal Coastal Belt which consists of subtropical forest and grassland (Mucina and Rutherford, 2006). Until the year 2010, the site consisted of sugarcane fields that have now been maintained for reforestation. Natural forest patches occur on some south-facing slopes and major drainage lines. These were not cleared during the sugarcane farming years. Other areas of the site are composed of small patches of native woodland and grassland (MacFarlane et al., 2011). The grass species found throughout the study site include *Panicum maximum* (Jacq.), *Eragrostis curvula* (Schrad.), *Sporobolus africanus* (Poir.), *Cynodon dactylon* (L). Pers. and *Melinis repens* (Willd.) Zizka, with thickets of the invasive alien plant, *C. odorata* (L.) R.M King & H. Robinson.

Buffelsdraai is under rehabilitation using a community reforestation programme that focuses on active involvement of nearby communities in raising seedlings and planting them on site. The community reforestation programme collaborated with a regional non-governmental organisation, the Wildlands Trust, and sought to mitigate against elevated greenhouse gas emissions emanating from the 2010 FIFA World Cup hosted in Durban (Douwes et al., 2015). This study used a former sugarcane field site planted with diverse native tree species. In this study, I selected the three most abundant tree species and sites with an abundant layer of *C. odorata* plants that invaded the site after abandonment of a sugarcane plantation. The trees were planted in the year 2010 and were, thus, 6 to 7 years old at the time of this study.

2.3.2. Study species

Chromolaena odorata (Asteraceae) is a fast-growing, perennial triffid herbaceous to shrubby weed that branches readily and reproduces within a year of germination from seed (McFadyen and Skarratt, 1996). The species can grow up to 2-3 m in height in favourable conditions, but can also attain heights of 5 - 6 m when supported by other vegetation

(McFadyen and Skarratt, 1996). It shows marked morphological variation of leaf shape, flower colour, hairiness, and plant architecture. The success of *C. odorata* may be enhanced by its copious production of light weight seeds which are mainly wind-dispersed (Witkowski and Wilson, 2001). A single *C. odorata* plant is capable of producing 800 000 seeds, of which 30% are germinable (Witkowski and Wilson, 2001).

Brachylaena discolour DC. (Asteraceae) occurs as single- or multi-stemmed evergreen shrubs or trees in lowland areas such as coastal and dune forests of South Africa in the KwaZulu-Natal, Eastern Cape and Mpumalanga provinces and extends to the southern parts of Mozambique and Swaziland (Boon and Pooley, 2010). The species can attain heights of 8-12m with a bark which is dark grey to brownish grey in colour (Palgrave, 1977; Boon and Pooley, 2010).

Vachellia natalitia E.Mey (Fabaceae), previously known as *Acacia natalitia,* is a tree native to southern and eastern Africa. In South Africa, the species is commonly found in the KwaZulu-Natal province, except in forests (Boon and Pooley, 2010). It is a common encroacher in grasslands (O'Connor, 1995). Individuals of *Vachellia* species mainly occur as single-stemmed trees of about 5 – 10m height under favourable conditions (Boon and Pooley, 2010). *Vachelia natalitia* occurs mostly in areas where the soils are sandy and loam. The tree is used for medicine, fuel-wood, and timber, among several other uses.

Erythrina lysistemon Hutch. (Fabaceae) is a small to medium sized broad-leaved deciduous tree that grows to a height of about 10 m (Boon and Pooley, 2010). It is relatively fast growing, and is widely distributed covering a wide range of altitudes and habitats in South Africa and extends as far as Tanzania in East Africa (Dao et al., 2009). In South Africa, it is mostly distributed in scrub forests, coastal dune bush, dry woodland as well as in high rainfall areas. The species is considered medicinal and used most widely in African traditional folk-lore (Palgrave, 1977; Dao et al., 2009).

2.3.3. Experimental design

To examine the influence of *C. odorata* and native grasses on native tree growth, 264 trees were used in this study. Three tree species *E. lysistemon, B. discolour* and *V. natalitia* with the height ranging from 1 - 3m were selected. Eighty-eight trees per species were randomly allocated to four treatments (i.e., 22 trees per treatment): (i) control, no *C. odorata* and grasses were removed underneath the planted trees canopy, (ii) removal of *C. odorata* underneath the planted tree's canopy, (iii) removal of grass underneath the planted trees canopy, and (iv) removal of both grass and *C. odorata* from underneath the planted trees canopy. The grass and *C. odorata* removal treatments were based on the assumption that important competitors of planted trees are plants that are rooted underneath the canopy of the growing trees.

2.3.4. Sampling

Tree size and growth were measured using trunk basal diameter, tree height, stem diameter growth and canopy diameter. On each of the trees, four stems were randomly selected and marked with different coloured strings for monitoring and measurement of subsequent growth. The length and the diameter growth of each stem were measured over one year. Height increment was measured as change in height over one year. Trees were measured before the removal of *C. odorata* and grass in October, 2016, and finally in September 2017. Tree size measurements (height, stem diameter growth and canopy diameter) and treatment maintenance were carried out at 3-month intervals to determine tree growth although only the initial and final size measurements were used to determine growth.

2.3.5. Data analyses

A Kolmogorov Smirnov normality test was conducted to test whether the data significantly deviated from the normal distribution, using a significance level of $\alpha = 0.05$. Because the data did not meet the assumptions for analysis of variance, a non-parametric Kruskal-Wallis test was performed to compare the tree species performance in terms of height across all treatments. A correlation test was performed to measure the relationship between the mean stem length and mean stem diameter. Treatments under each tree species were compared by tree species in terms of trunk basal diameter, stem, height, and canopy diameter using generalised linear models (GLMs), gaussian family (MASS package, Venables and Ripley, 2002). GLMs were conducted in R statistics (R Development Core Team, 2014), while Kruskal-Wallis and correlation tests were conducted in SPSS version 25 (IBM SPSS, 2017).

2.4. Results

2.4.1. Height growth

The grass and *C. odorata* removal treatments of the three tree species (*B. discolour* ($\chi^2_{(3)}$ = 0.848; P= 0.832), *E. lysistemon* ($\chi^2_{(3)}$ = 0.982; P = 0.760) and *V. natalitia* ($\chi^2_{(3)}$ = 2.168; P = 0.132) showed no significant effect on tree height growth. The pioneer species (species which are able to first colonize previously biodiverse ecosystems which had been disturbed) *V. natalitia* had the greatest height growth in the presence of *C. odorata* compared to when the trees were growing alone (Figure 2.1) (F= 0.001; df= 17; P= 0.103). *Erythrina lysistemon* had the greatest height where both grass and *C. odorata* were not cut, followed by when it was growing with the *C. odorata* alone, however there was no significant difference across these treatments. *Chromolaena odorata* showed no effect on the growth of trees in relation to tree height. *Brachylaena discolour* height growth was much greater when grown in the presence of grass, although no significant difference in these treatments. The height growth of *B. discolour* was lowest in the presence of both *C. odorata* and grass.

2.4.2. Canopy diameter

Similar to height growth, there were no significant differences in growth of the canopy across all treatments within and across the three tree species ($\chi^2_{(3)}$ = 4.836; P = 0.967). *Brachylaena discolour* had the most canopy growth compared to *V. natalitia* and *E. lysistemon* (Figure 2.2). However, there was no significant difference on canopy growth for *B. discolour* on grass and *C. odorata* removed treatment compared to no removal treatment (P> 0.05). When *B. discolour* was growing without *C. odorata* and grass, it attained high canopy diameter. *Vachelia natalitia* and *E. lysistemon* had the smallest canopy growth increment compared with the *B. discolour* trees (Figure 2.2).

2.4.3. Trunk diameter

Trunk growth was variable among species over the duration of the study but was highest when trees were grown in the absence of grass and *Chromolaena odorata*. While comparing the trunk diameter growth within the species, there was a significant difference between *E. lysistemon* that was growing with grass and *C. odorata* removed around it and that which was growing in the presence of *C. odorata* ($\chi^2_{(3)}$ = 2.058; P = 0.033). There was however, no significant difference on trunk diameter growth between all species (Figure 2.3).

There was no significant difference on the trunk growth of *B. discolour* within the four treatments ($F= 7.280$; $df= 17$; $P= 0.105$).

2.4.4. Stem length

There was no significant difference ($P \geq 0.05$) between the treatments in *V. natalitia* (Figure 2.4). Stems of *B. discolour* and *E. lysistemon* had the most growth on the control treatment. However, *B. discolour* had the least stem length when grown with *C. odorata* while *E. lysistemon* grew least when grown with grass (Figure 2.4). On the initial measurement, the stem diameter and length for *E. lysistemon* had a strong negative correlation ($r = -0.136$; $P = 0.025$). There was a positive correlation between stem length and diameter at the final measurement for *E. lysistemon* which was much stronger than that of the initial measurements ($P > 0.05$; $r = 0.31$).

