University of Venda

An inventory of woody and herbaceous invasive alien plants in Thohoyandou Botanical Garden, Limpopo Province, South Africa

By

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In the

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Declaration

I, Mboneni Neluonde, declare that this dissertation is my original work and has not been submitted for any degree at any other university or institution. The dissertation does not contain other persons' writing unless specifically acknowledged and referenced accordingly.

Signed (Student): Neluonde M Date: 07/10/2021

Dedication

I dedicate this dissertation to God Almighty my Creator, who has been the source of my strength throughout my studies. I also dedicate this dissertation to my entire family for supporting me throughout my study. To my friend Melford Mbedzi who has been there for me in terms of emotional support when I felt like giving up and who believed in me that I can make it, this dissertation is also dedicated to you.

Thank you.

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Abstract

Species migration out of their local range into new zones by people has brought about the rise of biological invasions. Exotic or non-native species need to conquer various boundaries to establish, naturalize, produce localized self-supporting populations, and in the end spread naturally before they are considered as invasive. These invasive alien species are currently perceived as key drivers of human's instigated global change as they have negative impacts on biodiversity, ecosystem services and human prosperity which makes the investigations on invasive species imperative to direct approach plan and management important. This study was motivated by the need to control invasive alien plants in protected areas and this inventory of invasive alien plants (IAP) marks the first step in creating baseline data for managing invasive alien plants in the garden.

Protected areas are threatened by the impacts of invasive alien plants that invade them and Thohoyandou Botanical Garden is no exception. Invasive alien plants are the second greatest threat to biodiversity worldwide and estimates suggest that invasive alien species cost the country over R 6.5 billion per year. The study aimed to document the woody and herbaceous invasive alien plant species in Thohoyandou Botanical Garden in the Vhembe District. Three belts of 100 m x 500 m were constructed and within them quadrats of 10 m x 10 m were randomly constructed. All the invasive alien plant species were identified and grouped as per their growth forms. Their impacts on the adjacent native species were investigated and their health status identified using a sliding scale of 1 – 5, where 5 is healthy and 1 is unhealthy. All the data were stored in Microsoft Excel and later analyzed using One-way analysis of variance(ANOVA) was also used to compare mean frequency count and mean height between invasive and native species. Data was analysed through the Microsoft Excel programme.

This study has shown that invasive species have the potential to reduce native species at a rate of 2.92 species per every 40 invasive species, on average. Thus, for every 40 invasive species that are found in any chosen area within Thohoyandou Botanical Garden, an average of 3 native species will be lost as a result. These results have a very strong implication in that they can guide management and control strategies aimed at ensuring that the impact of invasive species on the growth and survival of native is kept to minimal levels.

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List of Acronyms

ANOVA Analysis of Variance

CSR Competitors, Stress-tolerant and Ruderals

HIV Human Immunodeficiency Virus

IAP Invasive Alien Plants

IAS Invasive Alien Species

NPS National Park Services

R^2 Coefficient of Determination

USA United State of America

IUCN International Union for Conservation of Nature

GISD Global Invasive Species Database

Chapter one

INTRODUCTION

1.1 Background to the study

A vast wealth of knowledge has been accumulated in the field of alien species management since the SCOPE (Scientific Committee on Problems in the Environment) programme of the 1980s (Drake *et al*., 1989) asked the questions: which species invade; which habitats are invaded; and how can we control invasions? However, the number of alien species on all continents continues to increase. In many cases this increase has been much faster than historical invasion patterns (Groves and Hosking, 1998). Such increases place further demands on already limited resources, outpace research, and in many cases limit control options. In addition, many alien species introductions are irreversible, with eradication only feasible for new invasions (Rejmánek, 2001) or under special circumstances (Myers *et al*., 2000). While there have been a few successful control programs carried out on islands (Veitch and Clout, 2002), less successes have occurred on continents or for widespread alien species. If the invasion of alien species cannot be halted or established invasions eradicated, then we need to understand how to limit their spread, proliferation and impacts on invaded ecosystems. Such knowledge is needed to further develop management strategies and monitoring protocols for protected areas, which are often the last bastion for native species or populations.

Alien plants were first recorded during the initial botanical surveys of the Kruger National Park (KNP) in 1937 (Obermeijer, 1937). The highly complex biophysical context of SANParks makes it difficult to manage Invasive Alien Species (IAS) in their PAs, for instance the huge area, the number of parks, the distribution in South African biomes and the degree of their invasion make planning a challenge. when, where and how measures are to be implemented, therefore they should be prioritized according to the available resources (Forsyth *et al*., 2012), in most parks and in key areas within each park (Forsyth and Le Maitre, 2011), nonetheless current funding processes make prioritization difficult. (RouraPascual *et al*., 2009) Tough decisions must be taken to balance the benefits with the losses for different biomes and species, while accepting the fact that other negative impacts are unavoidable in certain localities or parks. Important criteria to consider in SANParks and recommended species to be managed (Forsyth and Le Maitre, 2011) Although invasive plants can be managed due to the species and biomes they are found in, other taxa such as fish cannot be settled. that pose significant threats to ecosystems and native species. The management strategy development process requires two sources of information, precise distribution data and species list. (Pyšek *et al*., 2012).

Researchers and wildlife managers express those forestalling new presentations of outsider species is the best method to decrease organic attacks (Olszańska *et al.,* 2016). However, counteraction isn't constantly fruitful, and control measures must be taken as the second line of safeguard after establishment. Dubious measures including “shoot first and pose inquiries later" approach regularly is still the main practicable option (Brown *et al.,* 2008). While radical removal measures have demonstrated effectiveness, they are mostly not used, because of ethical concerns (Van der Hoeven *et al.,* 2015).

1.1.1 Conservation concern

Invasions by alien plants are a significant conservation concern in numerous regions of the world (Mack *et al.,* 2000), South Africa included (Richardson and Rejmánek, 2011). Urban regions are hotspots for the introduction of invasive species (Vitousek *et al.,* 1997). Especially of flora utilized for ornamental cultivation (Reichard and White, 2001). Not surprisingly there is a strong connection between human population density and invasive plant richness (Spear *et al.,* 2013). Urbanization is expanding in all parts of the World and the greater part of the worldwide human population currently lives in urban areas and cities (UNAIDS, 2014). This pattern is probably going to increase into the future (Grimm *et al.,* 2008).

While increasing urbanization is probably going to fuel issues related with urban communities as sources of alien species propagules, historical pattern and procedures imply that there is as of now a huge invasion debt even without any more introductions, numerous species already introduced will get to be invasive after some time (Rouget *et al.,* 2016). Regardless of these discoveries and the undeniable risks, not many investigations have examined the structure and patterns of invasive alien plants inside urban spaces. Those that have been done have focussed on bigger urban communities (Lenda *et al.,* 2014) While big urban areas regularly have more alien species than small rural areas and towns and are frequently main localities in a nation to which an alien plant is introduced, smaller towns normally have a generally bigger urban wildland interface.

1.1.2 Invasive alien species

Attacks by invasive alien species are one of the major threats to the stability of marine vegetation and the services of the ecosystem (Westphal *et al.,* 2008). Encouraged by worldwide exchange the quantity of invasive alien species that were inadvertently or intentionally brought into new districts has increased quickly in recent decades (Pysek *et al.,* 2012). Invasive alien species from all major animal and algal phyla have built up in most coastal areas worldwide with genuine ramifications for marine environments (Katsanevakis *et al.,* 2014). Marine ecosystems are particularly susceptible to biological invasions since it is estimated that whenever a few thousand marine species are easily dispersed between biogeographic locales. The greater parts of the marine organisms are moved on transport structures or in the boat's ballast water. Other transportation ways incorporate aquaculture purposes, angling rigging or aquarium exchange (Schaffelke *et al.,* 2006). Invasive alien species can cause huge financial and ecological harm, even though the effect of invasive species is rarely predictable (Schaffelke and Hewitt, 2007).

Among plant species, invasive alien species are perceived as those that quickly widen their spatial distribution by venturing into existing indigenous plant vegetation (Richardson *et al.,* 2000). Even though invasions start from natural or human initiated dispersal forms, a scope of biological and physical components can give the chance to invasive alien plants to quickly outcompete indigenous species (Rejmanek *et al.,* 2005).

