IMPLICATIONS OF *LANTANA CAMARA* INVASION ON RURAL LIVELIHOODS AND NATIVE WOODY SPECIES IN THE KAVANGO ZAMBEZI TRANSFRONTIER CONSERVATION AREA, ZIMBABWE

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A thesis submitted in fulfilment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

Department of Environmental Science

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South Africa

15 January 2024

Abstract

Lantana (*Lantana camara* Linn, Verbenaceae) is an invasive alien species found in many disturbed and conservation habitats worldwide. Much work has been conducted on the impacts that *L. camara* has on ecosystems and communities living in affected areas. In some localities, interventions to eradicate *L. camara* have been made to reduce the impacts, with follow-up assessments on the benefits of these restoration efforts at some locations. However, *L. camara* continues to spread globally, and the Zambezi Transfrontier Conservation Area (KAZA TFCA) in Zimbabwe has not been spared. The KAZA TFCA is an important area for biodiversity conservation, ecotourism, and enabling and encouraging regional peace, cooperation and socio-economic advancement. This study was carried out in the KAZA TFCA, covering the Hwange District in Zimbabwe, with the primary aim to ascertain the effects of *L. camara* on rural livelihoods, native woody species and the soil properties of KAZA TFCA, and whether the policy environment and passive restoration efforts have been effective in controlling it in the area.

The study employed a mixed-methods approach using both qualitative and quantitative methodologies to resolve the research problem. Firstly, 102 plots were randomly established in the study areas where native woody species were recorded. In the first section of the Communal area, there were 60 plots in total comprising of 34 invaded and 8 uninvaded. In the other section of the Communal area, 21 plots were sampled, with 7 invaded, 7 control and the other 7 cleared. In the Victoria Falls National Park, 18 plots were established which comprised of 12 invaded and 6 uninvaded. In the Victoria Falls Rainforest area, 21 plots were established which comprised of 7 invaded, 7 cleared and 7 control. The criteria of selecting 102 plots was based on a number of factors which included; Research objectives, Randomisation, Historical data (From locals and Environmental Management Agency), Environmental variables, Spatial distribution if invasion, Accessibility, Budget and Time constraints. A total of 72 soil samples were collected for laboratory analysis of nitrogen, pH, organic carbon, phosphorous and potassium. The 72 soil samples were considered based on available funding for analysis. This soil samples collected samples were representative of the entire study site. In the Communal areas, 300 household questionnaires (representing approximately 12.35% of the total number of 2 430 households) were administered, two focus groups with elderly residents were conducted, and 11 key informant interviews were held.

One-way ANOVAs and several multivariate tests were conducted to assess the impact of *L. camara* invasion on native woody species composition and soil properties. Descriptive statistics were used to conduct a cost-benefit analysis of the presence of *L. camara* in the study sites. This was coupled with a principal component analysis to assess vulnerability of respondents to invasion, based on variables such as age, gender and level of education. One-way ANOVAs and multivariate analysis were also used to assess if passive restoration contributed to the recovery of native woody vegetation and the improvement of soil properties. In invaded plots, restoration natural recovery occurred after removing the *L. camara* invasion with no further intervention done after the removal. Document and thematic analyses were conducted to evaluate perceptions and knowledge of policy and legislation pertaining to *L. camara* management.

Lantana camara negatively affected the native woody species, as evidenced by a reduction in species richness, abundance, density, height and canopy cover. *Lantana camara* also altered soil properties such as texture, chemical composition, colour and porosity, as found by comparing invaded and non-invaded plots in both the Park and Communal areas. Once *L. camara* invades an area it tends to become dominant at the expense of native species, many of which are a source of livelihood for the local people. Thus, displacement of native species by *L. camara* potentially increases the vulnerability of local people, some of whom are already experiencing challenges, such as drought and human-wildlife conflict.

The invasion of *L. camara* was found to date back over eight decades ago. Based on focus group discussions with elderly participants, the initial invasion of *L. camara* in the Ndlovu Communal area was approximated to 1942. This invasion was noted to have begun in ward 2 of Ndlovu Communal area and spread to the other four wards in the study area over 16 years. Despite the challenges of losses and costs induced by the invasion of *L. camara*, benefits and gains were reported in some areas. For example, in the protected areas, the fruits of *L. camara* are eaten by birds, while bees use the flowers for honey production. It is noted that *L. camara* seeds spread through bird droppings become a cost to the environment through further proliferation of the invasion. In some communal areas, households use *L. camara* for hedges, herbal medicine, cane for furniture, and as an ornament. The relative losses and gains associated with *L. camara* are context-specific and are dependent on several factors.

In this study, the presence of *L. camara* resulted in negative impacts that included the reduced area of productive land for crops and grazing. Combined, the mean costs borne were US$ 483 household^{-1} year^{-1}. Furthermore, the combined mean benefits of *L. camara*, such as use as live fencing around homesteads and fields, mosquito repellent, fruit, fire kindling and ornaments, protection of fowls from raptors, and the prevention of gully erosion along watercourses, were valued at US$ 716 household^{-1} year^{-1}, thereby exceeding the costs by 48%. However, most (62%) households experienced a net negative cost-benefit ratio. The overall mean value across the sample indicates benefits outweigh costs because a minority of households experience large net benefits. Although *L. camara* is invasive in the area, communities have taken advantage of its presence to find uses that contribute to their day-to-day lives.

Despite the use of *L. camara* by communities, efforts to remove it have been made in some parts of the Ndlovu Communal area as well as in the Parks (Victoria Falls and Rainforest). The study investigated the impact of passive restoration on native woody species and soil properties. The findings showed a significant improvement in species richness and diversity in cleared plots compared with invaded plots. The species composition differed in the cleared, invaded and uninvaded plots with specific species favouring the different types of "treatments". This showed the relationship between invasion and associated species. Soil properties had visible differences due to passive restoration, with figures of cleared sites consistently being intermediates of invaded and uninvaded plots.

Despite the passive restoration projects, an analysis of knowledge and management showed that most local respondents whose age ranged between 25- 93 years, did not consider *L. camara* to be invasive but viewed it as part of the vegetation in the area. They also took little responsibility for its management. A further analysis of documents and policies related to the management of invasive alien species found that there were legal instruments in place for the management of invasive alien plants in Zimbabwe, including *L. camara*. The key informant interviews and the questionnaire surveys showed conflicting views on perceptions and opinions about whose responsibility it was to manage *L. camara*, with communities saying it is the authority's responsibility, while some of the authorities suggested that it was everyone's responsibility. However, some of the key informants highlighted the need to review the policies, guided by empirical studies on of the impacts of *L. camara*. Currently, it appears the policies are adopted from international bodies as a means of alignment to global policies, but they are not contextualised to suit the dynamics in the country.

Overall, the study showed that the presence of *L. camara* in the KAZA TFCA has notable negative impacts on the native woody vegetation, as well as on local livelihoods, but some households have adapted to using *L. camara* to benefit them. This, however, has implications for efforts to stop the spread of *L. camara* if its presence is considered a benefit by some and a cost by others. The efforts to remove *L. camara* show that the native woody vegetation recovered. The study showed the need to raise awareness of the invasive nature of *L. camara* because some respondents did not consider it invasive. Findings from this study have implications for the management of *L. camara* in KAZA TFCA and similar areas, and inferences can be made to assist in the management of other invasive alien plants.

Keywords: Biodiversity; Communal areas; Invasive alien species; Legislation; Passive restoration; Protected areas

Declaration

I **Buhle Francis** hereby declare that this thesis is my own original work, and that all the sources consulted have been duly acknowledged within text and in the lists of references. The thesis is submitted in fulfilment of the requirements for the degree of Doctor of Philosophy at Rhodes University, and has not been submitted for any degree or under examination at any other university.

Signature: **Date:** 15 January 2024

Dedication

This thesis is dedicated to my loving family

Acknowledgements

Funding

I appreciate the invaluable support from the European Union (DREAM project – FED/2013/333-266) in partnership with the Research Platform "Production and Conservation in Partnership". The contents of this thesis are the sole responsibility of the author and can under no circumstances be regarded as reflecting the position of the European Union. The National University of Science and Technology (NUST) of Zimbabwe is acknowledged for its contribution (partly) which assisted me with tuition fees to undertake this work at Rhodes University. The economic situation changed in Zimbabwe when I was in the middle of the PhD and NUST discontinued with the tuition support. Many thanks go to Rhodes University for staff remission and various support that ensured the completion of this work. The Environmental Learning Research Centre (ELRC) is gratefully acknowledged for providing the good environment and motivation that gave room for the completion of this work. My gratitude goes to the One Ocean Hub (OOH) project, specifically Dr Dylan McGarry's discretion funds contributed towards writing retreats and editing. I also acknowledge funding from OOH and the Coastal Justice Network (CJN) at Durban University of Technology under Dr Kira Erwin.

Supervisors

My gratitude goes to my supervisors Professor Charlie Shackleton and Professor James Gambiza for supervising this work, as well as to Professor Gladman Thondhlana for the critical supervisory role he played in the inception of the project to conduct fieldwork and partly the write up; his contribution is much acknowledged and appreciated. Although he was not my supervisor, some of this study resonates with Dr Sheunesu Ruwanza's area of expertise, and he took his time to check parts of the work and provide a valuable contribution and insights that shaped this work, and his effort is highly appreciated. During my PhD journey I faced many academic and personal challenges that sometimes interfered with my academic performance, ultimately hindering the progress of my thesis journey. However, Professor Charlie Shackleton took his time to provide comprehensive feedback, and to guide and mentor me in a very responsive way. Working through his guidance made me gain the confidence and motivation which was lost along the thesis journey; for that, I would like to sincerely extend my profound gratitude to Professor Shackleton.

Research assistants

Many thanks go to Tendai Manyuchi, Agrippa Ngorima, Sydney Dube, Martin Muzamba, Ngoni Chiweshe, Brandon Francis, Buhlebenkosi Ngwenya (Macroe), Promise Moyo, Tendai Tsopo and Dr Sibonokuhle Fortune Ncube (Sibo). Sibo and Tendai, you were my rock and pillar during this journey – thank you so much.

Research participants

I thank everyone that participated in this research, especially the communities of Hwange District in the KAZA TFCA as well as the key informants. Working with all of you was joyous and provided insight.

Various colleagues

At Rhodes University, at the Department of Environmental Science, I received support from the staff and a number of PhD and post-graduate colleagues, but mainly Dr Joana Bezzerra, Dr Menelisi Falayi, Dr Chennai Murata and Dr Ojong Enokwena – thank you so much. During my fieldwork, Kathy Kassidy and Monde Ntshudu made it possible by assisting with field equipment – thank you. I received assistance from Dr Frank Akamagwuna who was at the time at the Department of Zoology and Entomology. At the ELRC- OOH project, this work would not have been completed without Danel Janse van Rensburg, Taryn Pereira-Kaplan and Dr Dylan McGarry, who ensured that I receive all the emotional and financial support needed thank you so much. My most deep profound gratitude goes to Professor Heila Lotz-Sisitka, the Director of the ELRC, a center that provides a warm, welcoming and a friendly space to work. Professor Lotz-Sisitka encouraged and motivated me in more ways than one would ever imagine. Discussions on policy with many colleagues from the ELRC, such as Ben de Souza shaped this work to its finality, and I am ever grateful for that. I also acknowledge the wider CJN colleagues including; Dr Jackie Sunde, Dr Kira Erwin and Irna Senekal; I really appreciate all the support. I also acknowledge and thank colleagues from National University of Science and Technology, Professor Mlambo, Dr Sibonokuhle F. Ncube, the late Professor Peter Mundy, Dr Nare, Dr Maphosa and Dr Mabhena, the director of the Institute of Development Studies from where I was seconded to this doctoral work. From the DREAM project, all my colleagues are acknowledged especially Professor Olga Kupika, Richard Mwembe and Dr

Mathieu Bourgarel. I am so grateful for the editorial work of this thesis, provided by Bridgit Davis – thank you.