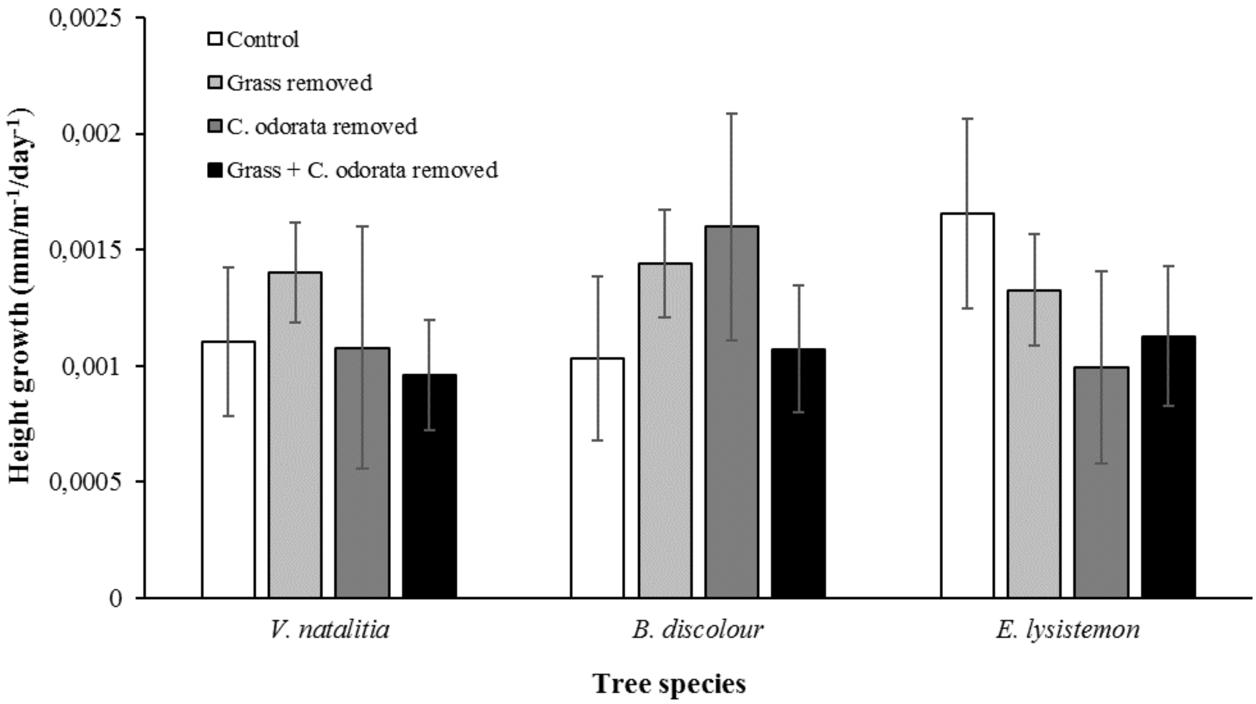

Figure 2.1. The mean (±SE) tree height growth per day of *Vachellia natalitia*, *Brachylaena discolour* and *Erythrina lysistemon* in four neighbour manipulation treatments.

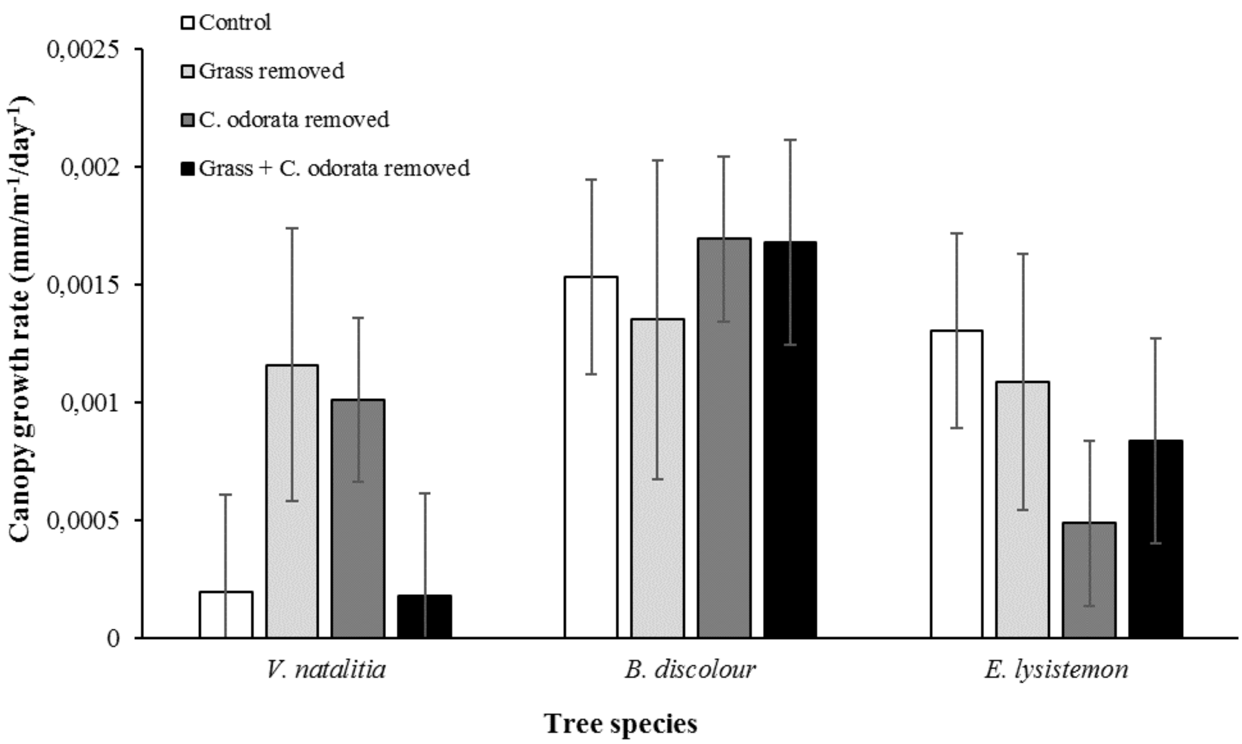

Figure 2.2. The mean (±SE) tree canopy growth of *Vachellia natalitia, Brachylaena discolour* and *Erythrina lysistemon* in grass and *C. odorata* removal treatments.

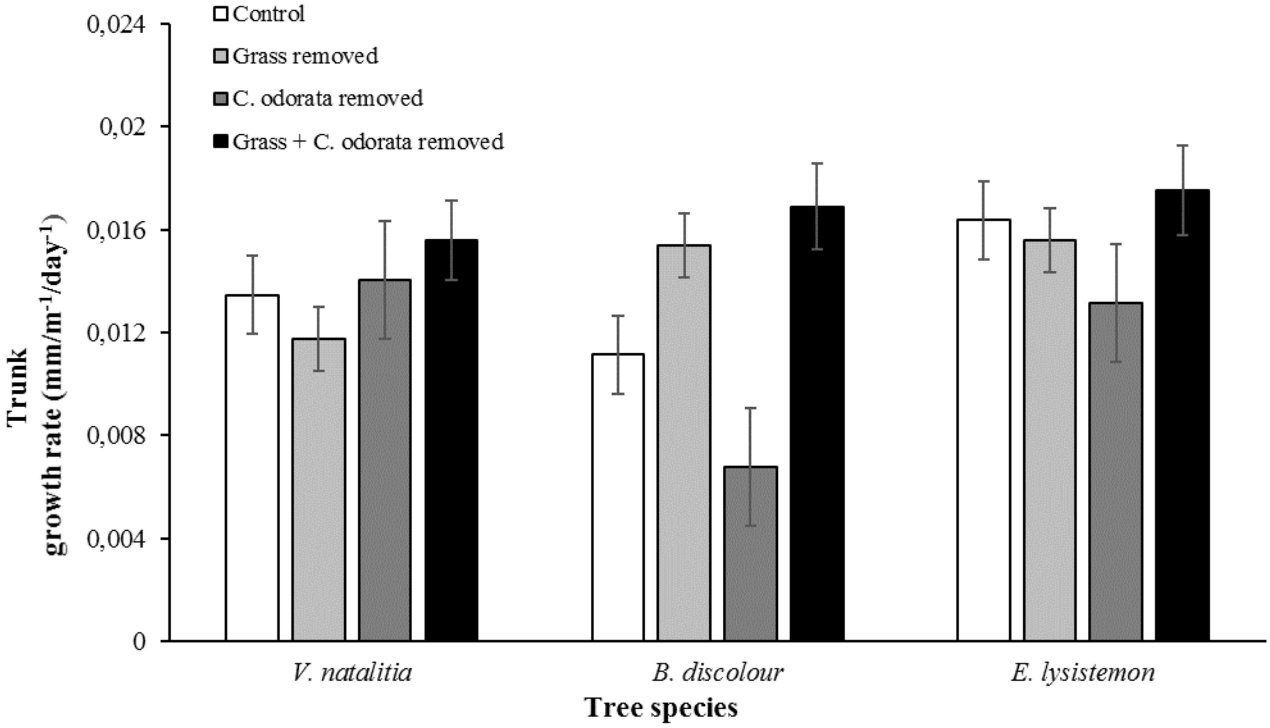

Figure 2.3. The mean (±SE) tree trunk diameter growth of *Vachellia natalitia, Brachylaena discolour* and *Erythrina lysistemon* in grass and *C. odorata* removal treatments**.**