The success of invasive alien plants is frequently credited to their ability for fast development via high resource accumulation, especially in non-resource restricted conditions (Pyšek and Richardson, 2008). Leishman *et al.* (2010), indicated that invasive alien flora and indigenous species lack a fundamentally different carbon capture strategies, supporting the idea of community invasibility, for example, attributes of the indigenous communities preferring invasion (Pyšek and Richardson, 2008). As indicated by trait-environment connections (Dalle Fratte *et al.,* 2019), the contrasts between invasive alien plants and indigenous plants can shpw environmental conditions of the regions where they occur as opposed to contrasts between species essentially (Leishman *et al.,* 2010).

Pyšek and Richardson (2008), evaluated literature trying to distinguish qualities and attributes of invasive vascular plant species. Proof was discovered just for certain characteristics that are generally associated with these species: stature, vigorous vegetative development, early and extended blossoming. Van Kleunen *et al.* (2015) showed that invasive alien plants outcompete indigenous species because of higher functional values qualities related to development rate and resource acquisition.

Additionally, consistent with a 'delicate leaves, quick development' strategy, the best invasive species limit carbon investment in leaf development, as observed both on land (Tordoni *et al.,* 2019) and aquatic vascular plants (Lukacs *et al.,* 2017). However, the discussion concerning indigenous versus invasive suites
of qualities is progressing since patterns or basic features are not always clear. Daehler (2003) recommended that there is no different set of characteristics liable for invasiveness, yet that various suites of attributes could clarify invasion achievement in various conditions (Tecco *et al.,* 2010).

Numerous adaptive characteristics can be summarized in terms of the ecological procedures, or the group of functional characteristics engaged with plant resource economics (deciding issue and energy turnover rates) and size (signifying the amount of matter and energy present inside organisms and inside biotic communities) (Pierce *et al.,* 2017). Grime's CSR theory of life provides a theoretical framework for organizing species into competitive (C), stress-tolerant (S) and ruderal (R) strategies (Grime and Pierce, 2012). As of now the main plant strategy theory that concurs with resource economics and size as the chief axes of adaptive variety is to put these with regards to environmental selection pressures (Pierce and Cerabolini, 2018).

1.1.3 Strategy theory

There is a scope of favourable circumstances in utilizing adaptive strategy theory, instead of single characteristics, to assess the interactions among invasive aliens and indigenous species (Guo *et al.,* 2018) and the consequences on ecosystem services provision because of plant invasion (Vicente *et al.,* 2013). The use of CSR strategy theory to investigations at a local to regional scale has proved that among invasive alien plant species, R-and C-selected species and their transitional strategies (CR) are pervasive, while S-selected species are under-represented (Dalle Fratte *et al.,* 2019b). Notwithstanding this example, also being apparent at the global scale brought up certain issues concerning the utilization of CSR theory to order invasive species, since it might mask individual characteristic differences (Guo *et al.,* 2018).

Though CSR strategies can conceivably give a strong hypothetical setting to predict the performance of species in a definite ecosystem (Grime and Pierce, 2012), it is important to experimentally validate the potential for CSR strategy theory to separate invasive alien species and to foresee their capacity to invade.

Small towns are also substantially many in numbers than larger urban areas thus collectively represent a higher threat of contributing invasive alien propagules into the encompassing regions. South Africa has instituted national legislation aimed at eradicating invasive alien species which has implications for the urban environment. In any case, most districts don't have the capacity to support the necessities of National Environmental Management: Biodiversity Act (NEM:BA, Act No. 10 of 2004), (Irlich *et al.,* 2017). While some data is accessible at a wide environmental scale on the presence and general location of invasive alien plant species outside of agricultural sites that will help municipalities in putting together their plans (Henderson and Wilson, 2017), there is next to no data on the area, identity, and spread of invasive alien plants in the urban spaces in the country.

Invasive alien species are turning into a certain negative impact to the public parks and protected zones of the US. In the US, the National Park Services (NPS) adopts the management strategies depending on preserving the natural integrity of explicit parks and ensuring ecosystem structures and functions. The NPS policy expresses that, "high priority will be given to overseeing alien species that have, or possibly could have, a considerable effect on park resources, and that can sensibly be relied upon to be effectively controlled. Where an alien species cannot be effectively removed, managers will seek to contain the alien species to prevent any further spread or resource damage" (NPS, 2006).

Despite the undeniable benefits (such as, timber, and medicinal) of invasive alien species, the management, park programs intended to control invasive alien species are regularly viewed to be unfavourable by the public. Numerous individuals contradict invasive alien species management since it can include activities for example, lethal removal that individuals find distasteful (Temple, 1990). Extreme adversaries have cited nativism or xenophobic thought processes in invasive species control (Simberloff, 2003).

Choices to remove invasive alien species have created differences because the general population's perspectives and observations about nature every now and again contrast from those of ardent conservationists and natural resource managers (Minteer and Collins, 2005). Eradication of invasive alien species that people view as useful to their visit has demonstrated to be particularly troublesome. For instance, in 1982 feral horses on Assateague Island National Seashore, Maryland, were considered an alluring item and managed as a historically and culturally significant wildlife species regardless of their impeding effects on island ecosystems (Sturm, 2008).

A general lack of comprehension of the management goals and objectives can likewise prompt disarray (McNeely, 2001). Although the decrease in harm brought about by invasive alien species is a definitive objective, numerous public stakeholders see the ideal result to be a general decrease in the quantity of invasive species – an objective that is not really feasible or practical (Lodge and Shredder Frechette, 2003). These differences show that invasive species control issues can be progressively political, economic, and cultural than biological (Tanentzap *et al.,* 2009). Subsequently, the human component of the invasive species problem ought not be disregarded or thought little of. Truth be told, numerous environmental managers refer to an absence of public awareness and support as one of the significant obstructions to effective invasive alien species based on the managers (Andreu *et al.,* 2009). Other investigators noticed that ecosystem-based control methods endure when logical arguments fail to reverberate with policy makers and the more extensive public (Tanentzap *et al.,* 2009).

1.1.4 Impacts of invasive alien species

The recorded impacts of invasive alien species are many, and incorporate direct dangers to human wellbeing, and loss or modification of goods and services with respect to fishery, farming, forestry service, drinking water, hydrology, climate stabilization, fertilization, culture, and recreation. Different effects are subtler and are the aftereffect of long-term changes influencing habitat and ecosystem functioning (Hulme, 2007). These modifications incorporate nutrient cycling (Chamier *et al.,* 2012), soil properties disturbance, regimes fire recurrence (Brooks *et al.,* 2004), trophic associations and biodiversity loss (Levin *et al*., 2006).

Specifically, invasive alien species can threaten biodiversity in different manners, from decreasing hereditary variety and dissolving gene pools to the extinction of endemic species, particularly in islands and freshwater biological systems. The expenses related with the effects of invasive alien species are consequently colossal, and amount annually to many billion dollars around the world. Pimentel *et al.* (2005) estimated that the damage and management of invasive alien species has an economic effect of 120 billion dollars annually in the USA alone. Essentially, in Europe, lost yield because of invasive alien species, wellbeing effects and expenditures to fix invasive alien species damage has cost the European Union at least 12 billion euros for each year in recent years (LIFE Guidelines, 2014).

Working for Water (WfW) has spent R15 billion, since 1995, on alien plant control programmes throughout South Africa. Since 2010, the amount has been increasing exponentially per year reaching around ZAR 2 billion per year in 2017. WfW has been able to clear an average of about 200,000 condensed ha per year and conducted follow-up monitoring on over 600,000 ha per annum. The clear reduction in treated area since 2014 is due to a relaxation of the obligation to register the treated area, which essentially leads to an underestimation of the most recent figures other groups that have been given large sums of money are *Lantana camara* (Lantana), trees of the

Genera *Prosopis* and *Eucalyptus* and *Chromolaena odorata* (Triffid Weed). to clean up invasive alien plants (van Wilgen and Wannenburgh 2016); The program has created between 2,000 and 23,000 full-time positions per year.