My family and friends

I thank all my family members for understanding and supporting me financially and emotionally during my thesis journey. To my mother, I am always indebted to you for the advice you gave me and time you spent with me during the last leg of the thesis journey, you would say; "yekela kunjalo" (isiNdebele word for "leave it as it is") and she will continue knitting. To my daughters, you were just amazing for understanding and giving me emotional support – I thank you for your patience. To my partner, thank you so much for all the remote support and the time you spent with the girls while I did the final push. I received a lot of motivational support from my aunt who spent so much time sending motivational voice notes – thank you.

I completed this thesis during my Earth Fellowship which I attained at Strathclyed University in Glasgow, at Scotland, in the comfort home of Caroline and Hugh Anderson. Caroline and Hugh were not just my host family but became friends and supporting elders who provided me with everything I needed to make this work possible and I will always be grateful to them- thank you. Finally, I sincerely appreciate the almighty God for His graces, strength and love. I am thankful to his generous kindness and mercies for guiding me through out. Thank you.

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Chapter 1: The Problem and its Setting

1.1. Invasive alien species

The world is currently faced with multiple threats to natural environments and people's livelihoods (Al Mamun et al., 2022; Keys et al., 2019; Teixeira and Huber, 2021). Among these threats are invasive alien species (IAS), a phenomenon that has been described as far back as the 18th century by biologists such as Charles Darwin and Augustin Pyramus De Candole, who defined them as being non-native in a region and on their potential to threaten native species (Gentili et al., 2021). The spread and establishment of IAS have been due to a combination of natural and human-made factors that encompass globalisation, climate change, habitat destruction, genetic adaptation, lack of natural predators, and intentional and accidental introductions (Black and Bartlett, 2020; Tsiamis et al., 2017; Turbelin et al., 2017). Once IAS occupy an area, they adapt to the new environment, and in some cases alter the dynamics of the system to their advantage (Emery-Butcher et al., 2020; van Wilgen et al., 2022). Through this, some IAS have been identified as major drivers of biodiversity loss and thus as contributing to one of the major crises facing the globe (Fall, 2021).

Work has been conducted on the impacts of IAS for over seven decades, resulting in a comprehensive body of evidence of their generally negative impacts on ecosystems, economies and humans (Gentili et al., 2021). An example is the work of Charles Elton in the 1950s. Elton, who is regarded as the father of invasion biology, revealed that IAS have the potential to disrupt native ecosystems by altering ecosystem dynamics, outcompeting native species or, in some instances, preying on them (Pyšek and Richardson, 2010). From then, more researchers explored this field and there has been increasing evidence to support the concerns about altered ecosystem processes, biodiversity losses and economic losses due to IAS (Davis, 2009). Therefore, these ideas could have contributed to the working definition of IAS by the Convention on Biological Diversity (CBD) (2014), which regards IAS as any plants, animals, pathogens and other organisms that are not native to a particular environment and cause adverse effects. However, Richardson and Rejmánek, (2011) has argued strongly about defining IAS this way, as some notably positive effects have also been found.

There are many negative effects. Most IAS out-compete native species thereby increasing the risk of the extinction of native species, affecting their genetic diversity of native species, causing disruptions in the normal functioning of affected ecosystems and, in some instances, threatening human life through disease transmission (Essl et al., 2020; Pergl et al., 2020; Pyšek et al., 2020). Therefore, IAS in most instances have been linked to environmental instability, and disruption of economic and health systems, and subsequently the livelihoods of vulnerable communities (Heringer et al., 2021). The negative effects of IAS have been recorded across different environments, ranging from marine (Egoh et al, 2020; Shaw et al., 2022) and fresh water (Ahmed et al., 2022; Egoh et al., 2020), to dry deserts (Sohrabi et al., 2023) to moist tropical forests (Rojas-Sandoval et al., 2022).

The economic and social impacts of invasive species include both direct effects of a species on property values, agricultural productivity, public utility operations, native fisheries, tourism, and outdoor recreation, as well as costs associated with invasive species control efforts.

Haubrock et al. (2021) state that IAS costs the global economy an estimated US$ 345 billion annually and a significant amount of such costs are attributable to invertebrates causing major damage to the marine environment. This cost is related to both economic and social costs associated with loss in agricultural productivity, fisheries and tourism. This also includes the cost associated with control programs. IAS that have caused major disruptions within marine systems include *Codium parvulum, Halimeda incrassata, Portunus segnis* and *Xenostrobus securis,* among others (Shannon et al., 2020; Tsiamis et al., 2020). *C. parvulum*, for example, was found to change the habitat it occupies in the Mediterranean to suit its proliferation and outcompete the native benthic communities (Bitar et al., 2017). Furthermore, the spread of *C. parvulum* impacted the economic and social activities of the Mediterranean through disturbing fishing activities and limiting access to sites for winter marine sports such as windsurfing (Israel et al., 2010).

Terrestrial ecosystems are also impacted by many IAS, which include invasive alien plants (IAPs), such as *Chromolaena odorata*, *Opuntia stricta*, *Prosopis juliflora* and *Lantana camara*. IAPs are non-native plants that invade ecosystems and, in the process, modify the conditions of such environments, which enhances their own success and survival (O'Connor and van Wilgen, 2020; Shackleton et al., 2019). For example, *C. odorata* is native to South and Central America and has invaded many regions of Africa and outcompetes the native plant species and reduces biodiversity (Aigbedion-Atalor et al., 2019; van Wilgen et al., 2023). The spread of *C. odorata* has negatively impacted the agricultural sector, biodiversity conservation and livelihoods (Shackleton et al., 2017; Shrestha and Shrestha, 2019). However, some communities residing in areas where *C. odorata* has invaded use it in agricultural crop protection as an alternative insecticide, fodder and the leaves are used for cough treatment (Mugwedi, 2020; Uyi et al., 2021). Other species such as *P. juliflora* were introduced in some parts of Africa as a way of combatting desertification, but eventually spread to such an extent that it had negative effects on the natural ecosystems, taking over grazing lands and water sources thereby impacting local livelihoods (Shackleton et al., 2015; Sintayehu et al., 2020).

Many IAPs are considered as drivers of environmental and ecological change that result in both social and economic costs locally and globally (Montagnani et al., 2022; van Wilgen et al., 2020). In the same vein, the issue of threats to global ecosystem sustainability is highlighted by van Wilgen et al. (2020) who note that IAPs are a threat to all of the world's ecosystems because they can increase at rapid rates, displace indigenous vegetation, disturb important ecosystem processes, have serious environmental impacts, and impact the stability and management of protected areas. For example, molecular studies on species such as *Ageratina edenophora* have shown that its success is based on the relationship it forms with the microbial communities that result in enhanced nitrate levels that suit its expansion. Species such as *Impatiens glandulifera* were found to favour the increase of microbes such as bacteria and a diversity of fungi in the new habitat, making it suitable for *I. glandulifera* to increase at the expense of the native vegetation.

Some IAPs benefit from adverse conditions, such as climatic change and environmental degradation, to dominate some ecosystems (Turbelin and Catford, 2021). A typical example is *Fallopia japonica* which spreads in stressed conditions such as high salt contents (Ashapkin et al., 2020). A study using modelling of IAPs dynamics in the Himalaya region, focusing on *Ageratum conyzoides*, *Chromolaena odorata*, *Lantana camara* and *Parthenium hysterophorus,* found that in the face of climate change these IAPs would not be adversely affected and their habitat would remain stable. This is probably due to their high tolerance of harsh environments (Lamsal et al., 2018).

1.1.1. Costs and benefits of IAPs

The estimated cost of mitigating environmental losses and damage to agricultural systems resulting from to IAPs annually in the United States of America (USA) is over US$ 600 million per annum (Rai and Singh, 2020), while the economic loss from IAPs in China is estimated at US$ 14.4 billion annually (Duncan et al., 2004; Pimentel et al., 2005; Potgieter et al., 2022). In Africa, the economic losses are estimated to be well over US$ 1 billion annually and are mainly due to loss of agricultural productivity (Rai and Singh, 2020; Sileshi et al., 2019).

Opuntia stricta is one IAP that causes problems in South Africa by invading farmlands and protected areas, such as the Kruger National Park (Hoffmann et al., 2019; O'Connor and van Wilgen, 2020). In South Africa, IAPs are estimated to cause losses of R 6.5 billion (US$ 342 million) per annum (van Wilgen et al., 2020). In the Afar region of Ethiopia, *P. juliflora* is causing economic losses in agricultural lands, with over 1.17 million hectares being lost to invasion (Sintayehu et al., 2020), which has serious consequences for the livelihoods of the local people who depend on farming. Furthermore, *P. juliflora* has thorns that can harm livestock and people, apart from causing both ecological and economic problems (Hussner et al., 2017; Sintayehu et al., 2020). In South Africa, the loss of fynbos due to IAPs is estimated to cost around ZAR 700 million (US$ 36.8 million) annually (McConnachie et al., 2012). McConnachie et al. (2012) further note that, in South Africa, IAPs reduce stream flow, which heavily impacts on livelihoods because people need water for their daily upkeep, crop irrigation as well as for sustaining their livestock.

IAPs can modify the ecosystems that they invade in many ways, such as altering nutrient cycling and decreasing litter decomposition (Holmes et al., 2009). Natural vegetation reduces the rate of carbon release into the atmosphere while IAPs tend to show the contrary (Radić and Gavrilović, 2021; Waller et al., 2020). The IAPs that invade and expand, but do not become dominant, will pose fewer changes to the environment (Heshmati et al., 2019). On the other hand, those IAPs that establish themselves efficiently will affect the environment within a short period and cause significant impacts to the native flora (Heshmati et al., 2019). However, over a long period most IAPs show significant impacts by affecting the abundance and diversity of native species, water flows, fire regimes and ecosystem dynamics (Heshmati et al., 2019). In Hungary and Romania, it was shown that there was reduced species diversity, and a reduction of native flowering vegetation, due to presence of twelve IAPs, thus negatively affecting the variabilities of species available to pollinators in the study area (Kovács-Hostyánszki et al., 2022).

The increase in IAPs is determined by their ability to exploit anthropogenic disturbances, the lack of other species that attack them in the new habitats as opposed to their areas of origin, and their allelopathic strategies of being competitive (Mhlongo, 2021; Ojija, 2020). Once IAPs occupy an area, a lack of effective management strategies may also aggravate the increase of IAPs (Pyšek et al., 2013).