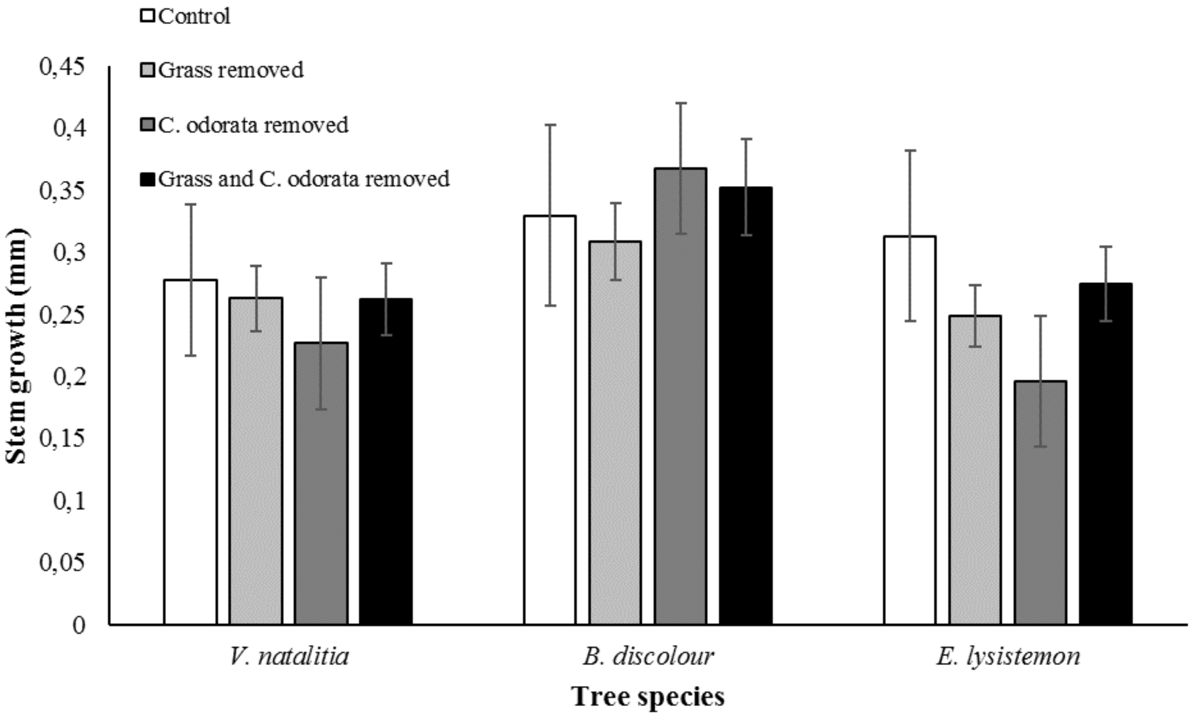

Figure 2 4. The mean (±SE) stem elongation of *Vachellia natalitia*, *Brachylaena discolour* and *Erythrina lysistemon* in grass and *C. odorata* removal treatments.

2.5. Discussion and conclusion

Many studies have reported that both invasive alien plants and grasses supress native tree growth through resource competition (e.g. Honu and Dang, 2000; Riginos, 2009; Flory and Clay, 2009). However, the results presented in this study suggest the opposite as there were no significant differences between growth of trees growing with or without *C. odorata* and grasses. Nevertheless, there was a significant difference in growth among the three species irrespective of treatments. In a study at Hluhluwe-iMfolozi Park, te Beest et al. (2012) found that the grass *Panicum maximum* was one of the most competitive grasses to *C. odorata* in areas of both low and high-water availability. The selected native trees in this study had out-grown establishment stage, which could be the other reason for the insignificant results presented by different treatments.

Over the study period, tree height and trunk diameter increased across all treatments, although these were not significantly different. This finding suggests that *C. odorata* and grass may not supress the growth of trees (> 1 m tall) in height. In other words, the mechanical control of the invasive *C. odorata* does not significantly improve the growth rate of established trees. An increase in tree height is one of the important factors that indicate the positive or negative response of trees to stress, competition or resource limitation (Honu and Dang, 2000). This is because as the trees grow higher, they are able to compete for more light, escape the browsing height and fire prone height (Honu and Dang, 2000). Contrast to this findings in a study in Ghana, Honu and Dang (2000) noted that seedlings significantly increased in height after the removal of the invasive *C. odorata* in three months. The constant and frequent clearing of invasive alien plants is necessary and is known to alter the native tree species diversity and seedling regeneration (Vidra et al., 2007; Simmons et al., 2016; Mugwedi et al., 2017). Therefore, resulting in an increase in canopy cover and high litter accumulation, which are conditions needed for regeneration of understory species (Vidra et al., 2007; Simmons et al., 2016; Mugwedi et al., 2017). Furthermore, following repeated removal of invasive alien plants, native plant communities may provide niche for indeginous trees to regenerate, reduce fire hazards that may be presented by alien invasives and improve native tree growth in a long term (Vidra et al., 2007). This may improve restoration success of studies like this at Buffelsdraai.

I found that native grass had no effect on the growth of trees, yet native grasses at mature stage in savanna ecosystems have been reported to supress tree growth often through resource competition (Scholes and Archer, 1997; Riginos, 2009; Priyadarshini et al., 2016). Grasses were reported to reduce the growth of native trees by limiting water recharge (Scholes and Archer, 1997; February et al., 2013; Wagner et al., 2014). However, the effect of grasses on native tree growth may not be comparable to that of invasive alien plants. The positive and negative tree-grass interactions may regularly be influenced by changes in the environment and plant growth requirements (Priyadarshini et al., 2016). The finding that grasses had no effect on the growth of trees can be attributed to the use of sub-adult trees for the experiment. In addition, Scholes and Archer (1997) mentioned that the combination of mature grasses and trees in savanna ecosystems is unstable.

In another study, Wagner et al. (2014) found that the diameter of trees increased as a result of soil water availability, which was influenced by the amount of rainfall. Seasonal drought and cold temperatures are abiotic conditions that limit the growth of trees and grasses given that the plants are poorly adapted to such extremes (Rundel et al., 2014). This study suggests that grasses have no impact on the growth dynamics of sub-adult trees (1-3 m tall) which may suggest that there is less resource competition between grasses and trees under the study's conditions.

Results from this study suggest that the removal of *C. odorata* at the Buffelsdraai community project does not facilitate the growth of established trees. The age of the trees, height and the duration of the study (which was very short, 12 months) could explain the similar response of these trees across all the treatments. Considering the similarity in terms of tree growth, *C. odorata* could be out-competed by pioneer species with bigger canopies in the long run since it is a shade intolerant plant (Honu and Dang, 2000). Based on the findings of this study, clearing of *C. odorata* underneath the established trees could provide positive outcomes such as minimal occurance of fire hazards at Buffelsdraai landfill Site, provide niche for native tree regeneration and also improve tree growth over a long period of time. In addition to these benefits is the reduction of *C. odorata* establishment with repeated clearing.

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Chapter 3: The responses of an alien invasive weed, *Chromolaena odorata*, to repeated cutting

3.1. Abstract

The invasive plant *Chromolaena odorata* (Asteraceae) is one of the major problems to global biodiversity, causing negative impacts on ecosystem structure, functioning and productivity. For the recovery of an ecosystem from alien plant invasion, management and an effective control of invasive alien plants are one of the most important tools that need to be implemented. The aim of this study was to investigate the effectiveness of mechanically controlling *C. odorata* through repeated cutting of stems at different heights. Twenty-eight plots (5 m × 5 m) with 15 selected *C. odorata* plants were established, at least 2 m apart, with one plot designated as the control where no cutting of the plants was undertaken. I had two treatment combinations: (1) cutting frequency made up of single, twice and thrice cutting; and (2) stem cutting height at 30, 100 and 200 mm aboveground. I used 420 plants (i.e. 3 cutting heights × 9 replicate plots × 15 plants + 15 control plants). Plants were first cut in November 2016 and the coppice regrowth was cut on plants in 18 of the plots in February 2017, and a third cut was made on plants in the last nine plots in May 2017. Stem basal diameter measurements and the number, length and diameter of resprouting shoots were measured at 3-month intervals. Mean number of resprouting shoots, mean shoot length and diameter of the resprouts, mean shoot length:diameter ratio (shoot taper function) of resprouts, mean number of shoots per unit stump circumference (shoot production: shoots mm^{-1}) and mean total shoot basal area:stump basal area ratio for all treatment combinations were analysed using the Friedman's test. The relationship between stem diameter of *C. odorata* plants and each of the number of resprouting shoots, resprout shoot length, and resprout shoot diameter were explored using Pearson correlation analysis. There was a significant difference in the number of resprouts produced across all treatment combinations ($P < 0.001$ for cutting frequency), a large number of resprouts were produced during the first cutting frequency from all cutting heights. Shoot length and diameter decreased with cutting frequency. Length and diameter of resprouts also showed significant differences between the cutting height and cutting frequencies ($P < 0.001$ in both cases). Shoot taper function and shoot production were also significant ($P < 0.001$) among treatments, with a smaller number of shoots produced in the 30 mm cutting height compared to other cutting heights. Total shoot basal area:stump basal area ratio did not differ significantly for both cutting height and cutting frequency treatments (P > 0.05 in both cases). Repeated cutting of *C. odorata* at a lower cutting height may deplete its energy reserves, reducing the number of resprouts produced, thereby leading to death of the plants. Repeated cutting at short-term intervals also prevents the plants from growing to reproductive maturity and seed production, leading to reduced infestations.

Keywords: Invasive alien plants, Cutting frequency, cutting height, resprouts, mechanical control.

3.2. Introduction

Invasive alien plants (IAPs) are one of the major threats to global biodiversity causing negative impacts on ecosystem functioning and productivity through competition and other economic, ecological and environmental problems (Shackleton et al., 2017; Dumalisile and Somers, 2017). Research on alien plant invasions and how they impact ecosystem structure and function is still relevant and important as knowledge generated from such research will continue to influence management decisions to control IAPs (Shackleton et al., 2017). However, the efforts to control and manage invasions have so far created controversy and conflicts due to many control methods being implemented, with the primary goal of alien invasive species management being elimination and reduction of their already established populations (Crowley et al., 2017; Prior et al., 2018). In ecosystems with established populations of IAPs, decisions need to be made on how to effectively manage and control these populations (Prior et al., 2018). The management methods for established populations should aim at reducing invasion to a level where their impacts will be minimised (Prior et al., 2018).

Methods implemented to control IAPs like *Chromolaena odorata* include mechanical clearing, the use of biological control agents and chemical control (De Rouw, 1991). Different studies which focus on controlling the spread of *C. odorata* have been carried out across the world, with most studies suggesting that such invasive plants must be prevented from spreading (De Rouw, 1991; Wittenberg and Cock, 2005; Prior et al., 2018). Mechanical control has been the most dominant and common method to eradicate *C. odorata* in South Africa (Malahlela et al., 2015). This method includes labour intensive manual weeding such as uprooting, slashing, burning and digging out of weeds (Van Wilgen et al., 2017). Although there are many constraints to mechanical methods such as labour intensity and being costly, it is still one of the widely used control methods due to availability of unskilled cheap labour (Dew et al., 2017). However, some alien invasive plants, including *C. odorata*, are also effectively controlled and managed by the use of fire (Witkowski and Garner, 2008).