In principle, if invasive alien species and even alien species do not cause significant damage, but only cause change, then changing the perspective is undoubtedly much cheaper than any other type of control measure. a piece of nature in pristine condition before the invasion, as the natural environment is an integral part of a community's cultural heritage (Hershner and Havens, 2008).

1.2 Aim and objectives

The aim of the study was to document the woody and herbaceous invasive alien plant species in the Thohoyandou Botanical Garden. To achieve the aim the following objectives were investigated:

i. To identify and list woody herbaceous alien plants present in the Thohoyandou Botanical Garden

ii. To investigate the impact of invasive alien woody and herbaceous plants on indigenous species in the Thohoyandou Botanical Garden

1.3 Problem statement and rationale

Protected areas are threatened by the impacts of invasive alien plants that invade them and Thohoyandou Botanical Garden is no exception. Invasive alien species (IAS) are the second greatest threat to biodiversity worldwide and estimates suggest that invasive alien species cost the country over R 6.5 billion per year, Biodiversity Act 10 of 2004 (NEMBA). This proposed project aims to identify and list woody and herbaceous alien plants present in Thohoyandou Botanical Garden. Also, current, and future distribution maps of woody and herbaceous alien plants present in the garden will be presented.

The proposed study is motivated by the need to control alien plants in protected areas and this inventory of invasive alien plants (IAP) marks the first step in creating baseline data for controlling invasive alien plants in the garden. Outcomes of the study will include the compilation of an invasive alien plants list, specimens for herbarium archiving and baseline distribution of information in the garden. This is crucial for enabling effective monitoring of current and future invasive alien plants distribution and detects new introductions in Thohoyandou Botanical Garden.

1.4 Hypotheses

The study tested the following hypotheses:

i. Thohoyandou Botanical Garden is highly infested with invasive alien plant species.

ii. Invasive alien plant species have devastating impacts (Such as, out competing them for space and other natural resource and thereby displacing them) on the indigenous plant species growing in the Thohoyandou Botanical Garden.

1.5 Dissertation structure

Chapter 1 dealt with the background of the study where the introduction of invasive alien plants was discussed followed by conservation concerns on native species. Chapter 2 literature review focused on the invasive alien species management, impacts of invasive species on abiotic variables.

Chapter 3 focused on the study area, the vegetation of Thulamela Municipality and its environmental factors and methods used to conduct the research.

Chapter 4 focused on the results which were transferred into Microsoft Excel, and they were analysed. Frequency distribution tables and graphs were used to interpret the summarised data.

Chapter 5 discussed the results on the impact of invasive alien species on native species.

Chapter 6 dealt with conclusions of the study and recommendations on what must happen in the next coming years at the botanical garden if we still want to see native species not getting extinct.

Chapter two

LITERATURE REVIEW

2.1 Invasive alien species management

Invasive alien species management is a control that despite everything battles with a one-sided approach and methodological constraints with regards to surveying the genuine effects of invasive alien species on native plant species (Mack and D'Antonio, 1998). The future impacts of invasive alien species are uncertain on the grounds that biotic associations are incredibly hard to foresee, particularly in a quick evolving world (Alexander *et al.,* 2016). These uncertainties have made a lot take for granted the negative effects of invasive alien species and to neglect conceivable commitments or even positive effects. Invalid assumptions and confusing arrangements can slow the progress of control and cause unrecoverable financial losses and irreversible ecological changes. Surveying invasive alien species impacts is drastically risky for the absence of well-defined characterized measurements (Sanz-Elorza *et al.,* 2008).

In addition, the complete restoration of indigenous biodiversity ought not to be the driving standard in ecological restoration, which ought to rather depend on species functional roles as opposed to taxonomic considerations. However, numerous conservationists, despite everything, consider the indigenous-versus-alien species dichotomy a central core value in ecological restoration (Minteer and Collins, 2005). Invasive alien species management can step forward if all stakeholders understand that invasive alien species cause changes however, not real damage. Invasive alien species are symptoms of indigenous habitats that are changing perpetually because of drivers, for example, climate change, nitrogen eutrophication, expanded urbanization and other land-use changes (Aronson *et al.,* 2014). To distinguish these tipping focuses, researchers, environmental managers, and policy makers should grasp an increasingly realistic methodology that stays away from stubborn thought, advances a dialogue with residents, and utilizes ecosystem services as measurements for evaluating invasive alien species impacts (Chamier *et al.,* 2012). The policy of South Africa's national parks is to phase out all exotic plants from staff and tourist facilities, in favor of native species. This will take time and will require not only systematic management programs to eliminate existing invasive populations, but also interventions to manage the pathways of introduction (Foxcroft et al., 2019), and the creation of buffer zones around the park (Foxcroft *et al*., 2011).

The South African National Working for Water Programme (WFW) is one of the largest alien plant control initiatives to be employed, worldwide (van Wilgen, 2004). Established in 1995, the Programme's aim is to control alien plants in watersheds to restore water flows, while creating employment for many South Africa's unemployed. The people employed in this programme are given training both in the control of alien plants and their impacts as well as other life skills. The enormous benefit of the Programme is clear in many areas of South Africa (van Wilgen, 2004).

Working for Water provides approximately R20-30 million per annum to SANParks for the control of invasive alien plants across all 20 national parks in South Africa. A special unit within SANParks implements the WFW programme nationally (Foxcroft *et al*., 2008). The annual funding provided to the KNP averages R5 million per annum, which employs up to 500 people and has seen many of the rivers, for example, the Sabie River, which were previously heavily invaded by alien plants, now at a maintenance level.

2.2 Impact of invasive species on abiotic variables

Producing a lot of nutrient rich litter is a known effect of intrusive acacias (Le Maitre *et al.,* 2011). A lot of litter can therefore prevent the recuperation of indigenous vegetation where it is consumed by fire in summer. The time needed for vegetation recuperation could be longer because of sweltering, harming fires (Holmes *et al.,* 2000). Shockingly, zones with one and two patterns of pine turn and all acacia invaded plots had lower measures of litter than the reference plots. This could be because of the way that the reference region is a mature vegetation (20 years post fire age) having gathered a lot of litter and senescent plant material (van Wilgen, 1982).

Thick layers of litter can also prevent germination of native seeds by insulating the soil from heat or acting as a physical barrier to seedling emergence (Friedman *et al*. 1996). Litter has been linked to increased levels of nutrients in the soil, where litter with high levels of phosphorus and nitrogen could increase the release of phosphorus to the soil (Stock and Lewis, 1986).

2.3 Invasive alien species and ecological factors

It is inevitable that invasive alien species determine an ecological effect on their host ecosystems, and such effects suggest changes. Be that as it may, should such biological changes be viewed as unsafe? Also, if indeed, to what degree? Unfortunately, most investigations have failed to give an answer. One usually perceived criticality to invasive alien species management is the troublesome identification and evaluation of the reaction metrics. However, the genuine basic deterrent that meddles with invasive alien species management is that with regards to eradication, choices are still to a great extent upheld by rare scientific proof (Yelenik *et al.,* 2004).

The alleged biological impacts of invasive alien species are, in reality, frequently underestimated and blended in the mantra "invasive alien species are the second reason for species extinction". Invasive alien species may, now and again, even give desirable capacities already depending on indigenous species. The instance of shrub eradication in the USA is exemplary. Salt cedar bushes *(Tamarix spp*.) were presented from Eurasia and Africa in the nineteenth century as ornamental species and later as shade trees for desert farmers (Iososova *et al.,* 2011).

However, during the 1930s, shrubs were blamed for devastating water supplies in dry lands, and from that point forward has become the objective of eradication campaigns (Chew, 2009). Afterward, scientists found that in addition to the fact that shrubs use an amount of water like that of their indigenous counterparts yet also they give nesting territories to the indigenous endangered southwestern willow flycatcher (*Empidonax traillii extimus*) (Stromberg *et al.,* 2009). Besides, tamarisks are drought, salt, and erosion-resistant plants able to survive under normal water management regimes that destroy indigenous vegetation and have a significant role in the working of man-modified riparian biological systems (Aukema *et al.,* 2010).