Countering the many negative impacts of IAPs, some of them have economic value for local people, contributing to the livelihoods of local people though occupying degraded environments (Kannan et al., 2014; Shackleton et al., 2014). For example, in semi-arid parts of Brazil, the local people prioritise IAPs as opposed to indigenous species, due to the benefits they derive, such as fuelwood, food, aesthetics and timber for various uses (Kull et al., 2011; Shackleton et al., 2014). IAPs such as *Pinus patula* and *Jacaranda mimosifalia* provide attractive and sought-after sites for social outdoor activities, such as picnics and cultural activities (Deena and Thoppil, 2000; Shackleton et al., 2014). In Eastern Ethiopia, IAPs such as *P. juliflora* are used for the commercial production of charcoal and livestock fodder, as well as for improving soil fertility, resulting in many local people being employed in the commercial sector (Haregeweyn et al., 2013).

In the Eastern Cape of South Africa, prickly pear (*Opuntia ficus-indica*) is relied upon by over 92% of households from one village; the people harvest it for food and fodder, and also use it for performing cultural rituals (Shackleton et al., 2007). Many health products have been derived from *Opuntia ficus-indica* such as health bread, drinks and medicine, and also it can be used as fodder and adds aesthetic value (Angulo-Bejarano et al., 2014; Novoa et al., 2016). Furthermore, it reduces soil erosion (especially in an environment where it establishes itself as a monoculture) (Angulo-Bejarano et al., 2014). Some IAPs provide fencing as well as energy in the form of firewood, ethanol and methane gas (Shackleton et al., 2007; Vera et al., 2022). Ngorima and Shackleton (2019) reported that *Acacia dealbata* is a source of firewood, timber and fodder for livestock in the Maclear, Matatiele and Mount Fletcher rural communities in South Africa. It cushions them against hardships when paying for electricity or buying gas (Kull et al., 2011; Ngorima and Shackleton, 2019).

1.2. The origins, ecology and history of *Lantana camara* as an IAP in Zimbabwe

Described as one of the world's most notorious invaders, *Lantana camara* (*L. camara*) is an IAP that is native to South America (Agaldo, 2020; Kato-Noguchi and Kurniadie, 2021). *Lantana camara* is an attractive shrub which usually grows in pure stands. It is, however, also capable of growing interspersed amongst other shrubs and trees (Muvengwi and Ndagurwa, 2015; Ruwanza and Shackleton, 2016). *Lantana camara* is a major weed and has over 600 varieties in its native regions (Assefa et al., 2015; Chatanga et al., 2008). Australia and South Africa have identified and recorded 29 and 40 hybrids of *L. camara*, respectively (Shyamna and Nasser, 2019). The trans-continental distribution of *L. camara* reportedly took place around the late 1600s when it was collected from South America and taken to the Netherlands by the Dutch (Day et al., 2003). It was then propagated in glasshouses that were established solely for its reproduction for export to other countries (Day et al., 2003). Between the 19th to 20th centuries, *L. camara* was spread to many parts of the world, where it naturalised. This was particularly so in Europe (Ibrahim et al., 2021; Thomas et al., 2021).

In South Africa, the introduction is generally accepted to have happened around 1858 and was driven by British colonialism (Ntalo et al., 2022; Vardien et al., 2012). Supposedly it then entered Zimbabwe from neighboring South Africa (Australian Centre for International Agricultural Research [ACIAR], 2012). In Zimbabwe, *L. camara* was deliberately introduced over 100 years ago as an ornamental plant, and quickly spread country-wide in both wet and dry environments (Chatanga et al., 2008). It has affected most native habitats throughout the country (Tarugara et al., 2022). It is a major invader in Zimbabwe, especially in agricultural and rangeland ecosystems (Muvengwi and Ndagurwa, 2015; Raj et al., 2018; Shackleton et al., 2020). Rangelands are the cheapest and most important source of livestock feed in Zimbabwe. The invasion of rangelands by *L. camara* creates a serious threat to many rural communities that, among other livelihood sources, depend on livestock as a source of food and income (Shackleton et al., 2020).

In Zimbabwe, *L. camara* is widespread in disturbed lands, for example, in areas disturbed by fires, abandoned crop fields, as well as overgrazed areas (Muvengwi and Ndagurwa, 2015). In these areas, *L. camara* easily establishes and outcompetes the native flora (Ruwanza and Shackleton, 2016). Indeed, *L. camara* grows in a variety of climatic conditions that facilitate its easy establishment in many countries, including Zimbabwe, which have diverse ecological regions (Raj et al., 2018; Shackleton et al., 2020). Sheppard et al. (2019) argue that the establishment of IAPs such as *L. camara* in rangelands promotes overgrazing because the livestock overgrazes the declining areas of grass land that can still be found. In support of this argument, Muvengwi and Ndagurwa (2015) posit that not only does overgrazing conveniently create the opportunity for weed species that are low in palatability to invade, but it also boosts the growth and establishment of IAPs such as *L. camara.*

1.2.1. Effects of *L. camara* on native woody species and soil properties

The effect of IAPs such as *L. camara* on native woody species is complex as it is specific to vegetation and soil types (Dyderski and Jagodziński., 2020; Erckie et al., 2022). A study in the United States specific to deciduous forests showed that IAPs may increase the rate of litter decomposition and subsequent nitrogen release (Ashton et al., 2005). *Lantana camara* in experimental sites in Ethiopia has been shown to produce allelopathic chemicals that alter soil chemistry and affect germination and root length of native species (Shiferaw et al., 2018). IAPs trigger biological processes within the soil that result in changes to the soil chemistry which gradually lead to changes in plant species composition. Such changes tend to favour the IAPs (Waller et al., 2020). The ability of IAPs to become dominant continues to be a subject requiring further understanding, although this has generally been found to be associated with plant-soil interactions (Aldorfová et al., 2020). *Lantana camara* alters the soil chemistry and enhances the nutrient status to levels that favour the further proliferation of the species (Agaldo, 2020; Kumar et al., 2021).

The production of allelopathic chemicals by some IAPs, such as *L. camara,* may inhibit the germination of native woody seeds and, in some cases, affect the root systems of native woody plants, and thus affect the ability of native woody species to thrive (Kato-Noguchi and Kurniadie, 2021; Kumar and Garkoti, 2022). *Lantana camara* also affects the soil nutrient cycling processes through the disruption of microbial activity (Kumar et al., 2021; Coonan et al., 2020). This has the effect of reducing plant species diversity (Bhatta et al., 2020; Rai and Singh, 2020). The soil nitrogen and phosphorus balance are important for plant growth and survival, and disruption of such important soil nutrients can heavily impact native species' survival (Wang et al., 2021; Zhang et al., 2022). Such has been the case in Sabuk National Park in Kenya (Wekhanya et al., 2020). Further evidence suggests that the invasion risk or the magnitude of the spread of IAPs is higher in areas that are prone to environmental disturbance and adverse climatic changes. For example, in the Ganga valley of the Himalayas, 94% coverage of *L. camara* was recorded in areas with climatic conditions suitable for tropical vegetation as a result of noticeable disturbances such as landslides. Frequent fires made conditions favourable to invasion (Mungi et al., 2020).

Even though IAPs are considered to affect biodiversity negatively, there is evidence that some can play a role in environments that are degraded, by improving soil conditions for secondary plant invasion (Shackleton et al., 2019; Vimercati et al., 2020; Vujanović et al., 2022). *Lantana camara* can assist through the mulching and subsequent moisture retention resulting in the gradual formation of a diverse woody species ecosystem as has been found in Western Himalaya (Schmerbeck and Gupta, 2022). In certain cases with appropriate management prescriptions, native woody species succession occurs through deliberate encouragement of conditions that promote the growth of native species while suppressing conditions that promote the growth of *L. camara* (Ruwanza and Shackleton, 2016).

1.2.2. Uses of *L. camara*

Mukherjee et al. (2015) posit that, in urban areas, *L. camara* may provide a food source for butterflies and other insects to thrive. On the other hand, Kannan et al. (2014) posit that the benefits are usually realised in areas where *L. camara* has invaded for a long time and contributes economically to local livelihoods. The impacts of IAPs on an ecosystem during their initial establishment phase may be minimal and may increase with abundance (van Wilgen et al., 2020). In addition, over time, other co-occurring species become accustomed to it, for example, *L. camara* provides fodder for livestock in some areas in Southern Africa (Ntalo et al., 2022).

In the agricultural sector, *L. camara* biomass is useful as a medium for growing mushrooms, as well as being an insecticide for post-harvest storage (Bouda et al., 2001; Nguyen et al., 2019; Patel, 2011; Sembiring and Zuhra, 2021). Bouda et al. (2001) suggest a strategy of using *L. camara* in the post-harvest control of insects in Africa because it is cheap and effective. Mamun and Ahmed (2011) and Negi et al. (2019) assert that extracts from *L. camara* leaves can kill nematodes and termites, which are a menace in the agricultural sector. Furthermore, *L. camara* is known to provide livelihood options for different communities (Patel, 2011). In parts of India, communities are using *L. camara* to craft furniture, baskets and ornaments, which they sell to raise income (Kannan et al., 2014; Sharma and Kumar, 2009). Various communities in

India use *L. camara* to weave baskets which they sell; they also use it to fence agricultural lands (known as a live fence), as fuelwood, as well as an alternative species to bamboo and rattan, which in certain parts of India were becoming less available (Kannan et al., 2014).

1.3. Management and control of IAPs

Various countries apply several strategies to control IAPs, including *L. camara*. The adaptive resilience and plasticity of *L. camara* make its management unique and specific management protocols that may be environment or habit-specific (Faralli et al., 2022; Kabir et al., 2020). In Zimbabwe, only mechanical removal of *L. camara* has been encouraged, but *L. camara* continues to spread (Maroyi, 2022). In neighbouring countries, such as South Africa and Zambia, biological control has been introduced but with limited success (April et al., 2021; Simelane et al., 2021). However, Zimbabwe does not allow the use of biological control. There is a general concern that the continual use of biological agents as a means of controlling IAPs may result in the agent jumping to indigenous species which may become a problem requiring future management; therefore, such complexities need continuous research (Hill et al., 2021; Hoffmann et al., 2019).

Patel (2011) posits that a sustainable way of eradicating *L. camara* is by use, as eradication by mechanical, biological or chemical means has proven ineffective, while an integrated approach of chemical and mechanical methods can only achieve partial eradication. Therefore, management by use would not only offer a solution to the negative environmental impacts of *L. camara* but also provide a livelihood to affected communities (Kannan et al., 2016). *Lantana camara* can be used for medicinal purposes in a wide range of pharmaceuticals for treating multiple human ailments (Deena and Thoppil, 2000; Souza-Alonso et al., 2017). However, to succeed, the management of *L. camara* must fit the local context in terms of soils, land uses, rainfall, grazing, fire regimes, and the composition and structure of the vegetation (Chatanga et al., 2008). For this thesis, I look at passive restoration and the use of legislation as management strategies that are practised in the region. Passive restoration is explored as an alternative method of management of *L. camara* in Hwange District as it is less labour intensive once the initial threat of the invasion has been cleared. It is expected that indigenous vegetation regeneration will occur naturally. It is not clear why the involved stakeholders opted for these, especially the passive restoration, hence there is a need to look at these options more deeply.