Chromolaena odorata is capable of resprouting after manual clearing and other forms of disturbance such as fire (De Rouw, 1991). Resprouting is the ability of plants to regrow after disturbances through the renewal of lost plant parts using stored resources (Cruz et al., 2003; Klimešová and Klimeš, 2003; Nzunda et al., 2008). Resprouting plants are known to have storage organs located belowground and most of them are believed to be deep-rooted which allows them more access to groundwater (Iwasa and Kubo, 1997; Bell, 2001; Paula and Pausas, 2006). However, the long-lived perennial plant *C. odorata* stores its reserves both in roots and stems (Dube et al., 2019). Storage of resources in the roots in *C. odorata* helps the plants to quickly recover from disturbances once photosynthetic organs have been removed and it is a mechanism to tolerate shoot damage (Iwasa and Kubo, 1997; Gurvich et al., 2005; Li et al., 2012; Dube et al., 2019). The amount of stored resources, frequency and timing of disturbance in plants are some of the main determinants of whether a plant will successfully resprout or not (Bellingham and Sparrow, 2000). It is important to establish the nature of persistence of IAPs so that control techniques to be implemented are effective (Verdú, 2000).

Some alien invasive shrubs exhibit resilience through their ability to resprout or regenerate after a disturbance or through successful seed production (Luken and Mattimirro, 1991). The control of IAPs is important since they are the biggest threat to biodiversity and they may out-compete native plants (Turpie, 2004; Nkambule et al., 2017). At Buffelsdraai Landfill Site, *C. odorata* is one of the main IAPs, in addition to *Lantana camara* L., *Melia azedarach* L., *Solanum mauritianum* Scop. *Chromolaena odorata* is cut once a year at about 5 cm above ground at Buffelsdraaai Landfill Site which allows it to resprout and regrow over time (Mugwedi, 2017). Consequently, this study aimed to contribute towards improving the techniques of controlling *C. odorata* at Buffelsdraai Landfill Site by investigating the resprouting ability of *C. odorata* after having been subjected to different cutting frequencies at different heights. I predicted that *C. odorata* would use up its reserves stored both above-ground on stems and below-ground on roots and also that cutting *C. odorata* plants at a lower height would reduce their resprouting ability with most of its stem biomass removed. This prediction is made on the basis that continually defoliating a plant should lead to a depletion of its stored resources leading to death of a plant.

3.3. Methods

3.3.1. Study area

The study was conducted in the buffer zone (757 ha) of the Buffelsdraai Landfill Site (29.6450˚S, 30.9780˚E) north of Durban in KwaZulu-Natal Province, South Africa. Land use in the buffer zone was sugarcane farming for about a century but has now been planted with 59 species of native trees as part of an ecological restoration programme that integrates community involvement in climate change mitigation (Mugwedi et al., 2017). The remaining part of the buffer zone is composed of woodlands, grasslands, and small patches of forest, and most of it is invaded by alien plants (MacFarlane et al., 2011). The vegetation type of this area is KwaZulu-Natal Coastal Belt made up of subtropical forest and grassland, which is considered endangered (Mucina and Rutherford, 2006). This area receives summer rainfall of about 440-1400 mm and some rainfall during winter, with the mean annual maximum temperature ranging from 22 ºC during the winter season to 27 ºC in summer (Von Maltitz et al., 2003). The geology of the area is dominated by the Dwyka Tillite and also composed of a glacial conglomerate parent material which is hard, base-rich and resistant to weathering (Ethekwini Municipality, 2014). Soils range from deep, well-drained red Hutton soil forms to shallow, poorly-drained Glenrosa soil forms (Marneweck and McCulloch, 2014). These soils are also reported to have low organic matter content with a pH of lower than 6.5 (Mugwedi et al., 2017).

3.3.2. Study species

The species chosen for this study is an invasive shrub, *Chromolaena odorata* (L.) R.M. King & H. Robinson, a member of Eupatoriene, which is within the sub-family Asteroideae and the Asteraceae family (King and Robinson, 1987), commonly known as triffid weed. This species originates from the neo-tropical regions and is regarded as one of the most aggressive weeds in parts of southern Africa (Goodall and Erasmus, 1996; Dumalisile and Somers, 2017). It is known for its large native and non-native distribution across most parts of the world (Zachariades et al., 2009; 2016). In its native range, *C. odorata* is largely distributed in the southern part of the United States of America to the northern part of Argentina (Zachariades et al., 2009; 2016). It was introduced in South Africa in 1940 through the Durban harbour (Van Gils et al., 2004).

In the 20th century, *C. odorata* became one of the most problematic plants in South Africa where it was distributed across the eastern part of the country (Zachariades et al., 2016). The species highly impacted disturbed land, native biodiversity as well as grazing and crop lands (Zachariades et al., 2016). *Chromolaena odorata* is a medium sized shrub that can grow to 2-3 m in height in favourable conditions (McFadyen and Skarratt, 1996). The shrub is able to grow in grasslands, woodlands and forest edges (te Beest et al., 2012). *Chromolaena odorata* shows marked morphological variation of leaf shape, flower colour, hairiness, plant architecture as well as on the smell of crushed leaves in its native range (te Beest et al., 2012). The success of *C. odorata* may be enhanced by its ability to resprout after disturbance and its copious production of light weight seeds which are mainly wind dispersed (Witkowski and Wilson, 2001).

3.3.3. Sampling

To investigate mechanical control of *C. odorata,* 15 multi-stemmed *C. odorata* plants of about the same height of between 1 and 1.5 m were selected in each of twenty-eight plots (5 m × 5 m) that were established at least 2 m apart at Buffelsdraai. One plot was designated as the control where no manipulation of the plants took place. I had two treatment combinations made up of (1) cutting frequencies made up of single, twice and thrice cutting; and (2) stem cutting height at 30, 100 and 200 mm aboveground. Plants in 27 plots were initially cut in November 2016 according to three cutting height treatments of 30, 100 and 200 mm aboveground, with 9 plots cut per height. A second cut was done on resprouts of the plants in 18 of the plots in February 2017, and a third cut was made on secondary resprouts of the plants in the last nine plots in May 2017. The design of the study was therefore based on 420 plants, i.e. 3 cutting heights × 9 replicate plots × 15 plants + 15 control plants. Measurement of stem basal diameter was undertaken before cutting as an indication of plant size for all selected plants. Number of resprouting shoots, length and diameter of each resprouting shoot were measured prior to cutting as appropriate for each plant in each plot. Survival of plants per treatment was also recorded at each sampling period.

3.3.4. Data analyses

The data for the total number of shoots, shoot length and shoot diameter were subjected to a Kolmogorov Smirnov test to determine if the data were normally distributed. The data had multiple/continuous measures of the same variables and did not meet the normality assumptions for repeated measures ANOVA. Therefore, the data were analysed using a non-parametric Friedman's test to compare the i) mean number of resprouting shoots, ii) mean shoot length and diameter of the resprouts, iii) mean shoot length:diameter ratio (shoot taper function), iv) mean number of shoots per unit stump circumference (shoot mm^{-1}) and v) mean total shoot basal area:stump basal ratio for the 30, 100 and 200 mm cutting height frequencies. The relationship between stem diameter of the chromolaena plants and the number of resprouting shoots, the resprout shoot length and resprout shoot diameter were explored using Pearson correlation analysis. These analyses were conducted in IBM SPSS Statistics version 25 (IBM SPSS, 2017). The following calculations were also derived from the raw stump and resprouting shoot data, to produce the following variables from Kaschula et al. (2005): (i) shoot length:diameter ratio (coppice shoot length in millimetres divided by the shoot coppice diameter); (ii) shoot production, calculated as, number of shoots per unit stump circumference (shoots mm^{-1} = number of shoots divided by the circumference of the stump, calculated from the stump diameter); and (iii) the index of allocation of resources to resprouting shoots relative to stump size, calculated as the total shoot basal area: stump basal area ratio (basal area of all shoots produced by the stump).

3.4. Results

3.4.1. Effect of cutting frequency and height on number of resprouting shoots

The total number of resprouts produced were significantly greater ($P < 0.001$) for the 200 mm cutting height than the 30 mm and the 100 mm cutting heights. The 30 mm height had the lowest overall number of resprouts (Figure 3.1). There were no resprouts produced three months after the last cut of plants (May 2017) for the 30 mm cutting height, while the 100 mm and the 200 mm cutting heights produced the lower number of resprouts (Figure 3.2). There were no significant differences ($P > 0.05$) in the mean numbers of shoots produced between the February and May cutting frequency for each of the 100 and 200 mm cutting height, while there was a significant difference between the February and May cutting frequency on the 30 mm height (Figure 3.1).

There was a significant difference between the mean number of shoots produced across all treatments ($P < 0.001$). The number of shoots produced after the first cutting regime was higher than the number of shoots produced in the second and third cutting regimes (Figure 3.3). The 30 mm cutting height produced the least number of resprouts during the overall study period (Figure 3.3). There was a strong positive relationship ($r = 0.91$; $P < 0.001$) between stem diameter and the number of resprouts produced across the cutting frequencies.

3.4.2. Shoot length

The 200 mm cutting height produced significantly longer ($P < 0.001$) shoots after the first cut in November 2016 compared to other cutting heights (Table 3.1). Shoots were longest after the second cut in February 2017, particularly on the 200 mm cutting height (Figure 3.4).