2.4 Effects of invasive species on vegetation structure, practical organization and species lavishness recuperation post-outsider clearing

Past investigations by different authors found that plots dominated by pines recovered fundamentally (as far as indigenous species spread) to levels near the vegetation in reference plots, though plots dominated by acacia did not. An absence of post-clearing recuperation in terms of indigenous vegetation spread, changes in vegetation structure and species richness following eradication of invasive acacias has been found in different investigations in the fynbos (Galatowitsch and Richardson, 2005).

The successful recuperation of native species spread, native guilds and generally native perennial richness in pine plots could be because of the persistence of the indigenous seed bank underneath the canopy (Heelemann *et al.,* 2013). Native vegetation can endure underneath pine shades for quite a while since it can take 13 years for complete pine canopy closure, allowing indigenous species a chance to establish and renew seed banks (Holmes and Marais, 2000). As far as functional guilds, ericoid bushes were found not to recoup to a similar level as reference plots.

2.5 Pathways of the invasive alien plants

The pathways of introducing invasive alien species with new zones are connected to anthropogenic activities worldwide (Thruiller *et al.,* 2006). This is consistent with speculations that relate urbanization and other human exercises to more elevated levels of invasive alien propagule pressure (Carboni, *et al.,* 2010). Understanding the systems clarifying how invasive species are spatially and naturally disseminated has become a focal research issue in environment and evolutionary biology, because of its close connection to the success of invasion (Alpert *et al.,* 2000).

A few researchers have addressed the importance of environmental (Marco *et al.,* 2011) or geographic distances (Minor *et al.,* 2009) in species structure dissimilarities (Keil *et al.,* 2012). Yet scarcely any contrasting the overall impacts of the two kinds of distances on patterns of (dis)similarity in invasive alien species richness and composition (Lososová *et al.,* 2011).

While assessing the significance of distances to clarify distributions of alien invaders, geographic separation between areas may assume a more significant role than environmental separation, not at all like what could be expected for indigenous biodiversity for certain scales (Li *et al.,* 2011). Undoubtedly, time since acquaintance and species attributes related with dispersal (for instance, seed or fruit size, seed or fruit resistance and life span, dispersal mechanisms) can mutually compel the capacity of invaders to colonize their potential climatic range in the recently invaded territory (Alpert *et al.,* 2000). In addition, it has been shown that habitat connectivity and the presence of man-made connecting infrastructure (for instance, roads) or different corridors (for instance, riverbanks) offer preferential courses for the spread of invasive alien species across the landscape or area (Minor *et al.,* 2009).

The expectation of biological invasions can concentrate either on attributes that favour species invasiveness or on features of the beneficiary environment that impact community or landscape invasibility (Vicente, 2010). One way to deal with surveying the environmental role (related to invasibility), and dispersal (identified with invasiveness) in controlling biological invasions consists of adapting distribution models that relate the variety of invasive alien species to environmental indicators, and to represent potential dispersal vectors (Thruiller *et al.,* 2006). During the most recent two decades, the utilization of such empirical statistical models has expanded in ecology to foresee the geographic dissemination of species and diversity measures, yet they can likewise be utilized for testing theories on the role of various environmental predictors (Broennimann and Guisan, 2008).

Invasibility and invasiveness, two key components of invasion, are believed to be principally controlled by environmental suitability and by propagule pressure and dispersal, respectively (Vicente *et al.,* 2014). Subsequently, invasiveness is required to be interceded by life strategies (for instance, by intrinsic life-history or species attributes (Castro-Díez, *et al.,* 2011) though invasibility is progressively related to local conditions at the site, habitat, or landscape levels (Vicente, 2010). For vascular plants, most utilitarian characterizations have been proposed over recent decades, considering life strategies, growth forms and reproductive strategies, permitting a wide range of groups to be tried in invasion biology.

With regards to plant invasion, the integrative intensity of the "C-S-R utilitarian mark” (that is, the relative abundance of competitors (C), Stress tolerant (S) and Ruderals (R) in each species pool; Grime, 1977) can be utilized to perceive community procedures, for example, obstruction, flexibility, eutrophication, and dereliction (Hunt *et al.,* 2004). Environmental conditions and dispersal constraints are known to shape the patterns of indigenous biodiversity at the regional scale, with ecological heterogeneity affecting species richness and dispersal (Ferrier *et al.,* 2007).

This general pattern is an after effect of a long history of collaborations among species and among these and their environment. In any case, the equivalent could in the long run not be valid for the patterns of invasive alien species in a given area since: (i) invasive alien species vary in the genuine attributes driving their invasiveness in the new region, (ii) their habitation time is very heterogeneous and frequently short (that is, few years or decades), (iii) their entire living arrangement time has corresponded with human-driven ecological heterogeneity and changes, (iv) different introductions (in reality) may deliver complex patterns of expansion and distribution, and (v) numerous association with indigenous species may additionally instigate complex patterns of invasion (Van Kleunen *et al.,* 2010).

2.6 Invasive alien species and health factors

The impacts of invasive alien species on human well-being are complex and may cause considerable expenses for avoidance and treatment. Mazza *et al.* (2014)

distinguished four classifications of hazards to human wellbeing related with invasive alien species: (1) species causing illnesses or infections; (2) exposing people to wounds from nibbles/stings, bio toxins, allergens, or toxicants; (3) favouring sicknesses, wounds, or death; (4) and delivering other adverse consequences for human livelihood. Invasive pathogens are commonly liable for the most genuine illnesses, and obtrusive plants cause essentially hypersensitivities and skin irritations though animal species act as principal vectors of ailments and diseases (Wolfe *et al.,* 2007). With respect to plant and animal invasive species, wellbeing concerns might be annoying however commonly not worrying.

Be that as it may, the indirect impacts of some invasive alien species on health may demonstrate to be very serious. For example, the water hyacinth, *Eichhornia crassipes*, a South American freshwater ornamental plant included for the list of the main 100 invasive species around the world, can host mosquitoes and snail species that are vectors of pathogens preferring the spread of deadly diseases, for example, schistosomiasis (Plummer, 2005). Aside from spreading pathogens and parasites, invasive alien species affect human wellbeing through different mechanisms (Holmes and Marais, 2000).

Some invasive alien plants, for instance, lessen protection against common risks, for example, fire and flooding: the former by replacing less inflammable vegetation and the last by deterring waterways (GISD, 2015). Invasive alien species may also cause malnutrition by diminishing harvests yields with financial losses adding up to several US$ billions (Gherardi, 2007).

Invasive alien species management is a top need when human wellbeing is in question, particularly on the off chance that we consider that invasive alien species impacts on human wellbeing are required to exacerbate because of the expanded chances of invasion supported by new ways of introductions, environmental degradation, and climate changes (Bergh and Linder, 2009). Contrasted with social, cultural, and ecological impacts, invasive species sway on human wellbeing can be distinguished more quickly and surveyed legitimately and unbiasedly. Specifically, because the fundamental measurement included, the quantity of influenced individuals is very much characterized and quantifiable. The quantity of infections and deaths brought about by invasive alien species gives a clear idea of the magnitude of disease (Grime, 2002).

2.7 Invasive alien species and economic factors

The financial evaluation of invasive alien species impact considers essentially the negative results without including any potential constructive outcome as a trade-off that may total economic effect of invasive alien species adjusted or even positive. Financial benefits obtained from invasive alien species, for example, fertilization by non-native honeybees or expenses paid to hunt and fish non-native species (Pascual *et al.,* 2009).

The outcomes of these gaps are hard to measure, however unquestionably may have favoured mismanagement of financial resources when it came to invasive alien species eradication. Aside from the barely recorded invasive alien species benefits, studies utilize moderately basic strategies for surveying damage expenses, and inability to utilize efficient empirical cost estimation techniques and the recounted idea of their assessments are also observed as significant imperfections (Lovell *et al.,* 2006).

Another criticism concerns the way that invasive alien species financial assessment fails to incorporate impacts on ecosystem services. Assessing the effects of invasive alien species on ecosystem services has only become an explicit goal of invasion biology (Richardson and van Wilgen, 2004), and some ecosystem services are certainly better understood than others. For instance, the effects of invasive alien species on provisioning services (nourishment, fibre, and fuel) are generally very much evaluated. Effects on other life support services, such as new water and most regulating services (for example, pollination, disease and pest regulation, flood and fire control, microclimatic attenuation) are sometimes determined. Finally, the association between invasive alien species and culture is likely one of the most mind boggling and less studied aspects of ecosystem services (Hoagland and Jin, 2006).