Some IAPs, such as *L. camara,* are known to reduce the viability of certain agricultural and indigenous native species and are therefore economically destructive (Ntalo et al., 2022). It is therefore important that the management of IAPs is taken seriously to control or reduce their negative effects that are associated with IAPs' proliferation. An integrated approach of managing IAPs is important in ensuring that biodiversity and productivity are maintained, especially in environments that are prone to invasions. In Bangladesh, as in other countries where IAPs are threatening the ecosystem, *L. camara* was found to reduce agricultural productivity and regeneration of native species (Rahman, 2023).

1.3.1. Passive restoration as a control strategy of IAPs

Condon et al. (2020) define passive restoration as the active removal of a disruptive situation within any ecosystem, to assist the system to return to its former state via natural transformative ecological processes. This ecological process helps shape the ecosystem after the exclusion of the cause of the undesired disruption. Passive restoration ensures that biodiversity is restored and maintained with the aid of natural biological processes, such as natural litter fall and recycling (Tonello et al., 2022). In some cases, passive restoration can be equated to ecosystem abandonment, with the assumption that natural processes will take place and result in the restoration of the ecosystem to its former state through mechanisms that are self-regulating with no direct influence from humans (Broughton et al., 2021; Moyo and Ravhuhali, 2022; Williams-Linera et al., 2021). The process of passive restoration aids natural plant succession within the ecosystem.

In the USA and Western Europe, the passive restoration of disused agricultural lands provides a classic example of the complexities and possibilities available through the use of passive restoration (Broughton et al., 2021; Lamb et al., 2022). In Southern Africa, passive restoration has been one of the methods adopted in rural lands and protected areas as a means of enabling invaded ecosystems to revert to their natural state (Chazdon and Guariguata, 2016; Singh et al., 2022). In Zimbabwe, passive restorative strategies have been adopted as a way of reclaiming lands in both rural communities and protected areas where *L. camara* has been problematic (Mujaju et al., 2021).

The enabling of natural succession in protected areas and rural farmlands in Zimbabwe has been made possible through community involvement in removing *L. camara* from the invaded areas, and then enabling the natural succession to progress (Moyo and Ravhuhali, 2022). The involvement of affected stakeholders in passive restorative activities is overseen by community leaders through government and donor-funded initiatives. Passive restoration may be expediated in cases where community intervention mechanisms are applied. This is generally possible since communities tend to develop fatigue and a lack of interest (Chazdon and Uriarte, 2016). For its success, passive restoration may require a series of physical interventions to enable natural succession to occur (Prach et al., 2020). This presents a major weakness in heavily infested sites, even though passive restoration is generally considered a cheaper option in comparison to active restoration which involves active management interventions, such as planting desired species, removing weeds and thinning out competing species (Chazdon et al., 2021). However, there is a general lack of adequate data on costs regarding restoration activities, due to minimal formal evaluation (Galatowitsch and Bohnen, 2020). This is because restoration efforts are poorly funded, with many of them relying on grants. In some areas in the United States, 20% of such efforts generally failed due to poor implementation strategies (Galatowitsch and Bohnen, 2020). In some African countries, forestry and landscape restoration is also impacted by complex issues such as stakeholder perception, financial considerations and traditional land uses. About 70% of such projects suffer from lack of funding, and from corruption and enforcement by government departments and non-governmental organisations (NGOs), who are the main enablers (Schweizer et al., 2021).

The success of passive restoration in semi-arid areas, such as in the open savannas of Zimbabwe, seems to be difficult, and semi-arid conditions allow *L. camara* to thrive. It is generally easier in Zimbabwe to monitor passive restoration initiatives in rural communities than in game farms where land is privately owned (Stoldt et al., 2020). Intervention strategies are hampered by a lack of proper monitoring and a lack of interest from officials responsible for managing the initiatives (Moore and Rutherfurd, 2017). Despite the complexities associated with restoration activities, passive restoration offers a better alternative for rural communities, as it is cost-effective and easier for monitoring the progress, even though there is still a need for extensive data collection as shown by work in Australia and the Maloti-Drakensberg areas of South Africa (Bullock et al., 2022). Communities interact with and relate more to the less complex activities involved, and there is less hostility and divergence of ideas, as found in eastern Madagascar where various methods were explored to enable ecosystem rehabilitation (Di Sacco et al., 2021).

1.3.2. Application of legislation and policies as a control strategy for IAPs

The use of legislation and policies in the management of IAPs is meant to ensure uniformity in enforcement of and subsequent compliance to management practices (Bennett and Van Sittert, 2019). Globally, the use of legislation and policies to map out strategies to minimise future invasions of IAPs is common, with countries such as the USA, Australia and New Zealand being at the forefront of such efforts. However, in many developing countries issues of biosecurity receive little attention within communities, and in developed countries, such as the USA, people have little knowledge of IAPs, although enforcement and control systems are better (Pyšek et al., 2020). Legislation and policies are used as mechanisms to enforce control of biological invasions (Ferreira-Rodríguez et al., 2019).

The legislation stems from policies and standards for communities, governments and non-governmental agencies to use in the management of IAPs, and seeks to ensure conformity in enforcement (Ginosar, 2021; Lukey and Hall, 2020). The ability to develop a convergence of ideas about the management of IAPs is important in ensuring that legislation and policies are developed to ensure control of biological invasions. Despite all the legislative instruments, such as the National Environmental Management: Biodiversity Act (NEMBA) of 2004 in South Africa, and the Environmental Management Act of 2002 (EMA) in Zimbabwe, there are challenges in enforcement, primarily due to the lack of funding, private land ownership, and the lack of education and manpower (van Wilgen et al., 2012; Hulme et al., 2018; Mujaju et al., 2021).

Funding for enforcement and control is generally limited in countries such as Zimbabwe, and is coupled with a lack of knowledge, effective management systems, collaboration between enforcement authorities and the communities impacted by IAPs, and by the use of some IAPs by communities (Chitakira and Nyikadzino, 2020; Mujaju et al., 2021). For instance, in South Africa, a substantive amount of the 3-year budget amounting, to R7.8 billion allocated to address environmental concerns dealing with IAPs, is used in policy enforcement and control (van Wilgen et al., 2012). Despite having other numerous challenges to deal with, South Africa is therefore a country in Southern Africa showing government involvement compared to other countries in the region, since IAPs are estimated to cover an estimated 7% of the country (van Wilgen, 2018). In contrast, in Zimbabwe all policies tend to give the responsibility of IAPs management to private land holders, who lack the financial resources and expertise to manage the problem (Chikowore et al., 2023).

In Southern African countries such as South Africa and Zimbabwe, some IAPs are used by communities as food sources, for medicinal purposes and in the provision of other ecosystem services (Reynolds et al., 2020; Shiferaw et al., 2018). This, therefore, results in a conflicted approach to the enforcement of legislation and policies, with some communities being reluctant to get involved in removal of IAPS. Within communities in developing countries such as Zimbabwe, there tends be a lack of knowledge about legislation and policies governing the control of IAPs, as compared to developed countries where there is more policy clarity and resources allocated to management and control initiatives (Chanza, 2018).

1.4. Aims and Objectives

The study aimed to investigate the impacts of *L. camara* invasion on rural livelihoods and native woody species in western Zimbabwe. To tackle this aim, the following objectives were addressed:

1. To assess the impacts of *L. camara* invasion on native woody species composition and soil nutrients in the KAZA TFCA,
2. To ascertain the effects of passive restoration on the native woody species and soil properties in the KAZA TFCA,
3. To determine the cost-benefit ratio and ascertain the local perceptions of the socio-economic effects of *L. camara* on rural livelihoods in the KAZA TFCA,
4. To investigate the stakeholders' knowledge of the management and policies pertaining to *L. camara* invasion in the KAZA TFCA.

1.5. Conceptual frameworks

The study used a combination of the Shackleton et al. (2007) analytical framework for the interpretation of the impacts of IAS on rural livelihoods (Figures 1.1 and 1.2) and the Sustainable Livelihood Framework (Figure 1.3). The frameworks incorporate the concepts of analysing the impacts of IAPs on ecosystems over time, taking into consideration that ecosystems provide goods and services that local people rely on for their livelihood outcomes (Boy and Witt, 2013; Kull et al., 2011). The livelihood outcomes are a result of how the resources are used to earn a living (Chambers and Conway, 1992; Department for International Development [DFID], 1999; Scoones, 1998).

1.5.1. An analytical framework of the impacts of IAS on rural livelihoods

While many IAS cause a great deal of harm, in many rural areas local people incorporate IAS into their day-to-day livelihood strategies (Geesing et al., 2004; Kannan et al., 2014; Siges et al., 2005). As such, dealing with IAS is complex and needs a deeper examination of their roles and impacts within a community to make informed decisions for managerial purposes (Kannan et al., 2014; Shackleton et al., 2007). Shackleton et al. (2007) proposed a conceptual framework for analysing the impacts of IAS on rural livelihoods (Figure 1.1), by taking into consideration the abundance, benefits, costs and vulnerability, which are presented as trajectories over time (Shackleton et al., 2007). Time (which depicts the period of invasion) is further divided into three phases which reveal the phase of progression (Figure 1.1).

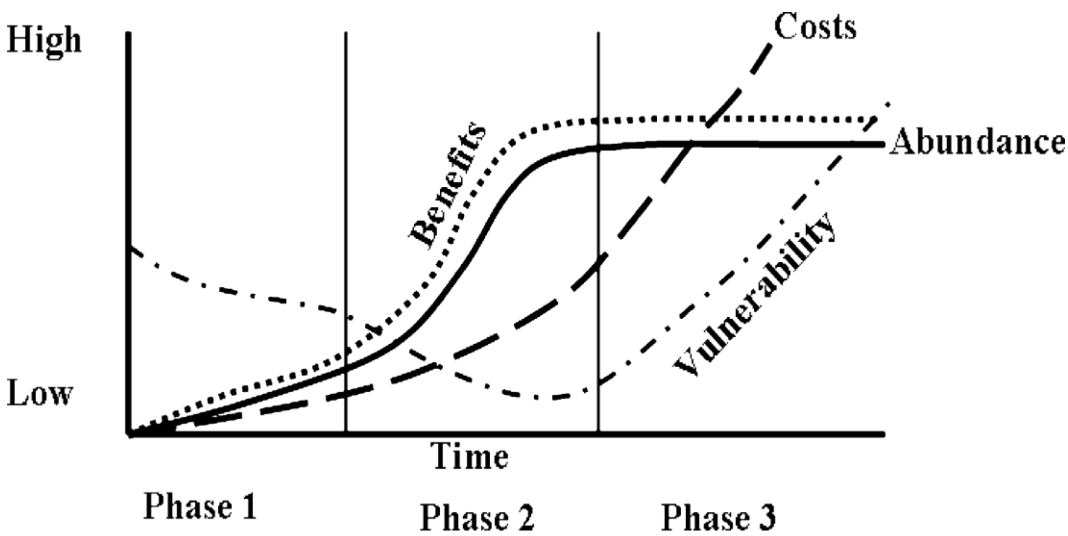

Figure 1.1: Conceptual framework for the interpretation of impacts of IAS on rural livelihoods (Source: Shackleton et al., 2007)

The abundance of the IAS curve begins with low abundance which increases over time, taking a sigmoid shape which is similar to a density-dependent logistical function (Figure 1.1) (Shackleton et al., 2007). The curve of benefits is initially characterised by minimal benefits (benefits are subject to the type and abundance of the IAS), but when abundance is high the benefits are also realised significantly, and this continues to follow the IAS abundance curve until a certain threshold is realised (Shackleton et al., 2007). The costs curve, which is exponentially increasing, represents the costs of the IAS that compound during the progression of time and abundance. The vulnerability curve represents the livelihood vulnerability of the local people and is associated with the relative costs and benefits of the IAS. Depending on the livelihood opportunities that may be offered by an IAS, the cost may be reduced; however, as the abundance increases the socio-economic and environmental costs increase as well, increasing more vulnerability for households (Shackleton et al., 2007).