3.4.3. Shoot diameter

The diameter of resprouts was significantly different ($P < 0.001$) (Table 3.1) among the 30, 100 and 200 mm cutting heights, with the 30 mm cutting height producing the smallest shoot diameters compared to those of the 100 and 200 mm cutting heights in the third cutting regime. After the third cut there were no further resprouts produced on the 30 mm cutting height treatment (Figure 3.5). After the last cut in May 2017, there was a lower number of resprouts, which were also of small diameter sizes on 100 and 200 mm cutting heights (Figure 3.5). Shoot diameter and length increased with bigger (diameter) stem sizes (Figure 3.6).

3.4.4. Shoot production

Shoot production was significantly higher in the 200 mm cut stumps after the first and second cuts, although it decreased after the last cut in May (Table 3.1; Figure 3.7).

3.4.5. Total shoot basal area:stump basal area ratio

The index of allocation of resources to resprouting shoot growth relative to the stump size, which is the ratio of total shoot basal area and stump basal area showed no significant difference ($P > 0.05$) on the February and August cutting cycles across all cutting heights (Table 3.1).

3.4.6. Shoot taper function

The resprout shoot length and diameter ratio showed significant differences ($P < 0.001$) among all treatment combinations of cutting height and cutting frequency (Table 3.1). There was a significant gradual increase in shoot taper function after the first cut in February until August for the 100 and 200 mm cutting heights (Figure 3.9). The 30 mm cutting height had the lowest shoot taper function throughout the study period, while the 200 mm cutting height had the highest (Figure 3.9).

Table 3. 1. Friedman's test results for resprouts response variables

Response variables	χ^2	df	p
Shoot length	370.4	2	< 0.001
Shoot diameter	386.7	2	< 0.001
Number of resprouting shoots	496.3	2	< 0.001
Shoot length:diameter ratio	546.3	2	< 0.001
Shoots per unit stump circumference	302.12	2	< 0.001
Total shoot basal area:stump basal area ratio	496.6	2	ns

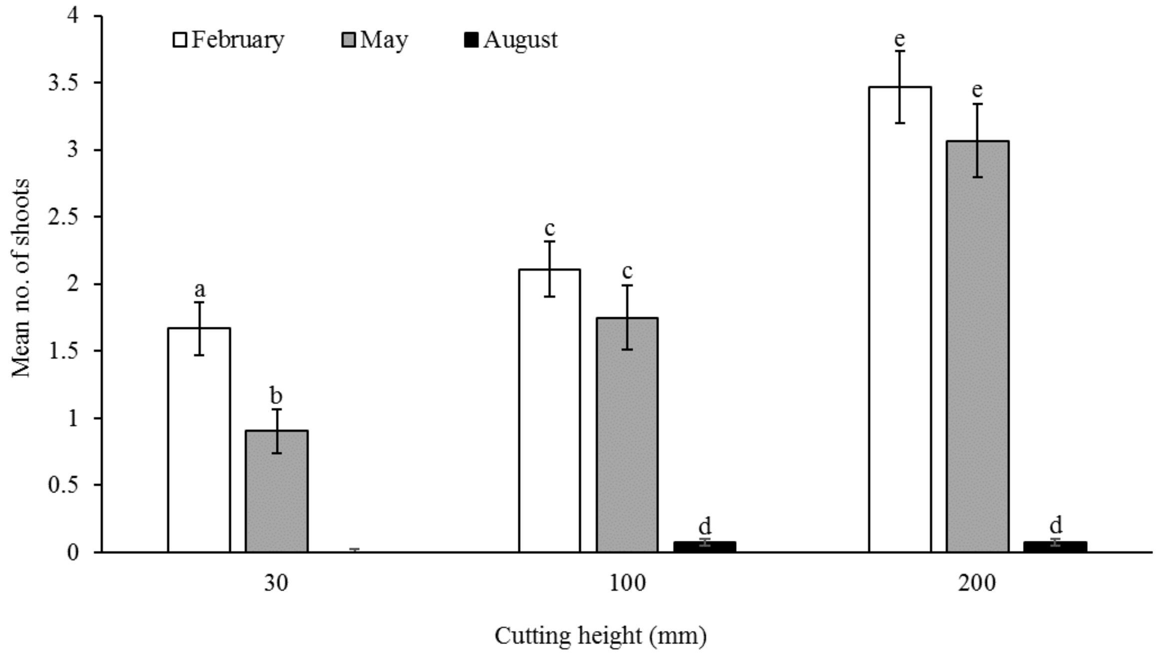

Figure 3.1. Mean (±SE) number of shoots produced by *C. odorata* plants cut at different heights and frequencies, shown as dates of measurements after 3 months of each cutting from November 2016 to May 2017. Lower case letters (a, b, c) indicate significant differences ($P < 0.05$) between treatments.

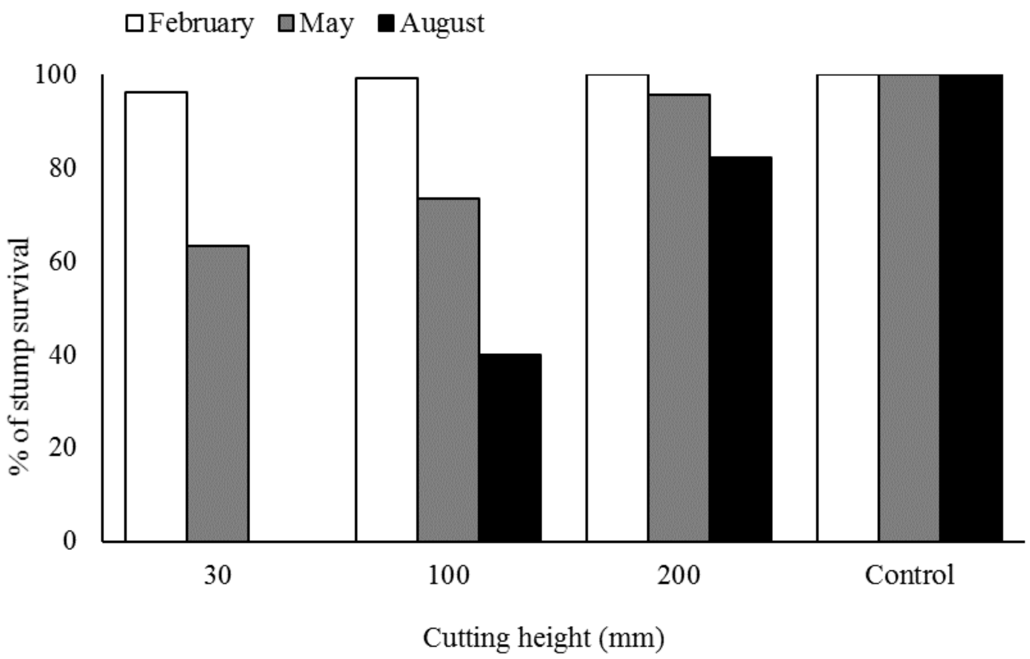

Figure 3.2. Percentage of stump survival after three cutting frequencies from November 2016 to May 2017 from each cutting height (30, 100 and 200 mm). Survivorship of control plants is also shown.

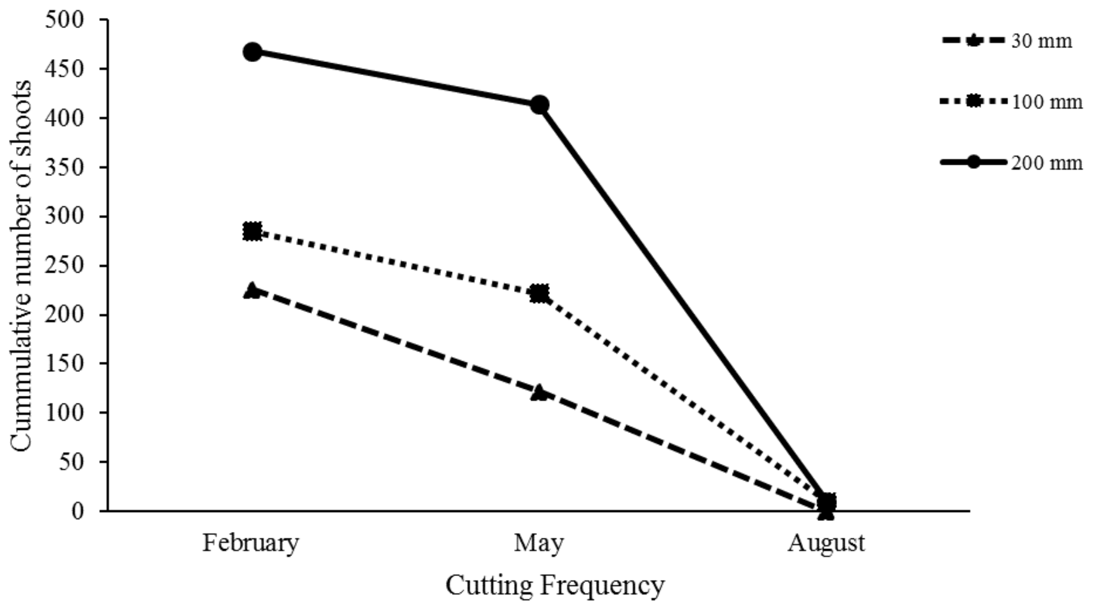

Figure 3.3. Cumulative number of resprouting shoots produced in each cutting frequency 3 months after each cut (November 2016 to May 2017) over the study period, based on 405 *C. odorata* plants.