2.8 Perspectives

Invasive alien species eradication is a dubious issue where economic, socio-cultural, and biological factors regularly conflict in the field of ecological management. Likely, some people would question endeavours by society of managing, or even killing, invasive alien species causing health concerns, economic loss, and damage to fundamental ecosystem services. Although help would likely not be as solid, many would likewise urge endeavours to manage or remove those invasive alien species that are truly compromising indigenous species with imminent extinction. In any case, it might be authentic to scrutinize the control and management endeavours involving invasive alien species that cause other undesirable environmental impacts (Ackefors, 1999).

These would incorporate species that modify the composition of historical communities, change disturbance systems, and influence environmental processes, but that are not compromising our wellbeing, economies, ecological services, or causing different kinds of great biological damage, for example, driving different species to extinction (Davis, 2009).

In general, by holding fast firmly to indigenous preference, we are bound to an existence of dissatisfaction and frustration, since we will probably be encompassed by increasingly more exotic species. The introduced species are not returning, and like it or not, they are our new occupants, regardless of what we decide to call them. Invasive alien species management is commonly influenced by a one-sided approach bringing about the common practice of actualizing removal endeavours without cost-benefit analysis bolstered by comprehensive scientific proof (Gherardi, 2007).

The unavoidable consequence is several non-native organisms targeted for management with basically no effect on the invaded biological systems *(*Meffin *et al.,* 2010). If wellbeing and financial concerns are primary purposes behind invasive alien species eradication, it is additionally evident that evaluating the size of their effect may prove to be simpler due to the quantifiable parameters generally considered, for example, number of infected individuals and money lost respectively (Hejda and Pyšek, 2006).

With regards to evaluating socio-cultural and ecological effects, rather, invasive alien species management battles with ill-defined metrics and methodological constraints. As a response to these vulnerabilities, logic and good judgment ought to be considered as supporting standards in any reclamation exertion. Managers should accept some invasive alien species as permanent members of communities, and invasive alien species providing important ecological services may be the most likely candidate for acceptance (Hershner and Havens, 2008). expensive compared to any other type of management action. However, society has every right to seek to restore or maintain part of nature in pristine “pre-invasion” state, as the natural environment is an integral part of a community's cultural heritage. (Hershner and Harvens, 2008).

Chapter Three

STUDY AREA, MATERIALS AND METHODS

3.1 Description of the study area

The study was conducted at Thohoyandou Botanical Garden located in Thulamela Local Municipality (Figure 3.1). It is one of the four local municipalities that make up the Vhembe District Municipality of Limpopo Province in South Africa.

Figure 3.1: Map of the Vhembe District, showing Thulamela Local Municipality and the Thohoyandou Botanical Garden.

The area is mainly covered by soils derived from quartzite and sandstone, which are generally shallow, gravelly, well-drained with few nutrients and acidic in nature (Mucina and Rutherford, 2006). The general types of soil within the municipality are sandy soils, loamy sands, loamy soils, and loamy soils found in the river valley.

The municipality of Thulamela is classified as a subtropical climate, with most of the rains falling during the summer months from October to March. The average annual rainfall for the Municipality of Thulamela varies between 400 mm (north and northeast section) and more than 1000 mm (southwest section) with an average of around 800 mm (Musetha, 2016).

3.2 Methodology

Three belts of 100 m x 500 m were constructed in the Thohoyandou Botanical Garden, and within each belt, 30 quadrats of 10 m x 10 m were randomly constructed. All the invasive alien plant species in each quadrat were identified, recorded and grouped as per their growth forms. The unknown species were collected as voucher specimen to be identified later at the University of Venda, Botany Department Herbarium. The habitat in which they occur was described. The impacts of invasive alien plant species on the adjacent native species were investigated and recorded. This was done by checking the health of native plant species using a scoreboard.

3.3 Data analysis

The data was transferred into Microsoft Excel and was analysed, frequency distribution tables and graphs were used to describe and summarise survey data. Descriptive statistics (mean, median, mode, standard deviation) were also used to further describe the survey data. One-way analysis of variance was also used to compare mean frequency count and mean height between invasive and native species.

Chapter Four

RESULTS

4.1 Distribution of invasive alien species by family names

Out of the 60 quadrats sampled, 32 species collected were identified and classified according to their families. Nineteen families were recorded, and Fabaceae was found to be the most represented (n = 3) followed by Combretaceae (n = 2). The rest of the families contributed one species each.

4.2 Distribution of invasive alien species

The frequency table below presents a summary of all invasive species whose stock were taken from 60 quadrats sampled from Thohoyandou Botanical Garden for the purpose of this study. Table 4.1 presents a summary of distribution of invasive species which were gathered from 60 quadrats sampled from Thohoyandou Botanical Garden.

Table 4.1: Frequency Distribution for invasive species names.

Family	**Invasive alien species**	**Frequency**	**Percent**
Asteraceae	*Chromolaena odorata* (L.)R.M.King & H.Rob.	136	61,0
Verbanaceae	*Lantana camara* L.	29	31,3

Solanaceae	*Solanum nigrum* L.	26	13,0
Asparagaceae	*Asparagus* L.		3.1
Rutaceae	*Citrus lemon* (L. Osbeck)	3	1,3
Anacardiaceae	*Mangifera indica* L.	3	1,3
Fabaceae	*Acacia cyclops* A Cunn. *ex* G.Don	1	0,4

The results presented in the table shows that most of the invasive species recorded at Thohoyandou Botanical Garden were *Chromolaena odorata* which accounted for 60% of the total number of species recorded from 60 quadrats considered in this study, followed by *Lantana camara* (31.3%) and then *Solanum nigrum* (13.0%).

Acacia cyclops constituted the least commonly found invasive alien species in Thohoyandou Botanical Garden with each accounting for 0,4% of the total species gathered from the Botanical Garden.

4.3 Distribution of native species

The frequency distribution table below presents a summary of all native species whose stock were taken from 60 quadrats sampled from Thohoyandou Botanical Garden. Table 4.2 presents a summary of distribution of native species which were gathered.

Table 4.2: Frequency distribution for native species names.

Family	**Native**	**Frequency**	**Percent**
Combretaceae	*Combretum erythrophyllum*	15	16,0
	(Burch.) Sond.		
Apocynaceae	*Rauvolfia caffra* Sond.	14	16.9
Phyllanthaceae	*Bridelia micrantha* (Hochst.) Baill.	4	4,8
Euphorbiaceae	*Euphorbia ingens* E.Mey. *ex* Boiss	4	4,8
Annonaceae	*Annona senegalensis* Pers.	3	3,6
Chrysobalanaceae	*Parinari curatellifolia* Planch. *ex* Benth	3	3,6
Myrtaceae	*Syzygium cordatum* (Hochst.)	3	3,6
Asteraceae	*Vernonia amygdalina* Delile	3	3,6
Fabaceae	*Acacia sieberiana* (DC)	5	6,0
Fabaceae	*Bauhnia galpinii* N.E. Br	5	6,0

Malvaceae	*Dombeya rotundifolia* (Hochst.) Planch.	2	2,4
Ebenaceae	*Diospyros mespiliformis* Hochst. *ex* A. DC	2	2.4
Fabaceae	*Peltophorum africanum* Sond	1	1.2
Phyllanthaceae	*Pseudolachnostylis maprouneifolia* Pax.	1	1.2
Meliaceae	*Trichilia emetica* Vahl.	1	1.2
Combretaceae	*Combretum molle* R.Br *ex* G.Dorn	26	31,3
Moraceae	*Ficus sycomorus* L.	10	4,5
Myrtaceae	*Psidium guajava* L.	2	0,9
Rhamnaceae	*Ziziphus mucronata* Willd.	1	0,4

Nearly 31% of the native species recorded from Thohoyandou Botanical Garden were *Combretum molle* followed by *Combretum erythrophyllum* followed by *Rauvolfia caffra*, *Bridelia micrantha* and *Euphorbia ingens*. Native species with the lowest frequency count were the *Peltophorum africanum*, *Diospyros mespiliformis*, *Pseudolachnostylis maprouneifolia* and *Trichilia emetica* each accounting for 1.2% of the total native species recorded in the botanical garden.