Phase 1 depicts the initial introduction of the IAS, which is characterised by low abundance and hence minimal costs on the ecosystem and for its control (Shackleton et al., 2007). Furthermore, Shackleton et al. (2007) argue that livelihoods are vulnerable due to other challenges and not necessarily due to the introduction of the IAS. This view is supported by Macdonald et al. (1986), who posit that the age of invasion plays a critical part in the ecosystem, with a focus on IAPs, meaning that when IAPs are in low abundance or new to the environments, the negative impacts are low too, as are the costs to manage them. Therefore, other factors, such as land use changes, socioeconomic factors and other non-invasive species-related challenges like human-to-wildlife conflicts, agricultural practices, reduced access to resources, shifts in market conditions, and other socioeconomic dynamics impact vulnerability. For example, Rogers et al. (2017) highlight other factors such as marginalisation by the state as a major cause of vulnerability among pastoralists in the eastern Ethiopia.

While the Shackleton et al. (2007) conceptual framework postulates that adverse effects become more pronounced as the IASs expands, it is clear that increasing abundance over time impacts biodiversity. Macdonald et al. (1986) give an estimate of two hundred years as the period in which IAPs may start to develop enemies that are equivalent to enemies from their place of origin, and argue that most invaders in Southern Africa have not reached this age. In South Africa, it was established that the density of *Hakea sericea* decreased due to the indigenous fungus *Colletotrichum gloeosporiodes*, and the seed availability of the invasive alien *Acacia cyclops* was decreasing due to predation by local *Hemipteran* insects (Macdonald et al., 1986). The point being made is that Phase 1 provides the invader with a suitable environment to establish itself, considering that control mechanisms are not in place and that the locals may not even be aware of its presence (Shackleton et al., 2007).

In Phase 2, the abundance of the IAS has significantly increased, resulting in rural communities becoming aware of its presence, be it in their fields, around homesteads, in grazing areas or their general surroundings (Shackleton et al., 2007). Assuming that the IAS has some benefits, this may reduce livelihood vulnerability because the locals begin to diversify their livelihoods through the use of the IAS (Shackleton et al., 2007). This diversification of livelihoods may be short-lived as costs continue to increase, thereby catalysing intervention strategies from local authorities and other organisations that work locally (Shackleton et al., 2007). As an example, in Zimbabwe, *Cylindropuntia fulgida* var. *fulgida* invaded the Province of Matabeleland South in the early 1960s but was only declared an invasive plant in 2008 when it had reached crisis levels (Francis et al., 2018). This shows that when the invasion is noticed or people become aware of it, the IAS is useful to them and/or it is costly to remove (Shackleton et al., 2007). Costs of trying to control *Cylindropuntia fulgida* var. *fulgida* were realised in Zimbabwe when the government tried to intervene by introducing control measures, which were not effective and, above all, were not in consultation with the local people (Masocha, 2010; Costanza et al., 2016).

In Phase 3, the costs finally go beyond the benefits, with the ratio becoming increasingly negative unless the IAS is controlled or eradicated, or new major benefits are recognised, or the rural people live with it, resulting in prejudiced livelihood options leading to increased vulnerability (Shackleton et al., 2007). Hulme (2006) suggests that, when dealing with IAPs, the general rule that is cost-effective is to ensure that prevention measures are in place, followed by early detection, eradication and control.

The shape and steepness of the curves of costs, benefits and abundance are determined by the features of the species and local context (Shackleton et al., 2007). Four different classes of species were categorised, based on the competitive ability of the IAS (strong or weak) and its usefulness (high or low) (Figure 1.2) (Shackleton et al., 2007).

The importance and relevance of the Shackleton et al. (2007) conceptual framework in the study are that the framework gives insights into *L. camara* by relating it to the impacts of the four curves and phases over time as a guide to how *L. camara* has impacted the benefits, costs and vulnerability in the environment.

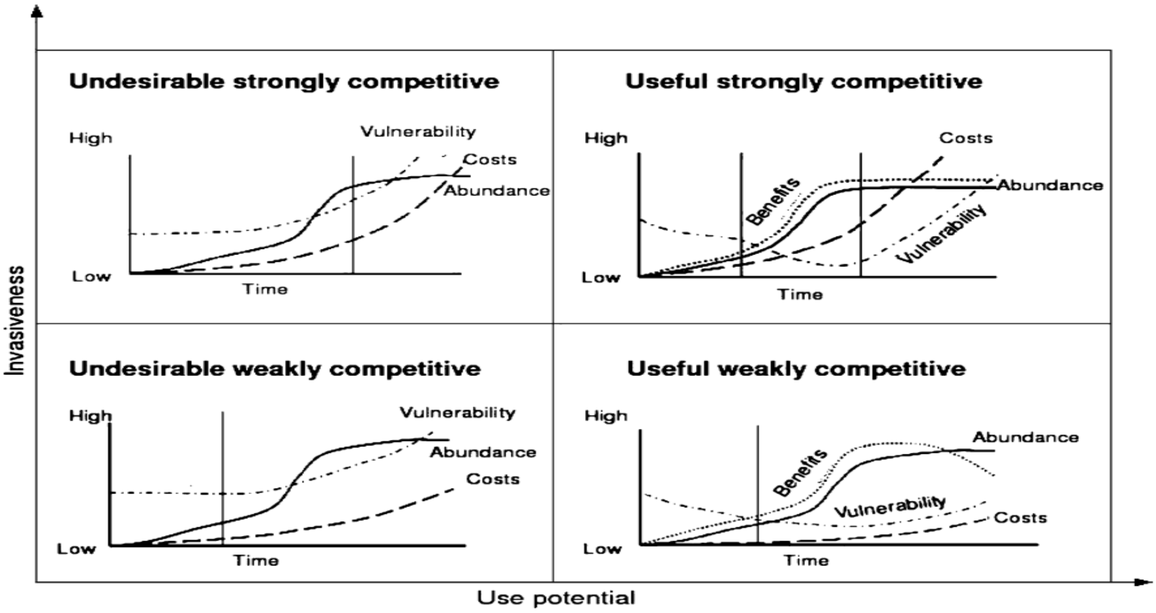

Figure 1.2: Conceptual framework of different IAS (Source: Shackleton et al., 2007)

1.5.2. The Sustainable Livelihoods Framework

The Sustainable Livelihoods Framework (SLF) was established to provide a tool to improve the understanding of livelihoods, especially the livelihoods of the poor (Serrat, 2017a). It has been adopted by academics and policymakers as a pillar in applied research and fieldwork in economically less developed countries (Natarajan et al., 2022). The term livelihood has been defined differently by different scholars. According to Scoones (1998), a livelihood refers to the capabilities, assets and activities needed to create a living. Niehof (2004) posits that a livelihood is a multi-faceted concept of what people do and how they do it to produce results. The definition of livelihood entails functions that relate to people deriving income and resources that are beneficial to survival (Kumar et al., 2019). For rural communities, livelihoods tend to encompass activities carried out both on the land holding and its peripheral systems (Lin et al., 2022; Makwindi and Ndlovu, 2021). It can further be said that livelihoods refer to how humans engage in the means to sustain themselves (DFID, 1999). The SLF is used as an analytical tool for poverty reduction (Fahad et al., 2022; Zhang et al., 2020).

A livelihood is sustainable if it can recuperate from stresses and shocks, or increase capabilities and assets whilst not compromising the natural resource base (Karki, 2021; Serrat, 2017b). The core principles of the SLF are that it is people-centred, responsive and participatory, multi-level and dynamic (Pahl-Wostl, 2009). Households that are unable to withstand shocks that are brought about by the presence of IAS may be vulnerable (Scoones, 1998). The deteriorating conditions can be used to identify unsustainable livelihoods, as they fail to survive short and/or long-term changes, thereby making them more vulnerable to the effects of the IAS (Scoones, 1998).

Rural communities rely on various and numerous livelihood options that usually include harvesting natural resources (Yiridomoh et al., 2021). Some of the numerous livelihood strategies that rural people engage in involve activities such as fishing, harvesting and selling of firewood, timber resource exploitation, and harvesting of non-timber forest products such as honey, mushrooms and mopane worms (Kamwi et al., 2020). In addition, livestock production, off-farm employment, agricultural land utilisation, small businesses and other opportunities form part of livelihood survival strategies. Sustainable livelihoods are based on the accumulation and use of a variety of assets which are human, financial, natural, physical and social (Bvuma and Marnewick, 2020). Furthermore, to achieve sustainable livelihoods, such effective interventions should be made by policy makers and other stakeholders to ensure that these assets are protected from shocks and stresses (Diedrich et al., 2019). Scoones (1998) argues that it makes sense to consider psychological, political and self-evaluation issues as well, for example when dealing with employment. Figure 1.3 shows the context in which livelihood assets are impacted, and the required mitigatory strategies to maintain positive outcomes through effective utilisation of the five capitals.

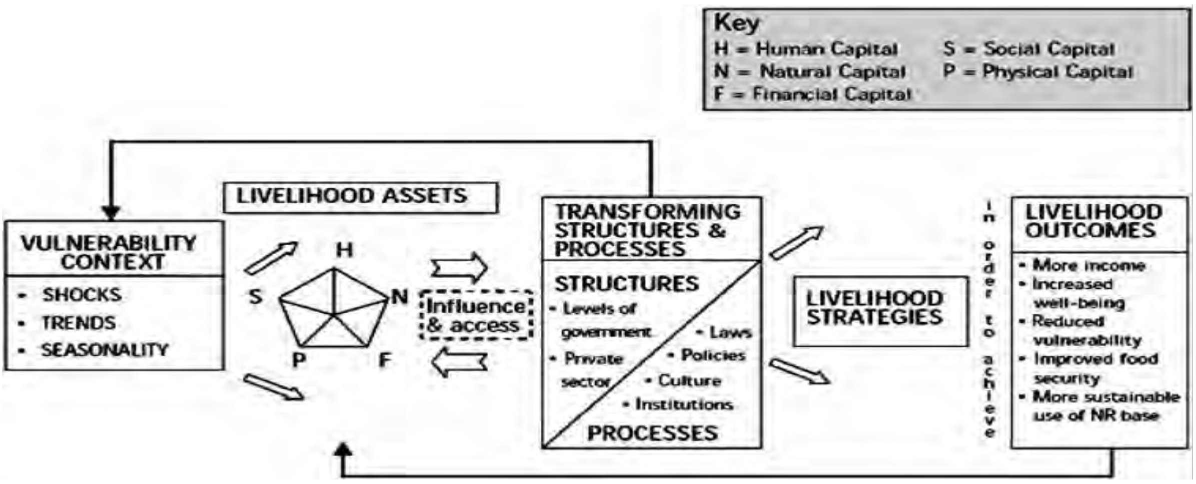

Figure 1.3: The Sustainable Livelihoods Framework (Source: DFID, 1999)

Livelihood assets, vulnerability context, transforming structures, livelihood strategies and livelihood outcomes are the five main parts of the SLF (Figure 1.3). However, this study only applied livelihood assets, vulnerability context and transforming structures to the subject matter because this study was also using the Shackleton (2007) framework.