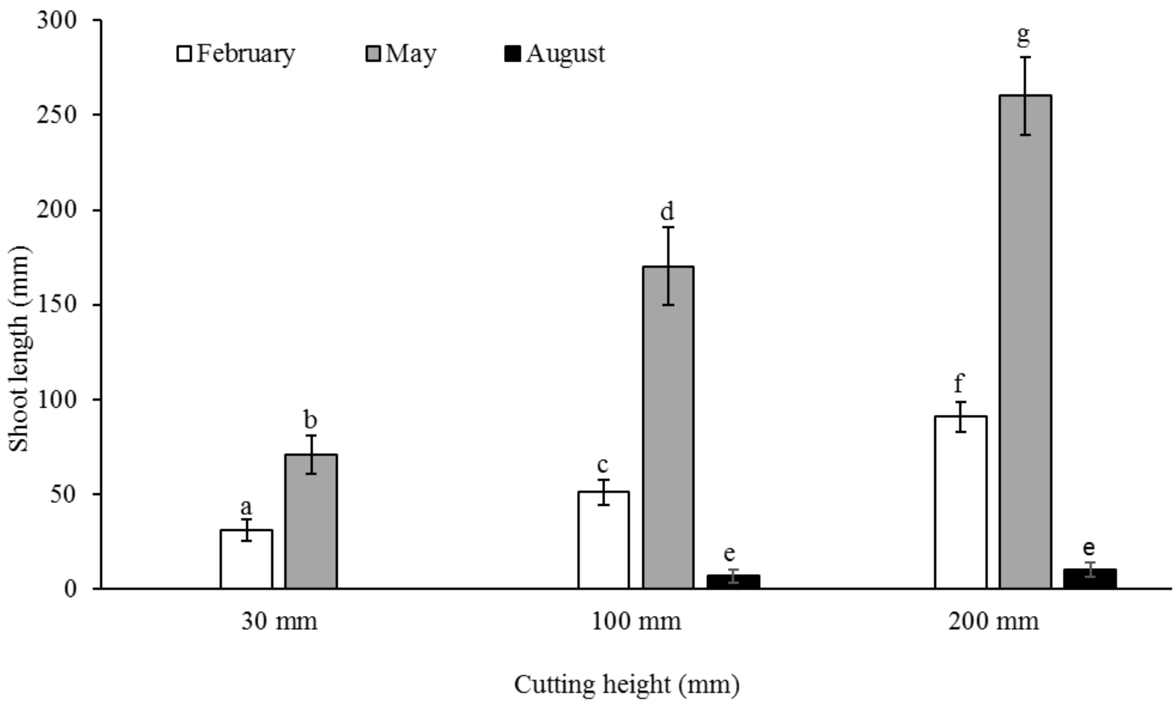

Figure 3.4. Mean (±SE) resprout shoot length of *C. odorata* 3 months after being exposed to different cutting heights and cutting frequencies carried out in November 2016, February 2017 and May 2017.

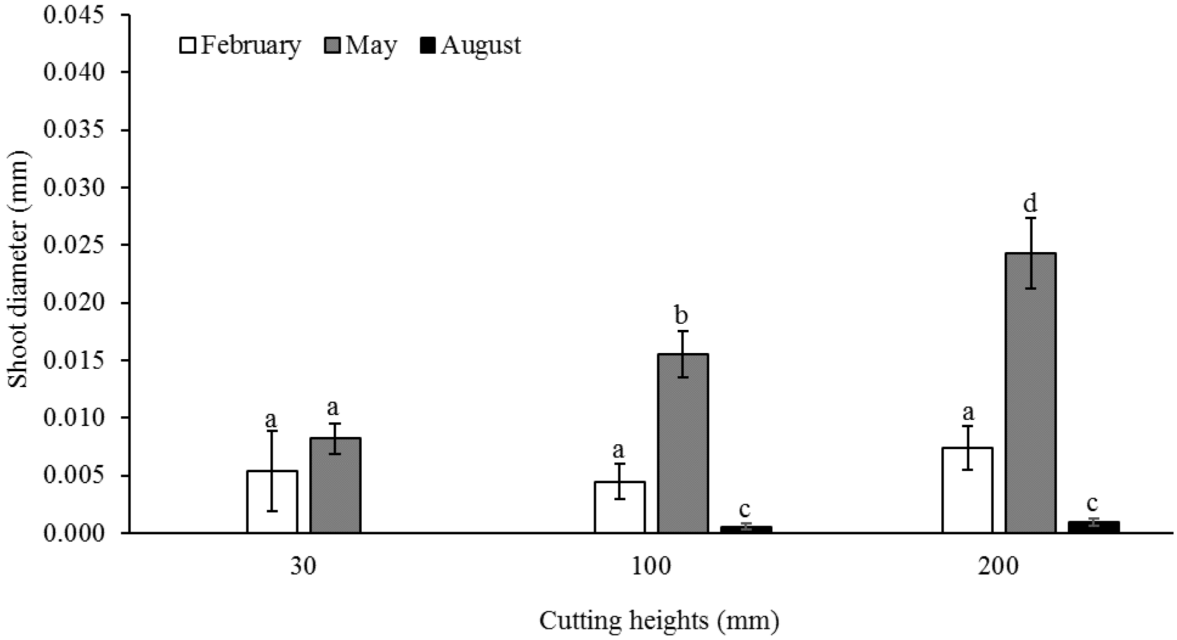

Figure 3.5. Mean (±SE) diameter of resprouts produced 3 months after *C. odorata* was cut at different heights (30, 100 and 200 mm) and frequencies (November 2016, February 2017 and May 2017).

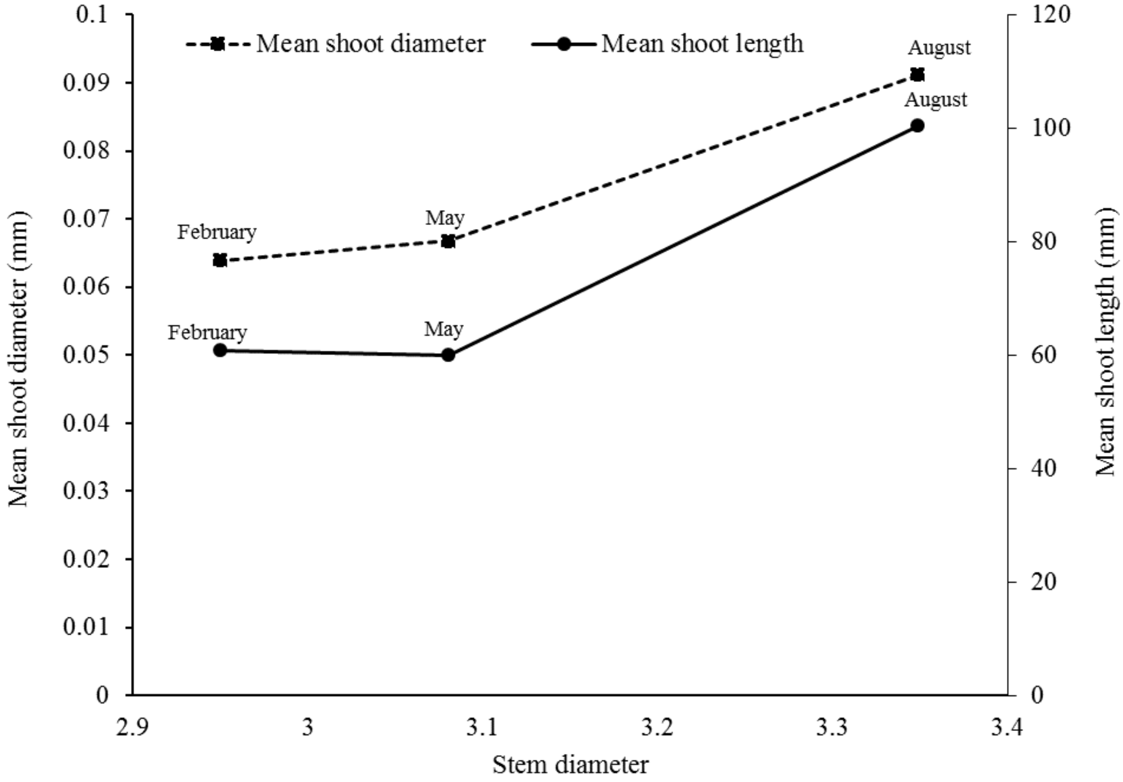

Figure 3.6. Stem diameter and mean shoot diameter and mean shoot length produced in February, May and August, i.e. 3 months after the cuts in November 2016, February and May 2017, respectively.

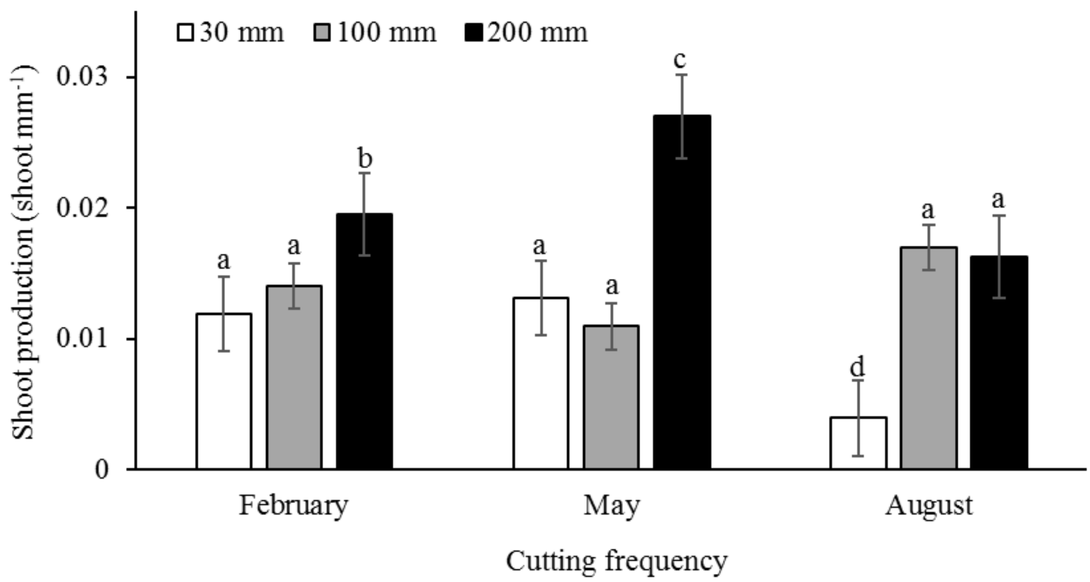

Figure 3.7. Mean (±SE) number of shoots (shoot mm^{-1}) produced across different cutting frequencies 3 months after each cut from November 2016 to May 2017.