4.4 Descriptive Statistics

4.4.1 Number of species

Table 4.3 presents the descriptive statistics for the number of species variable for the native and invasive species. The mean number of invasive species is 40.780 with a reported standard deviation of 80.6344. This shows that the number of invasive species greatly differs from quadrant to quadrant as evidenced by the extremely high range value of 499 (maximum value –minimum value). The minimum number of invasive species recorded was 1 whilst the maximum number of native species recorded was 500. On the other hand, the mean value of 8.448 and a standard deviation of 10.591 are reported for the number of invasive species. The minimum and maximum number of 1 and 54 are reported for the native species, respectively (see table below for detailed descriptive statistics).

Table 4.3: Descriptive statistics for the number of species variable.

Is the species invasive or native?	Statistic		Statistic value	Standard Error
Invasive	Mean		40.780	6.0608
	95% Confidence Interval for Mean	Lower Bound	28.818	
		Upper Bound	52.741	
	5% Trimmed Mean		27.989	
	Median		7.000	
	Variance		6501.900	
	Std. Deviation		80.6344	
	Minimum		1.0	
	Maximum		500.0	
	Range		499.0	

	Interquartile Range		23.5	
	Skewness		2.852	0.183
	Kurtosis		8.928	0.363
Native	Mean		8.448	0.9840
	95% Confidence Interval for Mean	Lower Bound	6.499	
		Upper Bound	10.397	
	5% Trimmed Mean		6.866	
	Median		5.000	
	Variance		112.319	
	Std. Deviation		10.5981	
	Minimum		1.0	
	Maximum		54.0	
	Range		53.0	

	Interquartile Range	7.8	
	Skewness	2.412	0.225
	Kurtosis	6.161	0.446

4.4.2 Plots for mean number of species.

The following graph compares the two mean numbers of species per each group (native and invasive). Overall, the mean number of species per each quadrat is less for invasive species (mean = 40.780, SD = 80.6344) compared to native species (mean = 8.448, SD = 10.5981) (figure 4.1 below).

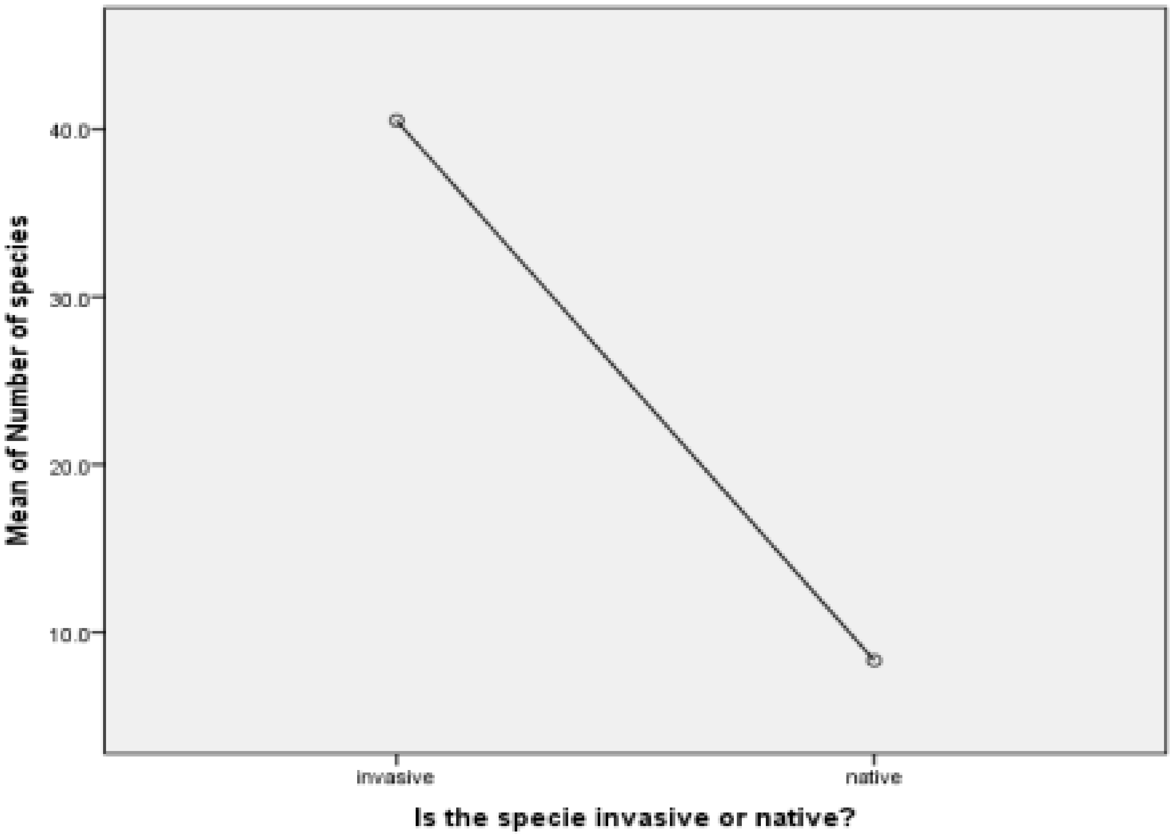

Figure 4.1: A plot for the mean number of invasive and native species.

4.4.3 Height of both invasive and native plant species sampled

The table below presents the descriptive statistics for the species' height variable for both native and invasive species. The mean invasive species' height of 4.169 m and a standard deviation of 4.1655 m are reported. This shows that invasive species' height greatly differs from quadrant to quadrant as evidenced by extremely high range value of 499 (maximum value –minimum value). The minimum and maximum invasive species' height of 0.4 and 20.2 respectively, are reported. On the other hand, the mean height of 9.060 and a standard deviation of 20.1724 are reported for native species. This also shows how the heights for native species differed greatly as evidenced by the minimum and maximum species' height of 0.3 and 215.0 as clearly shown on table 4.4.

Table 4.4 Descriptive statistics for the species' height variable**.**

Is the species invasive or native?	Statistic		Statistic value	Std. Error
Invasive	Mean		4.169	0.3131
	95% Confidence Interval for Mean	Lower Bound	3.551	
		Upper Bound	4.787	

	5% Trimmed Mean		3.661	
	Median		2.500	
	Variance		17.351	
	Std. Deviation		4.1655	
	Minimum		0.4	
	Maximum		20.2	

	Range		19.8	
	Interquartile Range		3.4	
	Skewness		1.910	0.183
	Kurtosis		3.255	0.363
Native	Mean		9.060	1.8730
	95% Confidence Interval for Mean	Lower Bound	5.350	

		Upper Bound	12.770	
	5% Trimmed Mean		7.037	
	Median		6.500	
	Variance		406.927	
	Std. Deviation		20.1724	
	Minimum		0.3	
	Maximum		215.0	
	Range		214.7	
	Interquartile Range		9.3	
	Skewness		9.409	0.225
	Kurtosis		96.452	0.446

4.4.4 Plots for mean height

The following graphs compare the two mean height values between two species' groups (native and invasive). Overall, the mean height per each quadrant is less for invasive species (mean = 4.169, SD = 4.1655) compared to native species (mean = 9.06, SD = 20.17).