1.5.2.1. Livelihood capitals

Livelihoods are a function of five capitals that include natural, financial, physical, human and social capitals, that rural communities depend on (Diedrich et al., 2019; Scoones, 1998). Natural stocks such as environmental goods and services, which include biodiversity, soils and water, are known as natural capital (Ignatyeva et al., 2020). The importance of natural capital is realised fully in rural communities, especially in sub-Saharan Africa, where many livelihood strategies rely directly on natural capital (Nguyen et al., 2019; Patel, 2011; Scoones, 1998; Sembiring and Zuhra, 2021).

The destruction of natural capital results in serious consequences for rural people whose livelihoods are derived from natural capital. As evidence of the importance of natural capital to rural livelihoods in the Kgalagadi Transfrontier Park in South Africa, if local communities were not to include income from natural capital, then the poverty incidence and poverty gap would increase by 13 and 7% points, respectively (Thondhlana and Muchapondwa, 2014). Pratt et al. (2017) point out that poor rural communities are most affected by IAPs, and White et al. (2022) note that the increase in woody IAP cover could modify the supply of ecosystem provisioning services, and this may worsen the situation of already vulnerable communities that rely on natural resources.

According to Scoones (1998), cash assets, savings and credit capabilities, that are used for production and in support of households, are all referred to as financial capital. Martin and Lorenzen (2016) postulate that financial capital that can be realised by rural communities may come from entrepreneurship, which in turn is used for livelihood activities. This could be in the form of selling the products that they derive from natural resources, such as agricultural products, furniture, medicine and timber. On the other hand, physical capital refers to tools and equipment, infrastructure such as roads, access to healthcare and water, and any services that could influence people's ability to earn a satisfactory living (Scoones, 1998). This means that, if the necessary infrastructure is not in place, then it could have adverse effects on livelihood outcomes.

Human capital is defined as the skills and the means to work in good health as well as being physically fit (Scoones, 1998). Mehdi and Chaudhry (2015), in agreement with Skoufias (2001), postulate that sometimes human capital takes note of the quality and quantity of work, as well as the resources such as the tools and equipment, that a household has. The recognition that quality of work can influence household work outcomes through pursuing opportunities in the economy, and by maintaining a healthy well-being in the work environment, prompted the examination of human capital (Rakodi, 2002). James (2000) suggests that the human quest to pursue survival measures and mechanisms makes employment and preservation of the various capitals important, with a major focus on social capital so as to preserve economic well-being.

The Department of International Development (DFID) (1999) highlights that there has been a debate regarding the meaning of the term 'social capital', as scholars interpret it differently. However, the SLF is about the social resources that people rely on to meet their livelihood needs. These are established through networks and connectedness among individuals or groups whose interests are the same, and this may grow their trust and capability to work together and widen their access to more institutions which could be political or civic (DFID, 1999). Social capital plays a very important role because it builds and increases mutual trust which may impact other capitals by creating innovations such as new knowledge development, which is part of human capital (DFID, 1999).

1.5.2.2. Vulnerability context

Vulnerability refers to the combination of weakness, insecurity, and exposure to stress, risk and shock (Chambers and Conway, 1992; DFID, 1999; Lwanga-Ntale and Owino, 2020; Mthethwa and Wale, 2020). The inability to avoid danger and threats, being ineffective in terms of politics, and being poor to the extent of living in a state of danger are termed as being vulnerable (O'Riordan and Stoll-Kleemann, 2002). According to Adger (2006), the concept of vulnerability deals with marginalisation, helplessness, and being prone to harm. Makwana (2019) argues that livelihood sustainability can decrease vulnerability.

Resilient livelihood strategies for some households may result from self-assessments which trigger mechanisms that result in preparedness (Koech et al., 2020; Zimbabwe Vulnerability Assessment Committee [ZimVac], 2020). Vulnerability assessment refers to some standardised methods that are laid down to integrate and scrutinise associations between humans and their physical and societal environments (Anderson et al., 2019; Armaş and Gavriş, 2016). From

DFID's point of view, Dauda's (2017) analysis of vulnerability allows the incorporation of issues of preparedness, prevention and mitigatory recovery mechanisms in a disaster. The relationship that exists between vulnerability and poverty cannot be disputed, and thus the battle against poverty ought to involve the decrease of vulnerability (Adger, 2006; Dauda, 2017; Kasperson et al., 2022). Bearing this in mind, the aim ought to be to safeguard and strengthen livelihoods in such a manner that livelihood strategies turn out to be very resilient (Heck et al., 2020; Zhao et al., 2022).

1.5.2.3. Transforming structures and processes

Transforming structures and processes within the SLF refer to public and private organisations (institutions), norms, culture and other factors that influence people's lives in a community (Shoeb-Ur-Rahman et al., 2020; Natarajan et al., 2022). Institutions function in many ways, which could be formal or informal, at local, national, regional and/or international levels (Nordberg et al., 2020). The livelihood assets that contribute meaningfully towards livelihood outcomes, and the transforming structures and processes, provide an enabling environment for promoting survival strategies. This might entail the development of systems and processes that promote transformative social innovation which is important in sustaining rural livelihoods (Castro-Arce and Vanclay, 2020). Transformative processes may help harness human capacity that enhance livelihoods (Urban and Mutendadzamera, 2022).

1.5.2.4. Livelihood strategies and outcomes

A series and combination of actions and selections that people take and make to achieve their livelihood goals, including productive actions, investment strategies and reproductive choices, are defined as livelihood strategies (DFID, 1999). Livelihood outcomes are the result of the strategies; examples of outputs include income, well-being, self-esteem, health status, access to services, a sense of inclusion, and reduced vulnerability (Scoones,1998).

Livelihood strategies adopted to address needs are similar at village, ward, District, Province or national levels. There are various ways to characterise the strategies of households as a means of reducing vulnerability. In South Africa, issues of food insecurity have increased even though the country is considered to be much better economically in comparison to other regional peers. Policy makers have adopted various strategies which include putting in place a social grant system to cushion the vulnerable (Mthethwa and Wale, 2020). In Zimbabwe, with higher levels of poverty, some households have pooled their resources and formed community gardens to supplement their meagre resources and hedge themselves against the shocks and stresses of IAPs, droughts and other economic stressors (Mwakiwa et al., 2018; Zivenge et al., 2013). Therefore, the SLF guided the identification of the livelihood capitals that were impacted by the presence of *L. camara* and how it affected local people, and possible measures to mitigate the effects.

1.5.2.5. Weaknesses of the SLF

The SLF generalises the issues that impact vulnerability and does not consider the macroeconomic changes and unforeseen conflict impacting livelihoods at both a local and national level (Serrat, 2017a; Serrat, 2017b). The framework also assumes that decision-makers must apply uniformity in addressing inequalities, and it assumes that capital assets can be applied in an incremental and generalized trend, which is very unlikely considering the variables that are involved, such as the conflicts and the level at which livelihoods are impacted (Serrat, 2017a; Tambe, 2022). It also applies a uniform approach in assuming that the capital assets are expandable and can be applied in incremental trends without due consideration of the shocks and stresses that can impact such efforts. In that regard, it also fails to recognise the inequalities that exist within power structures and its application (Tambe, 2022). The SLF also understates that the measures that might uplift the livelihoods of some households might trigger conditions that compromise other groups (Serrat, 2017b; Tambe, 2022). In this study the SLF is therefore used in combination with the Shackleton et al. framework (2007). Further, the SLF does not fully address the ecosystem and ecological interactions, which play a fundamental role in shaping the livelihoods of the people (Yletyinen et al., 2021).

1.6. Selection of study site and overview of Kavango Zambezi Transfrontier Conservation Area

This study is part of the research work that is promoted by the Centre de Coopération Internationale en Recherche Agronomique pour le Développement (CIRAD). This centre is a French agricultural research and cooperation organisation that works in over 50 countries, focusing on climate change, food security, natural resource management decreasing inequalities, and poverty mitigation. To achieve its mandate, CIRAD works in partnership with other research organisations, and they identify possible study sites and specific areas of research that are in line with its focus. Over 25 years ago they established themselves in

Zimbabwe where they partnered with a few universities. They selected the Transfrontier Conservation Areas (TFCAs) as their study sites, namely Great Limpopo (GL), Kavango Zambezi (KAZA) and Lower Zambezi (LZ) (French Embassy in Harare, 2021). In 2007, CIRAD brought researchers drawn from selected universities together and established the Research Platform Production and Conservation in Partnership (RP-PCP) (https://www.rp-pcp.org/). The vision of RP-PCP is to add value through research to the lives of the people living in the margins of protected areas and wildlife of Southern Africa.

To meet this vision, the RP-PCP finds funding to support research capacities at national and regional levels by carrying out action research that is demand-driven and is approached from an interdisciplinary and transdisciplinary perspective. Furthermore, the development of demand-driven research is meant to produce knowledge and innovation that responds to local needs, and acquiring funding to achieve this is one of the key roles of the RP-PCP. Once the funding is established, the RP-PCP lists the topics that need to be researched which would have been requested by stakeholders (demand-driven). In 2014, the RP-PCP sourced funding from the European Union (EU) under the project entitled, ‘Delivering innovation and technology through the REinforcement of Agricultural and Multidisciplinary (DREAM) project’. In line with the vision of CIRAD and RP-PCP, the DREAM project aimed to implement action-research that produces new knowledge that responds to the coexistence of agricultural production and the conservation of natural resources, for the benefit of rural communities (Guerbois et al., 2013). *Lantana camara* was identified as one of the topics that needed to be researched in KAZA TFCA, hence the establishment of this project with funding from the DREAM project which covered the fieldwork.

The KAZA TFCA was established in 2011 and it is the world’s largest TFCA, spanning over 520 000 km^2 and covering parts of Angola, Botswana, Namibia, Zambia and Zimbabwe (Stoldt et al., 2020). In Zimbabwe, the KAZA TFCA falls within the northern part of Matabeleland Province. It comprises diverse areas used as protected conservancy areas, and communal lands (Guerbois et al., 2013). The protected areas designated as a national park extend towards Victoria Falls, which is a prime tourist destination. The communal lands are mainly used for human settlements, cultivation and livestock grazing (Stoldt et al., 2020). KAZA TFCA holds biodiversity resources that are crucial to the sustainable development of the region (Cumming, 2008) and that could buffer the region against the effects of climate change. However, the problems of IAPs hamper such initiatives (Salerno et al., 2021).

The area has not been spared from invasion by *L. camara* due to the dry climatic conditions which make conditions favourable for encroachment. Climate modelling shows that the habitats prone to *L. camara* are likely to increase from 3 to 33%, which is likely to add approximately 6 000 square kilometres of land vulnerable to invasion within the Matabeleland Province in the southern part of Zimbabwe within the next 50 years (Ncube et al., 2020). Such a scenario is likely to result in an increase in the invasion of *L. camara* within the KAZA TFCA, which will have effects on both rural livelihoods and the tourism industry, and also on biodiversity.