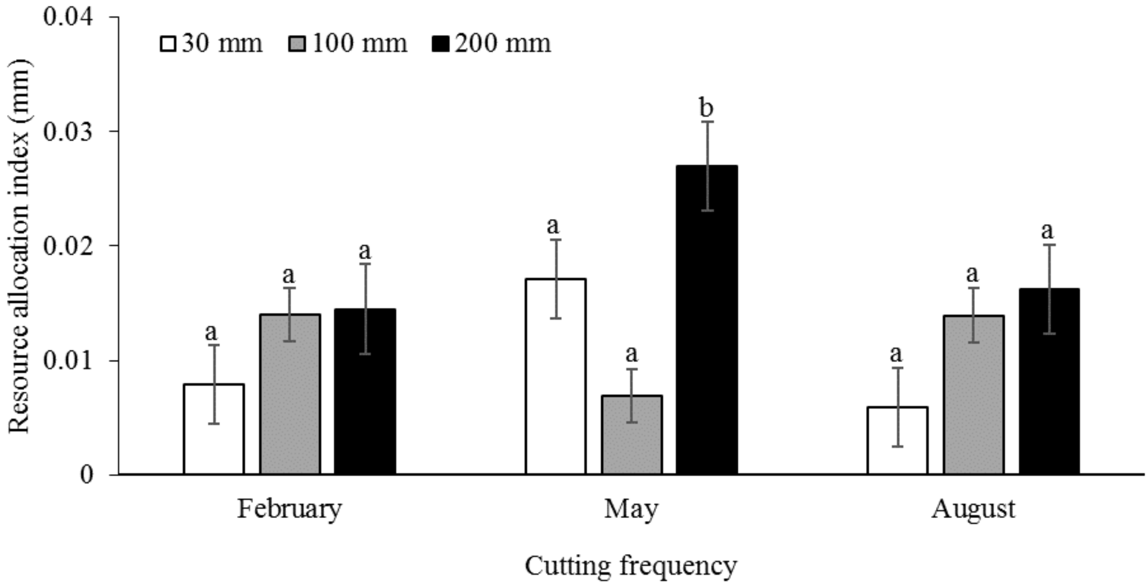

Figure 3.8. Mean (±SE) ratio of total shoot basal area and stump basal area (resource allocation index) of shoots produced in all treatment combinations.

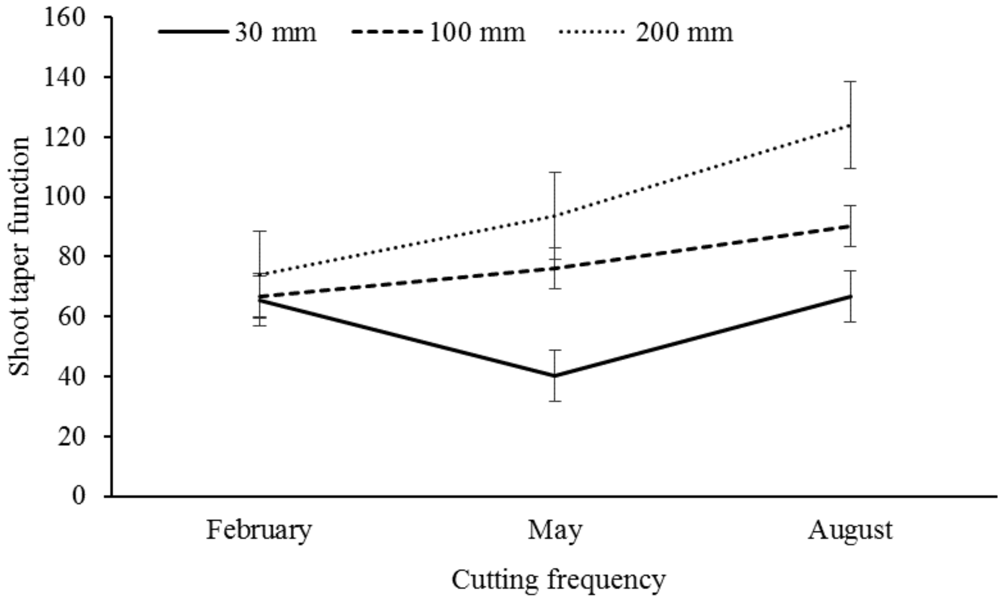

Figure 3.9. Mean (±SE) shoot taper function of resprouts produced 3 months after treatment.

3.5. Discussion

In the current study, *Chromolaena odorata* produced numerous resprouts on all the cut stems following different cutting regimes, more especially on longer remnant stumps. These results are consistent with those of Shackleton (2001), who demonstrated that the cutting of *Terminalia sericea* at a greater height had a positive effect on the number of resprouts produced in the savanna in Bushbuckridge, South Africa. Also, Syampungani et al. (2017) reported an increase in resprouting abilities in large diameter stumps which is consistent with the findings from this study. The current study also reported that as the stump diameter increased so did the length and the diameter of the resprouting shoots. Furthermore, an increase in cutting height had a positive effect on the number of resprouts produced. This is because shoot production was found to be highest in the 200 mm cutting height. This production of resprouts was related to the index of resource allocation to resprouting shoots parallel to stump size which was obtained from the total shoot basal area and stump size basal area ratio (Canadell et al., 1991; Shackleton, 2001; Kaschula et al., 2005). The ability of a plant to resprout after a disturbance is an important strategy adapted by plants to advance and increase their longevity and survival (Shibata et al., 2016). Therefore, these resprouting abilities may be a result of a trade-off between the growth of a plant and the storage reserve where the plant allocates resources and that may also differ with species (Syampungani et al., 2017).

While results from the current study show that stumps with higher cutting height produced more resprouts after the cutting cycles, Shackleton (2001), however, reported that larger diameter stem sizes take a longer time to produce resprouts following the first cutting. The same author further suggested that when trees are cut at a higher height have the ability to regrow at a fast rate compared to shorter stumps. This is because resprouting in most plants is facilitated by the amount of reserves stored in the stump and also by the underground bud activities which is then determined by the stump diameter (Kaschula et al., 2005).

Chromolaena odorata is an active and intensive resprouter which is capable of withstanding abiotic stress like droughts, also surviving harsh and severe disturbances such as fire and loss of aboveground biomass through fast regrowth from basal stems (te Beest, 2012). Experimental studies from te Beest et al. (2009; 2012) showed that *C. odorata* plants heavily invest their reserves in the stem biomass. Bond and Midgley (2001) mentioned that responding to disturbance like repeated cutting, *C. odorata* shrubs were assumed to utilise their below ground stored reserves for rapid recovery. Perenials such as *C. odorata* also utilise their roots as storage reserves as a way to tolerate aboveground disturbance (Li et al., 2012). A study by Li et al. (2012) revealed that there was substantial regrowth following the entire shoot damage in *C. odorata*. Due to repeated cutting at a lower stem height, this study showed that the stored reserves might get depleted for *C. odorata* because there was high plant mortality for the stumps that were repeatedly cut at 30 mm.

Ability to produce large numbers of seeds and to resprout after a disturbance facilitates invasive and distribution success of *C. odorata* (te Beest et al., 2012). The soils in which *C. odorata* grows may play a role in its establishment and success (Koutika et al., 2005). This species is known to grow on soils that range from dune-sands to soils with excessive clay (Agoume and Birang, 2009; te Beest et al., 2012). Also, soils under *C. odorata* consist of improved physical properties and are high in minerals, these soils consisted of higher amounts of organic carbon, total phosphorus, total nitrogen, magnesium and extractable calcium (Agoumé and Birang, 2009; te Beest et al., 2012). In this present study the soils where *C. odorata* was growing had high calcium, potassium and magnesium content and these soils also had high clay content (Appendix 3.2). These results are consistent with those of te Beest et al. (2012)'s study at Hluhluwe-Imfolozi Park, where they found that *C. odorata* had an adaptation to clay soils that were rich in nutrients and had high organic carbon. Mandal and Joshi (2014) mentioned that *C. odorata* increases the soil organic content as well as soil nutrients which is most beneficial to its neighbouring plants. This is because *C. odorata* is able to shield the soil resulting in increased soil moisture, organic matter and biomass. and also it is able to propagate the surface soil with its roots, improving soil physical and chemical properties (Mandal and Joshi, 2014).

The results of this study show that repeated cutting of *C. odorata* over short intervals could control the spread of this weed and achieve complete eradication. In addition to that, Mugwedi et al. (2017) mentioned that repeatedly cutting *C. odorata* followed by planting a high density of fast growing pioneer plants would decrease the establishment of *C. odorata* at Buffelsdraai Landfill Site. Furthermore, it is suggested that targeting small and young infestations could lead to a successful control of this weed. Based on the current study's findings, it is suggested that the current management method (brush-cutting the stem near the soil surface (50 mm above the ground) once a year) used at Buffelsdraai Landfill Site to control *C. odorata* is not effective. Instead of cutting this plant at above 5 cm once a year, repeated cuttings at three months intervals should be implemented. This will reduce the resprouting ability of *C. odorata*, thus reducing seed production and eliminating its competitive advantage over native plants. In addition, enhancing a healthy ecosystem and increased native plant productivity.