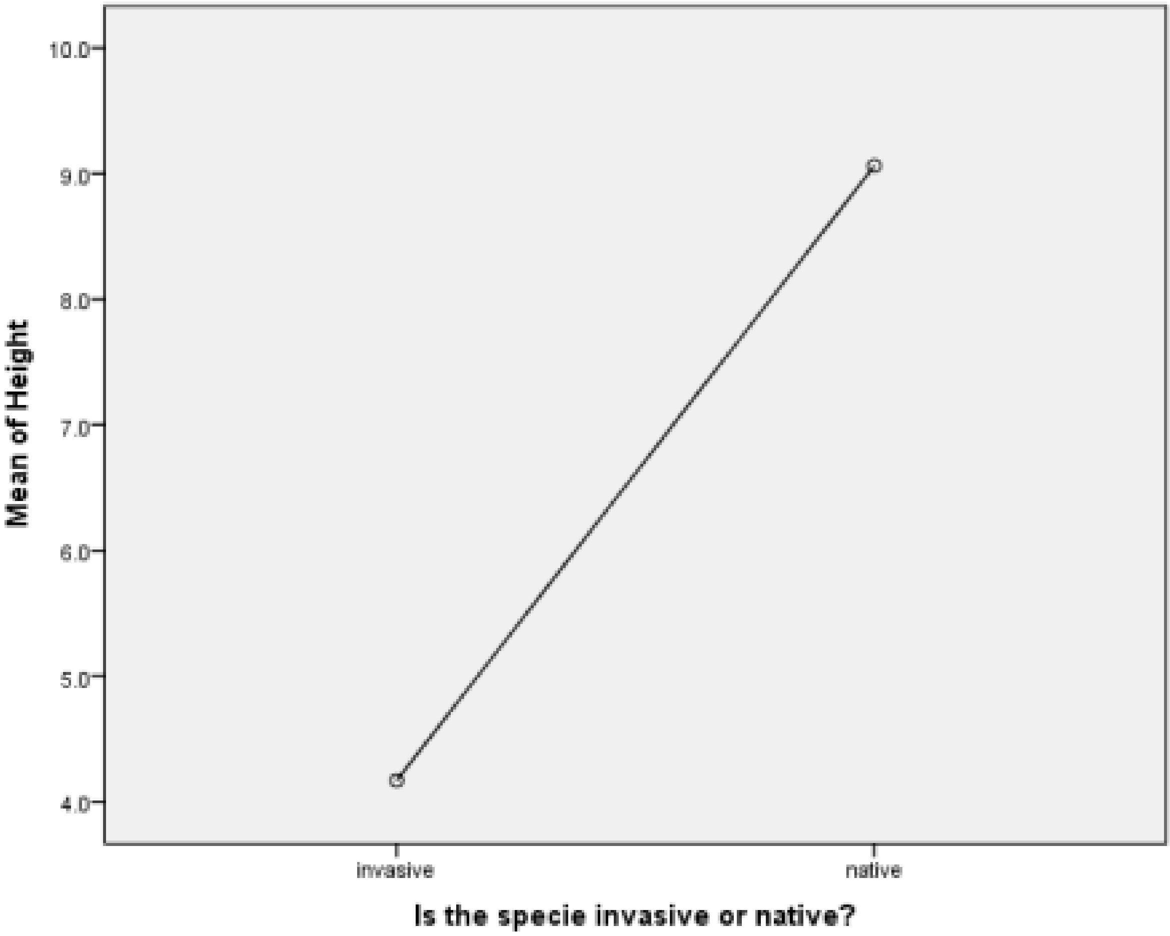

Figure 4.2: A plot for the mean height for the invasive and native species.

4.4.5 One-way analysis of variance (ANOVA)

In this section, we employ the ANOVA technique to compare means between the two species' groups (native and invasive) for the number of species and species' height variables.

4.5 Descriptive statistics of both native and invasive species

The descriptive statistics, including the mean, standard deviation and 95% confidence intervals for the dependent variables (number of species and height) for each separate species group (native, invasive) are as previously reported. Nonetheless, the table below is necessary as it represents the descriptive statistics when the two groups are combined (Total) (see table 4.5 below). For both variables, we see that the fixed effects are statistically significant.

Table 4.5: Summary descriptive statistics for the ANOVA results.

		N	Mean	SD	Std. Error	95% CI for Mean			Maximum	Between Component Variance
						Lower Bound	Upper Bound			
Number of Species	Native		40.51	81.29	5.88	28.91	52.11	1.0	500.0	
	Invasive	120	8.33	10.47	0.96	6.43	10.22	1.0	54.0	
	Total		28.09	65.86	3.73	20.74	35.44	1.0	500.0	
	Fixed Effects			64.07	3.63	20.94	35.24			

	Random Effects					16.46 -				490.02
Height	Native	177	9.06	4.17	0.31	3.55	4.79	0.4	20.2	
	Invasive	116	4.17	20.17	1.87	5.35	12.77	0.3	215.0	
	Total	293	6.11	13.28	0.78	4.58	7.63	0.3	215.0	
	Fixed Effects			13.09	0.76	4.60	7.61			
	Random Effects				2.49	-25.50	37.71			10.74

4.6 Test for equality of variances.

Table 4.6 below shows Levene's Test for Equality of Variances results which provides a way to test if the variance in the two groups is equal (that is, similar or homogeneous). The p-values are less than 0.05 for both number of species and height, hence indicating that the variances in groups (native or invasive) being compared were different.

Table 4.6: Test of homogeneity of variances

Variables	Levene Statistic	df1	df2	Sig.
Number of Species	64.148	1	309	0.000
Height	7.885	1	291	0.005

Table 4.7 below shows the output of the ANOVA technique for making comparison of variable means and whether there is a statistically significant difference between native and invasive group means. We can see that the significance values are 0.000 (i.e., $p < 0.05$) and 0.002 (i.e., $p < 0.05$) for number of species and height of species respectively, which are both less than 0.05 and, therefore, there is a statistically significant difference in the both the mean number of species and the mean height of the species between native and invasive species.

Table 4.7: Analysis of Variance Results.

Variable	Source of Variation	Sum of Squares	Df	Mean Square	F	Sig.
Number of Species	Between Groups	76331.416	1	76331.416	18.594	0.000

	Within Groups	1268484.063	309	4105.126		
	Total	1344815.479	310			
Height	Between Groups	1676.615	1	1676.615	9.787	0.002
	Within Groups	49850.417	291	171.307		
	Total	51527.031	292			

Based on ANOVA results presented above, there was a statistically significant difference in the number of species between the two species' groups (native and invasive) as determined by one-way ANOVA F (1, 309) = 18.594, $p < 0.05$). Similarly, a statistically significant difference in the number of species between the two species' groups (native and invasive) is reported F (1, 291) = 9.787, $p < 0.05$).

4.7 Regression analysis

In this section, we employed the regression modelling technique to show that the number of invasive species is a significant predictor of the extinction of native species. Correlation analysis was also used to show that a statistically significant negative relationship exists between the number of invasive species and native species per each of the 60 quadrats that were considered in this study. A negative correlation shows the extent to which the numbers of native and invasive species are inversely related to each other (that is, as the number of invasive species increases, the number of native species decreases and vice versa).

4.8 Descriptive statistics

Table 4.8 below presents the mean and standard deviation values for invasive and native groups for 54 quadrats, 6 quadrats were excluded in this analysis due to missing values.

Table 4.8: Mean and standard deviation for the invasive and native species.

Variables	Mean	Standard Deviation	Number of species
Number of invasive species	18.20	19.231	54
Number of native species	119.22	96.780	54

4.9 Correlations results.

Table 4.9 below shows the correlation results obtained from relating the number of native species and invasive species. A statistically significant correlation value of -0.367 is reported ($p < 0.05$).

Table 4.9: Correlation analysis results.

		Number of invasive species	Number of native Species
Pearson Correlation	Number of native species	1.000	**-0.367**
	Number of invasive species	**-0.367**	1.000
Sig. (1-tailed)	Number of native species	.	0.003
	Number of invasive species	0.003	.
N	Number of native species	54	54

	Number of invasive species	54	54

4.10 Coefficient of determination (R2)

The following results present a summary of the regression model results. The adjusted R squared value of 0.118 shows that approximately 12% of the total variation is explained by the number of invasive species. The change of R-square value between a model with a constant and the model with number of invasive species as a predictor variable is statistically significant {F (1, 52) = 0.006}. This shows that the number of invasive species in surveyed quadrats was a significant predictor of the extinction of native species in that quadrat.