1.6.1. Hwange District

To manage the complexities that are faced within the TFCAs, with KAZA included, the actual study was conducted in the Hwange District of Zimbabwe (Figure 1.4). Stoldt et al. (2020) note that the major aim of the KAZA TFCA is to manage the protected areas within its borders, which involves making mutual agreements and arrangements between five member states that have different legislation, land uses and attitudes to conservation, and are also at diverse phases of development, which makes a very complex situation. Therefore, the Hwange District became a more manageable unit of analysis for this study. In the Hwange District, specifically, sampling was undertaken in Victoria Falls National Park (VFNP) (17.91° S, 25.84° E), the adjacent Ndlovu Communal land (18.06° S, 26.19° E) and at the Victoria Falls Rainforest (VFRF) (17.55° S, 25°51 E). The communal land has human settlements while the VFNP and the VFRF are protected areas for all forms of wildlife. The communal area was given the name Ndlovu (a Ndebele name which means elephant in English) by the locals due to the high number of elephants (*Loxodonta africana*) in the area. All three sites fall within the Hwange District, which has a population of 68 541 people (ZimVac, 2020). However, *L. camara* is only found in four Wards comprising of 2 430 households.

The study area falls under Agroecological regions IV and V in Zimbabwe and is characterised by low precipitation, generally 400 to 500 mm annually (Nhemachena et al., 2014a). The climate in the area is typical of semi-arid savannas, with mean daily temperatures ranging from a minimum of 19º C to a maximum of 30º C (Nhemachena et al., 2014b). Soils are mainly loose Kalahari sands, which are characterised by heavy leaching and low fertility, though sandy loam soils are found in isolated patches (Arraut et al., 2018). The vegetation in the area is savanna, which is dominated by deciduous woodlands, and common tree species are *Afzelia quazensis,* *Baikiaea plurijuga*, *Brachystegia bohemmi*, *Colophosphermum mopane*, *Combretum collinum,* and *Pterocarpus angolensis* (Nhemachena et al., 2014a). Hwange District is characterised by low-grass cover, and the most commonly occurring grasses are *Aristida junciformis, Dactyloctenium aegyptium* and *Heteropogon contortus* (Guerbois et al., 2013). The land tenure and allocation system in post-colonial Zimbabwe resulted in a mixed land-use system where traditional communal areas co-exist with the once large-scale, commercial farming systems (Mhlanga and Pisa, 2022). In the Hwange area, there are still large-scale commercial, conservation and protected areas that are interspersed with small rural land holdings (Andersson and Cumming, 2017). The household land holding in the Hwange District varies from 0.5 to 6 hectares, with the average utilisable size per household approximating 3 hectares (Tambo et al., 2021).

To separate the three land uses, the Ndlovu Communal area will be referred to as the 'Communal area', and the VFNP and VFRF as the 'Park'. Despite both locations falling under the same District, they are administered differently, with the Communal area falling under the Hwange Rural District Council, while the Park falls under the jurisdiction of the Zimbabwe Parks and Wildlife Management Authority. The Park was established in 1979 when it was separated from the Zambezi National Park (Stoops et al., 2001). It covers 23 000 hectares, dominated by *Brachystegia* woodlands (Nhemachena et al., 2014a; Stoops et al., 2001) and is home to the 'big five' of Africa (*Diceros bicornis*, *Loxodonta africana*, *Panthera leo*, *Panthera pardus* and *Syncerus caffer*), as well as to over 400 different bird species (Kent, 2011). The Park is not fenced, which allows unrestricted movement of both humans and wild animals to both sites.

In the Communal area, livestock rearing, rain-fed subsistence farming (mainly maize and sorghum), and art and crafts are the main livelihood activities (Ndebele-Murisa and Mubaya, 2015). Yields are generally low from rain-fed farming; hence the majority of villagers supplement their livelihoods with the use of natural resources (Nhemachena et al., 2014b). Human-wildlife conflicts are widespread because of wild animals which roam the Communal areas, damaging crops and killing livestock and sometimes humans (De et al., 2020; Guerbois et al., 2013). Poverty levels in the area are high due to the poor agricultural conditions for both crop production and livestock rearing and due to the destruction of the produce by wildlife, hence there is a lot of donor aid activity in the District (Vorlaufer, 2002). Despite poverty, the literacy level of the population is high in general, with over 70% having attained basic primary education. The predominant language in the area is Ndebele, with a few villagers speaking Nambya and Tonga.

The VFRF area is a globally recognised United Nations Educational, Scientific and Cultural Organization (UNESCO) world heritage site in the northwestern part of Zimbabwe, and is a unique ecosystem within the greater VFNP, which covers about 23.4 km^2. It is located in the extreme north-western part of the Park extending towards the town of Victoria Falls which is a prime tourist destination in Zimbabwe. The town of Victoria Falls is located on the international boundary between Zimbabwe and Zambia at an altitude of about 900 meters above sea level (Mushawemhuka et al., 2021). The town of Victoria Falls hosts a permanent population of around 31 000, apart from tourists that visit the resort town daily (Makwindi and Ndlovu, 2021). It is associated with one of the world's widest sheets of waterfalls, stretching more than 1 700 meters in width with a height of around 108 meters (Rebus, 2019).

The VFRF presents a unique ecosystem which is constantly moist due to the spray and the humidity emanating from the falls (Dube and Nhamo, 2020; Zhou, 2018). The constant mist ensures that temperatures averaging 20º C are maintained within the rainforest. It results in a unique ecosystem that supports a variety of plant species, such as *Phoenix reclinata, Colophospermum mopane*, *Adiantum capillus-veneris*, and the near-endemic *Cheilanthes farinose,* which in turn host a variety of faunal species (Makoni and Chikobvu, 2018; Rebus 2019), including birds (Dube, 2018). Some of the main tree species are *Afzelia quanzesis, Antidesma venosum*, *Berchemia discolar*, *Colophospermum mopane*, *Combretum imberbe* and *Kirkia acuminata*, among other typical species prevalent in rainforest habitats in Zimbabwe. There is a significant herbaceous cover. Some of the ground plants are *Haemanthus filiflorus*, *Calanthe corymbosa* and *Gloriosa superba* (Zimbabwean national flower).

The uniqueness of the VFRF habitat has attracted a lot of interest from visitors, international conservationists and biologists alike. There is a general focus towards balancing the world heritage nature of the habitat and its potential as a driver of economic development in Zimbabwe through the exploitation of its tourism appeal. Under whose custodianship the Victoria Falls Rainforest area lies has been a topical question in numerous debates, due to its uniqueness as a World Heritage-designated ecosystem (Makwindi and Ndlovu, 2021; Makuvaza, 2012).

The VFRF is considered a cornerstone of the Zimbabwean tourism economy (Makwindi and Ndlovu, 2021). The falls are recognised as one of the Seven Natural Wonders of the World, and, as a result, it is one of the most visited areas in Zimbabwe (Makoni and Chikobvu, 2018). Due to its national importance, the Victoria Falls rainforest area is managed by the ZimParks. The 2015 tourism report revealed that 70% of the tourists who visited Victoria Falls were foreigners and about 30% were locals; as such, the destination is a key source of foreign currency for the country (Makoni and Chikobvu, 2018). According to the 2015 Annual Tourism Trends and Statistics, Tendai and Chikobvu (2017) posit that the VFRF received an average of approximately 11 000 visitors every month and the majority of these pass through the rainforest area as they seek to view the mighty gorge of the Victoria Falls waterfall. Such a high volume of traffic, therefore, requires a high degree of management to preserve the continued ecosystem balance of the tourist attraction (Zhou, 2018). The main land-use for the VFRF is primarily tourism and, as such, any management interventions should continue to ensure that this area is maintained in its natural state both for tourism and for environmental conservation considerations. It therefore becomes important that any encroachment by IAPs such as *L. camara* is checked, to ensure continued biodiversity in the VFRF.

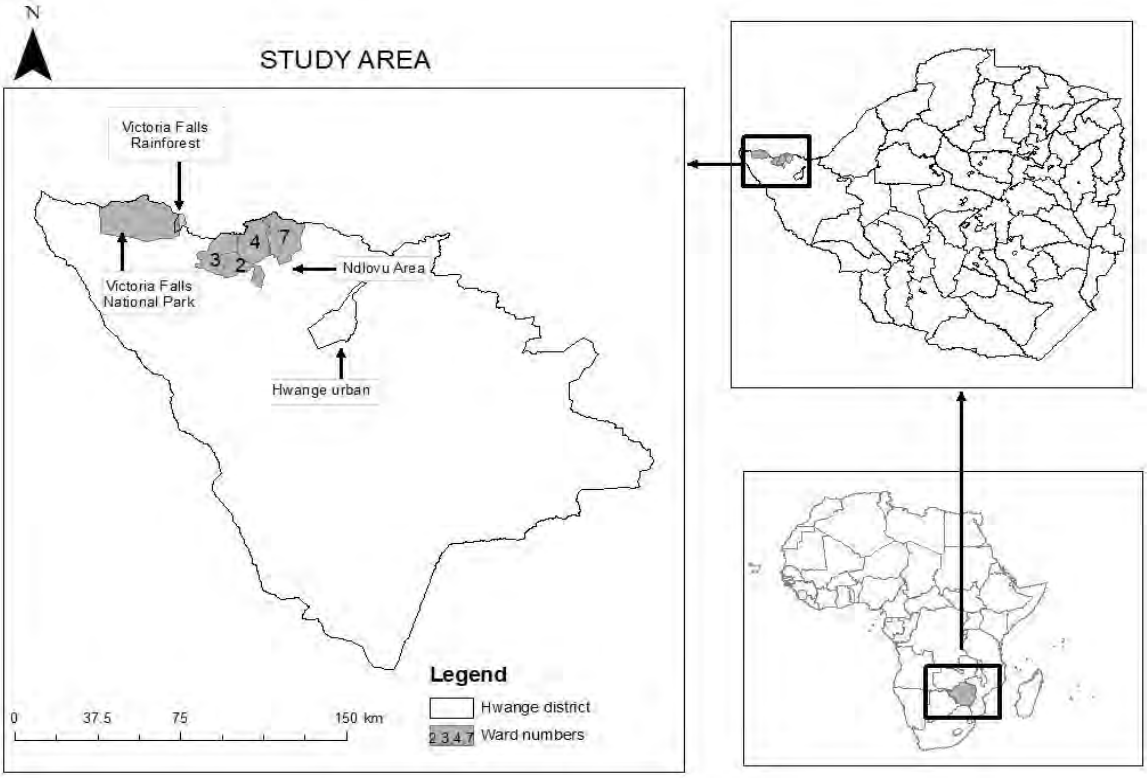

Figure 1.4: The study sites

1.7. Ethical considerations and ethical clearance ES17/39

Ethical considerations are the rules and ways to which the intended research is subscribed (Creswell, 2014). Research ethics is a critical component that every researcher needs to respect. To observe and comply with ethics, the following was done:

The Rhodes University ethical clearance document was read and understood, and its rules and regulations, which included applying for the clearance, were agreed upon. The ethics approval (ES17/39) was obtained. Before the study commenced, all fieldwork protocols, which included meeting up with traditional leadership and other key institutions, were implemented.