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Chapter 4: Summary and Recommendations

4.1. Summary

The Buffelsdraai Landfill Site is under restoration through a tree planting programme that emphasises active involvement of nearby communities in propagating seedlings and planting them on site (Douwes et al., 2015). The restoration of previously degraded ecosystems plays an important role in conservation of biodiversity and improving ecosystem function (Benayas et al., 2009; Stanturf et al., 2014). Restoration also plays a key role in dealing with potential and existing threats that may slow down the recovery of the ecosystem (Ruiz-Jaen and Aide, 2005). Amongst other threats, the invasion of natural ecosystems by alien invasive plants is a known threat to biodiversity and ecosystem function (van Wilgen et al., 2008; te Beest et al., 2012). Restoration success at Buffelsdraai Landfill Site is threatened by dense infestations of invasive alien plants.

In addition to the heavy infestation at Buffelsdraai Landfill Site, *Chromolaena odorata* has become one of the most problematic invasive plants in South Africa, which highly impacts disturbed land, native biodiversity as well as grazing and crop lands (Zachariades et al., 2016; Shackleton et al., 2017). The control of *C. odorata* in South Africa has been ongoing for many years through the use of biological control agents, mechanical and chemical control methods, although successful cases are where one control method is coupled with the other for complete eradication and also frequent follow ups (Zachariades et al., 2016). Conservation efforts and restoration initiatives in invaded areas need to prioritise the management of alien invasive plants to promote successful ecosystem restoration and biodiversity enrichment (Mugwedi, 2017). This study investigated the influence of an invasive plant *Chromolaena odorata* and native grasses on native tree growth at the Buffelsdraai Landfill Site and the effectiveness of mechanical control of *C. odorata* on site.

Firstly, I investigated the effects of *C. odorata* and grass competition on the three native tree species (*Brachylaena discolour*, *Erythrina lysistemon*, *Vachellia natalitia*) in terms of their growth dynamics. The three tree species used had different growth dynamics in terms of their height, stem basal diameter and canopy length. However, I found no significant differences amongst the four *C. odorata* and grass removal treatments in terms of the measured parameters. Trees grew well in the presence and absence of *C. odorata* and grass. The results of this study did not show the competitiveness of *C. odorata* and grass on the native trees. However, there is potential for the restoration of Buffelsdraai Landfill Site after the removal of this invasive plant. Most of the trees planted on this site such as *Erythrina lysistemon*, *Brachylaena discolour* and *Vachellia natalitia* are pioneer species, which are fast growing and able to outcompete and shade out *C. odorata*, which is shade-intolerant and cannot grow under tree canopies (Mugwedi, 2017). Alien invasive plants are known to supress native plant communities. However, in this study I did not find any effect of the alien plant *C. odorata* on native trees. The use of native sub-adult trees and the 12 months duration of this study could have been the reason for these current results, showing little or no competition between *C. odorata* and native trees. Clearing invasive plants in restoration initiatives not only does it reduce resource competition with native plants but also improves species diversity and enhance canopy closure while promoting native tree species regeneration (Bullock et al., 2011).

In the second part of the study, I focused on contributing towards improving the techniques and approaches of mechanically controlling *C. odorata*. This was done by investigating the effectiveness of subjecting *C. odorata* plants to different cutting frequencies at different heights in terms of its survivorship and resprouting ability. Mechanical control of *C. odorata* at Buffelsdraai has been done by brush-cutting the stem near the soil surface (50 mm above the ground) once a year Mugwedi (2017), and this was seen to be ineffective. In this study however, cutting *C. odorata* plants at a higher frequency resulted in greater numbers of resprouting shoots while repeated follow-up cutting at a lower height decreased the number of resprouts. Rusdy (2015) reported that for effective means of controlling *C. odorata* mechanically, small infestations should be targeted and the weed should be removed entirely from the roots through hand weeding for complete eradication. In addition, targeting this invader on its earlier stage will assist in less productivity due to less stored reserves as well as less infestation from no seed dispersal

One of the ways to manage invasive alien plants is through prevention of establishment and spreading, for example, preventing disturbances leading to soil exposure for IAPs to colonise. This would end the spread of weeds such as *C. odorata* and at the same time eliminating future problems of weed infestations. At Buffelsdraai Landfill Site the control of *C. odorata* should be through repeated cutting at short time intervals of about three months with frequent follow up treatments to remove small infestations, and as showen by this study, tree mortalities were high on stumps that were cut at a lower height compared to those that were cut higher up.

4.2. Recommendations

- I recommend that restoration initiatives beset by problems of invasive alien plants should consider planting fast-growing native trees such as *Apodytes dimidiata*, *Kiggelaria africana* and other fast growing pioneer trees that would out-compete and shade out invasives such as *C. odorata*.
- Planted trees species should be monitored on the basis of their performance and in terms of which species are doing better in the presence and absence of IAPs. This monitoring not only helps to evaluate the success of restoration but also points out the failures that would need attention.
- At the Buffelsdraai Landfill Site, instead of cutting *C. odorata* once a year, repeated cuttings at a height of 30 mm above the ground of this species is needed within short intervals of 3 about months (especially during the wet season because the stumps can easily grow back) as it reduces survivorship and the number of resprouts and seed production.
- In areas of small *C. odorata* infestations, I recommend that the control of this species is through hand-pulling as it reduces the necessity of follow-up operations.
- In areas where dense *C. odorata* populations have been cut, re-establishment through active restoration of native trees should be implemented to reduce regrowth and re-emergence of this species at the same time reducing soil exposure and the risk of soil erosion.
- Future research is needed to investigate the effectiveness and benefits of a long-term *C. odorata* management programme.
- A combination of mechanical control with other methods such as chemical should also be investigated as it may yield effective and more economical results.
- There should be an initiative such as capacity building workshops and skills development workshops where the community people involved in the project are equipped with skills to manage and control alien invasive plants on site.

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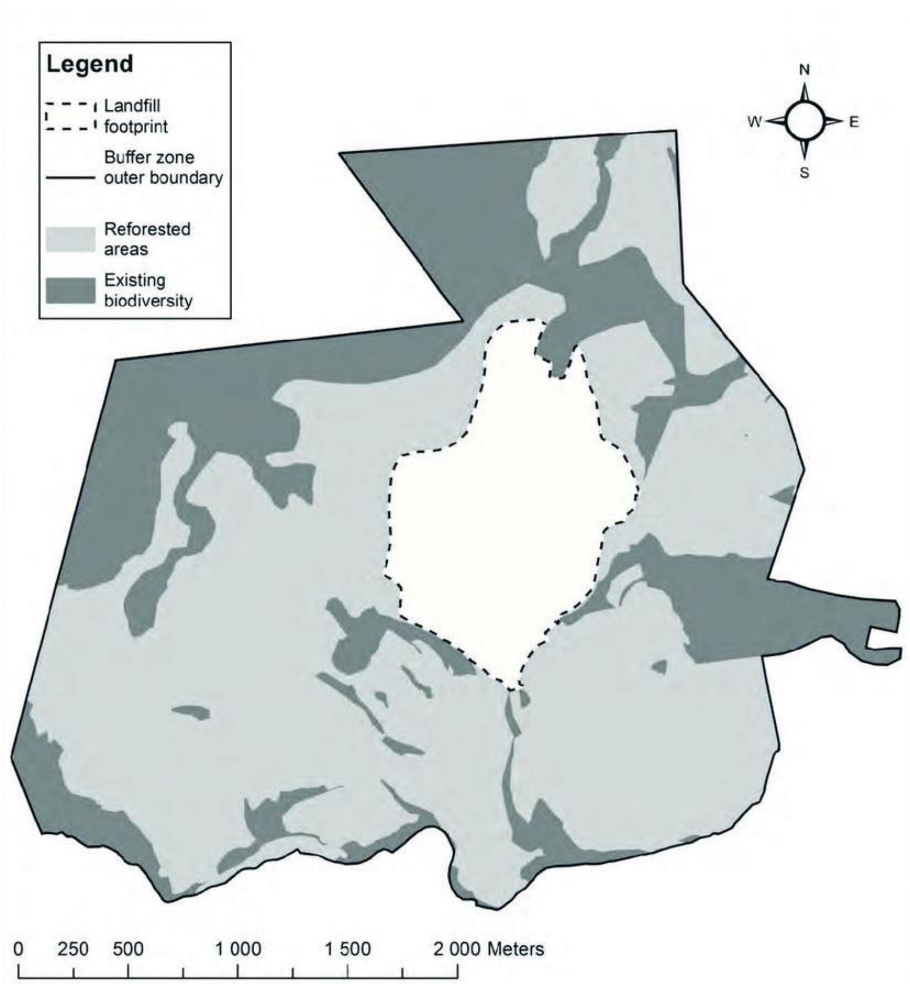
Appendices



Appendix 3. 1. Map of Buffelsdraai Landfill Site.

Appendix 3. 2. Mean ± SE of soil physical and chemical properties based on soil samples collected from 0-10 cm, 10-20 cm and 20-60 cm depths.

	Soil depth		
Soil properties	0-10 cm	10-20 cm	20-60 cm
Soil pH			
Soil pH (KCL)	4.62 ± 0.26	4.5 ± 0.27	4.68 ± 0.25
Soil pH (H_2O)	6.33 ± 0.072	6.52 ± 0.078	6.92 ± 0.19
Soil chemical properties			
P (mg/L)	5 ± 2	4.33 ± 0.33	4 ± 0.58

K (mg/L)	215.67 ± 17.17	143 ± 4.04	109 ± 4.1
Ca (mg/L)	1480.33 ± 241.18	1541.67 ± 208.62	1420 ± 38.44
Mg (mg/L)	929.67 ± 206.40	921.33 ± 182.25	852.33 ± 138.3
N (kg/g)	1.90 ± 0.23	1.83 ± 0.23	1.77 ± 0.41
Soil physical properties			
% silt	38.29 ± 1.14	35.82 ± 1.32	34.29 ± 1.72
% sand	29.29 ± 4.23	27.35 ± 5.85	21.95 ± 4.38
% clay	37.3 ± 4.06	39.02 ± 4.84	44.57 ± 6.86