Table 4.10: Regression Model Summary

Model	R	R Square	Adjusted R Square	Std. Error of the Estimate	Change Statistics				
					R Square Change	F Change	df1	df2	Sig. F Change
1	0.367[a]	0.135	0.118	18.058	0.135	8.106	1	52	0.006[b]
a. Predictors: (Constant), Number of invasive species									
b. Dependent Variable: Number of native species									

4.10.1 Regression model fitness

The ANOVA results presented in the table below provide a way to test the overall fitness of the regression model. In other words, the analysis of variance results presented below are important for testing the null hypothesis that the regression model is unfit versus the alternative hypothesis which says that the model fits the data very well. As clearly shown below, the F-value of 8.16 is reported {F(1,52)} = 8.106, $p < 0.05$) providing evidence to conclude that the simple linear regression model used to model the relationship between the number of native species and that of invasive species was statistically significant at 5% level of significance.

Table 4.11: Analysis of variance results.

Model	Source of variation	Sum of Squares	Df	Mean Square	F	Sig.
1	Regression	2643.529	1	2643.529	8.106	0.006[b]
	Residual	16957.231	52	326.101		
	Total	19600.759	53			

b. Predictors: (Constant), Number of invasive species

4.10.2 Model coefficients

The table below provides a summary of regression model coefficients and other associated statistics. The reported constant value of 26.904 provides the number of native species when the number of invasive species is set to 0. The constant value is statistically significant ($t = 6.861$, $p < 0.05$). The slope value of -0.073 is reported for the regression model implying that each unit increase in the number of invasive species is accompanied by a decrease in the number of native species by a factor of 0.073. By merely looking at the slope value of 0.073, it may appear as if the effect of invasive species on native species is insignificant. However, if we consider the fact that, on average, nearly 40 invasive species were found, the impact becomes visible as 40 invasive species have the potential to reduce the native species at a rate of 2.92 species per every 40 invasive species. The slope is also statistically significant at 5% level of significance ($t = -2.847$, $p < 0.05$). The negative value for the slope serves to confirm that the number of invasive species is a statistically significant predictor of native species' extinction.

Table 4.12: Regression model coefficients.

Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.
		B	Std. Error	Beta		
1	(Constant)	26.904	3.921		6.861	0.000

	Number of invasive species	-0.073	0.026	-0.367[a]	-2.847	0.006
a. Dependent Variable: Number of native species						

Chapter Five

DISCUSSIONS

5.1 Distribution of species by family names.

Fabaceae is the third largest family of the angiosperms after Orchidaceae family and it includes extra than six hundred genera and approximately 12 000 species of trees, shrubs, vines, and herbs and is distributed worldwide (Dyer, 1975). It was therefore not surprising that it dominated on the species collected at Thohoyandou Botanical Garden followed by the Combretaceae family, which consists of 19 genera and more than 600 species (Dyer, 1975).

5.2 Distribution of invasive species

Chromolaena odorata, Solanum nigrum and *Lantana camara* are the top three most found invasive species in Thohoyandou Botanical Garden. This is because they are more competitive than the other species, in terms of spreading, vigorous growth and the way they are distributed(aggregated) in nature. In the context of *L. camara* Angiras *et al.* (1988), reported that aqueous extracts from *L. camara* affect germination of chickpea. Gentle and Duggin (1998) reported that *L. camara* had the potential to restrict local tree species recruitment by resource competition and allelopathy. This may help to explain the observation made in this current study in which the mean number of invasive species of 40.780 (SD = 80.63) was reported, much higher than the mean number of native species of 8.448 (SD = 10.5981). The overall smaller number of native species reported per quadrat may be closely explained by the observation made by Lamb *et al.* (1988), who submitted that the reduced number of growth and recruitment of native plant species has been driven by competition with *L. camara* for light and nutrients. Pei (2013) suggested that the establishment of alien species need to be prevented to discourage species invasion, environmental damage, and ecological risks.

However, the study conducted by Azan *et al*. (2016) demonstrated that most plants marketed in Canadian aquariums are of tropical and subtropical origin and that only those that can adapt to harsh cold conditions become invasive. If the Canadian climate warms in the future under climate change scenarios, the Azan study showed that even aquarium plants that are not currently invasive would become invasive species with a high probability (Verlinden *et al*., 2014). In this regard, the reconstruction of ecological niche models of exotic plants under climate change becomes important, in the sense that these models can help identify (i) plants that could expand their geographic range when monitoring climates. favourable, and (ii) areas likely to be invaded by climate change.

The numbers of native species are lower than the number of invasive in most of the quadrats that were surveyed by this study. Most of the invasive alien plant species were identified as being woody in most cases, with few herbaceous ones. According to Dalmazzone and Giaccaria (2013), the number of invasive alien species, as recorded by the IUCN Global Invasive Species Database (GISD), has been accused of causing severe ecological and economic hardship in a single recipient country (IAS). Moreover, the number of invasive species was used as a proxy of the intensity of threat to native plants at the national level (Dalmazzone and Giaccaria, 2013)). In this study, the number of invasive species was shown to be a statistically significant predictor of native species extinction.

These results might be explained by the conclusions reached by previous studies on the impact of invasive species on abiotic variables. For instance, Le Maitre *et al.* (2011) reported that producing a lot of nutrient rich litter was a known effect of invasive species such as acacias by potentially preventing the recuperation of indigenous and/or native vegetation. Previous studies postulated that invasive species contribute to native species extinction by producing a lot of litter which are further consumed in fire during summer thereby causing vegetation to take longer times to recuperate because of sweltering, harming fires (Blanchard and Holmes, 2008).

Another previous study reported the potential damage that climate change could have on native species by facilitating the further spread of alien plants (Willis *et al.,* 2010). For example, climate change may help reduce the ability of species to spread in the future, thereby making control measures more effective. Vaclavik and Meentemeyer (2009) reported that the ability of exotic plants to propagate and occupy all climate-appropriate habitats is limited by their dispersal ability. As a result, a changing climate that warms can cause currently non-invasive alien plants to become invasive in the future. The results revealed that there are differences in the size and functional diversity of native biota, relative to the number of species, and that their functional diversity has adapted to a similar climate in remote parts of the world (Thomas and Ohlemüller, 2010).

These findings can be important as they can be used to provide the basis for planning, design and implementation of control or eradication strategies to ensure the effective and efficient monitoring of invasive alien species that might be known to truly compromise the survival and growth of native species. Consequently, our findings provide a basis for scrutinizing management and control endeavours aimed at minimising the undesirable environmental impacts of invasive alien species.

Chapter Six

SUMMARY, CONCLUSIONS, AND RECOMMENDATIONS

6.1 Summary

In this study, the effects of invasive species on the growth and survival of native species were investigated. Specifically, the effects of number and height of invasive species as potential predictors of native species extinction were investigated using regression modelling technique. Regression modelling results revealed that height of invasive species was an insignificant predictor of native species' extinction. On the other hand, the number of invasive species has been shown to be a statistically significant predictor of the extinction of native species.

6.2 Conclusions

This result agreed well with previous studies which reported that invasion of native vegetation by invasive species (for example *Lantana camara* etc.) adversely affected native diversity of plant species by reducing the richness and abundance of species, which in turn changed species compositions. Following (Gooden *et al.,* 2009), the results of this study may be used to suggest the need for continued monitoring of native species' re-establishment following invasion by invasive species to prevent secondary weed invasion. This will further help in determining whether long-term communities could occur in the future because of such invasions by invasive species.

This investigation proved that invasive species have the potential to reduce native species at a rate of 2.92 species per every 40 invasive species, on average. Thus, for every 40 invasive species that are found in any chosen area within Thohoyandou Botanical Garden, an average of 3 native species will be lost as a result. This result has a very strong implication in that it can guide management and control strategies aimed at ascertaining that the impact of invasive species growth and survival of native is kept to minimal levels.

6.3 Recommendations

Upon reflection of the higher frequency and infestation of invasive alien plants and their impact on indigenous plants within Thohoyandou Botanical Garden, it is highly recommendable that there be effective management of the alien invasive plants within the botanical garden.

It is important that the invasive alien plants be monitored, and there be environmental education of invasive alien plants implemented, therefore it is highly recommended that there be continued monitoring of native species regeneration following invasion by invasive species to prevent secondary weed invasion.

There should be control endeavours aimed at minimising the undesirable environmental impacts of invasive alien species. The introduction of invasive alien plants must be avoided to prevent invasion of species, environmental damage, and ecological risks.

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