A study permit (Appendix 1) was provided by ZimParks, which granted permission for the research to be carried out within the Park. The permit stipulated the terms and conditions that it was granted under, such as the submission of a monthly progress report to the Area manager, not interfering with any animals, not disturbing the habitat such as by destroying trees, and not collecting samples without authorisation, among other requirements.

All interviews including the questionnaire administration (Appendix 5, 6 and 7), were conducted in the local languages which the researcher and the research assistants are fluent in. The respondents were afforded transparency and respect. Furthermore, they were informed that the project was for academic purposes, hence there was no remuneration for their participation. The procedures of informed consent, anonymity and confidentiality were upheld during the study. Also, if a surveyed household wished not to participate, the next available household on the random sampling was selected. More details about this are contained in the consent form in Appendix 4. The researcher held workshops at the study sites to provide the participants with feedback.

The RP-PCP organises knowledge-sharing platforms, where study research findings were shared. Some of the findings were shared with the communities and stakeholders at the study site. Disseminating the findings to the local communities was not only ethical, but it also gave confidence to the locals, built trust and created a warm welcoming environment.

1.8. Structure of the thesis

This thesis is structured into six chapters with chapters 2 to 5 dedicated to empirical study results, while chapters 1 and 6 introduce and conclude the study, respectively. To understand the context of the problem, Chapter 1 begins by giving a background of the problem and its setting. The study contextualisation is written in such a way that it introduces the IAS in general and narrows it to the IAPs. Furthermore, it reveals the cost and benefits that are encountered, specifically concerning *L. camara* which is the focus of this study. The conditions that promote *L. camara* invasion are described, including some of the management aspects that have been applied. The objectives of the study are given. The analytical frameworks of IAS (Shackleton et al., 2007) and the SLF (DFID, 1999), which underpin this study, are introduced and explained. The study site which is central to this research is also described.

Chapter 2 is the first results chapter and deals with the natural capital whereby it focuses on the impacts of *L. camara* on native woody species and soil properties. To achieve this, plots were established within the Park (Victoria Falls National Park) and at the Communal area (Ndlovu Communal area) which is an area with human settlements. The study addresses two main research questions: (i) What is the impact of *L. camara* invasion on native woody species composition and structure in the KAZA TFCA, and (ii) What is the impact of *L. camara* invasion on soil nutrients in KAZA TFCA?

Chapter 3 focuses on a cost-benefit analysis of the presence of *L. camara* on the livelihoods of the local communities. The chapter is underpinned by the livelihood assets and the vulnerability context of the SLF, with *L. camara* viewed from the lens of being a shock and stressor. To achieve this, two focus group discussions were held with the elderly members of the community and 300 household questionnaires were administered. The main research questions addressed by the study are: (i) What is the invasion history of *L. camara* in the Hwange District, (ii) What are the costs of *L. camara* on local livelihood assets, (iii) What are the benefits of *L. camara* on local livelihood assets, and (iv) What is the vulnerability status of local people to exposure to *L. camara*?

Chapter 4 focuses on the natural capital in passive restoration of the native woody species composition and soil properties degraded by *L. camara* invasion in the KAZA TFCA. Experimental plots were established in the Park (Victoria Falls Rainforest area) and within the Communal area (Ndlovu area). The specific questions addressed are: (i) What are the effects of passive restoration (weeding of *L. camara*) on native woody species structure, and (ii) What is the effect of clearing *L. camara* on soil properties?

Chapter 5 is concerned with the stakeholders' knowledge of the management and policies pertaining to the *L. camara* invasion in the study site. This chapter is premised on transforming structures and processes specific to the levels of government laws and policies that guide the management of *L. camara*. The research questions addressed are: (i) How do existing legal and policy frameworks, if any, frame and govern the management of the IAS, (ii) What are the views of the stakeholders on legislation pertinent to the *L. camara* invasion in the study site, and (iii) How can the management of *L. camara* in the KAZA TFCA be improved? In this chapter, the term IAS is used as opposed to IAPs because the legal documents do not specifically refer to IAPs. To respond to the questions, 11 key informant interviews were conducted, and 300 household questionnaires were randomly administered.

Chapter 6 summarises the key findings of the study from chapters 2 to 5 and integrates them to show the relationship and implications of the study. It is a concluding chapter which synthesises the key messages, highlights the limitations of the study and makes recommendations which revolve around policy and further studies.

All four result chapters are written in a paper-based based format and are in preparation for submission to peer-reviewed journals. Each empirical result chapter is composed of an: Abstract, Introduction, Methods, Results, Discussion, Conclusion and References. To avoid repetition, the study site is described only in section 1.6.1. and all empirical result chapters refer to it.

References

Adger, W.N., 2006. Vulnerability. *Global Environmental Change*, 16(3), 268-281.

Agaldo, J.A., 2020. Ecology of *Lantana camara:* An invasive plant species in Amurum Forest Reserve, Jos Plateau, Central Nigeria. *African Journal of Ecology,* 58(2), 291-298.

Ahmed, S.F., Kumar, P.S., Kabir, M., Zuhara, F.T., Mehjabin, A., Tasannum, N., Hoang, A.T., Kabir, Z. and Mofijur, M., 2022. Threats, challenges and sustainable conservation strategies for freshwater biodiversity, *Environmental Research*, 214, 113808.

Aigbedion-Atalor, P.O., Day, M.D., Idemudia, I., Wilson, D.D. and Paterson, I.D., 2019. With or without you: Stem-galling of a tephritid fly reduces the vegetative and reproductive performance of the invasive plant *Chromolaena odorata* (Asteraceae) both alone and in combination with another agent. *Biological Control*, 64, 103-114.

Aldorfová, A., Knobová, P. and Münzbergová, Z., 2020. Plant–soil feedback contributes to predicting plant invasiveness of 68 alien plant species differing in invasive status. *Oikos*, 129(8), 1257-1270.

Al Mamun, A., Islam, A.R.M.T., Alam, G.M., Sarker, M.N.I., Erdiaw-Kwasie, M.O., Bhandari, H. and Mallick, J., 2022. Livelihood vulnerability of char land communities to climate change and natural hazards in Bangladesh: An application of livelihood vulnerability index. *Natural Hazards*, 115(4), 1-27.

Anderson, C.C., Hagenlocher, M., Renaud, F.G., Sebesvari, Z., Cutter, S.L. and Emrich, C.T., 2019. Comparing index-based vulnerability assessments in the Mississippi Delta: Implications of contrasting theories, indicators, and aggregation methodologies. *International Journal of Disaster Risk Reduction*, 39, 101128.

Andersson, J.A. and Cumming, D.H., 2017. Defining the edge: Boundary formation and TFCAs in Southern Africa. In J. Andersson, M. de Garine-Wichatitsky, D. Cumming, V. Dzingirai and K. Giller (eds), *Transfrontier conservation areas*, 25-61. Routledge, Oxfordshire, United Kingdom.

Angulo-Bejarano, P.I., Martínez-Cruz, O. and Paredes-Lopez, O., 2014. Phytochemical content, nutraceutical potential and biotechnological applications of an ancient Mexican plant: Nopal (*Opuntia ficus-indica*). *Current Nutrition Food Science*, 10(3), 196-217.

April, V., Simelane, D.O. and Robertson, M.P., 2021. Co-existence between two leaf-feeding biological control agents of *Lantana camara* alters their herbivory under semi-field conditions. *Neotropical Entomology*, 50(6), 959-965.

Armaş, I. and Gavriş, A., 2016. Census-based social vulnerability assessment for Bucharest. *Procedia Environmental Sciences*, 32, 138-146.

Arraut, E.M., Loveridge, A.J., Valls-Fox, H., Macdonald, D.W. and Chamaillé-Jammes, S., 2018. The 2013–2014 vegetation structure map of Hwange National Park, Zimbabwe, produced using free satellite images and software. *Koedoe: African Protected Area Conservation and Science*, 60(1), 1-10.

Ashapkin, V.V., Kutueva, L.I., Aleksandrushkina, N.I. and Vanyushin, B.F., 2020. Epigenetic mechanisms of plant adaptation to biotic and abiotic stresses. *International Journal of Molecular Sciences*, 21(20), 7457.

Ashton, I.W., Hyatt, L.A., Howe, K.M., Gurevitch, J. and Lerdau, M.T., 2005. Invasive species accelerate decomposition and litter nitrogen loss in a mixed deciduous forest. *Ecological Applications*, 15(4), 1263-1272.

Assefa, Y., Tiroesele, B., Segwagwe, A. and Madisa, M.E., 2015. *Lantana camara* and its biocontrol agent, *Teleonemia scrupulosa* in Botswana. *African Journal of Ecology*, 53(3), 381-384.

Australian Centre for International Agricultural Research [ACIAR]. 2012. *An assessment of the status, impact and management of weeds in Zimbabwe, with a particular focus on biological control, in the context of food security and sustainable production from smallholdings and water use in lakes and rivers – Zimbabwe*. Technical Report. Australian Centre for International Agricultural Research, Canberra, Australia.

Bennett, B.M. and Van Sittert, L., 2019. Historicising perceptions and the national management framework for invasive alien plants in South Africa. *Journal of Environmental Management*, 229, 174-181.

Bhatta, S., Joshi, L.R. and Shrestha, B.B., 2020. Distribution and impact of invasive alien plant species in Bardia National Park, western Nepal. *Environmental Conservation*, 47(3), 197-205.

Bitar, G., Ramos-Esplá, A.A., Ocaña, O., Sghaier, Y.R., Forcada, A., Valle, C., El-Shaer, H. and Verlaque, M., 2017. The introduced marine macroflora of Lebanon and its distribution on the Levantine coast. *Mediterranean Marine Science*, 18(1), 138-155.

Black, R. and Bartlett, D.M., 2020. Biosecurity frameworks for cross-border movement of invasive alien species. *Environmental Science and Policy*, 105, 113-119.

Bouda, H., Tapondjou, L.A., Fontem, D.A. and Gumedzoe, M.Y.D., 2001. Effect of essential oils from leaves of *Ageratum conyzoides*, *Lantana camara* and *Chromolaena odorata* on the mortality of *Sitophilus zeamais* (Coleoptera, Curculionidae). *Stored Products,* 37, 103-109.

Boy, G. and Witt, A., 2013. *Invasive alien plants and their management in Africa*. Centre of Agricultural and Biosciences International, Nairobi, Kenya.

Broughton, R.K., Bullock, J.M., George, C., Hill, R.A., Hinsley, S.A., Maziarz, M., Melin, M., Mountford, J.O., Sparks, T.H. and Pywell, R.F., 2021. Long-term woodland restoration on lowland farmland through passive rewilding. *PloS One*, 16(6), 0252466.

Bullock, J.M., Fuentes-Montemayor, E., McCarthy, B., Park, K., Hails, R.S., Woodcock, B.A., Watts, K., Corstanje, R. and Harris, J., 2022. Future restoration should enhance ecological complexity and emergent properties at multiple scales. *Ecography*, 2022, 4.

Bvuma, S. and Marnewick, C., 2020. Sustainable livelihoods of township small, medium and micro enterprises towards growth and development. *Sustainability,* 12(8), 3149.

Castro-Arce, K. and Vanclay, F., 2020. Transformative social innovation for sustainable rural development: An analytical framework to assist community-based initiatives. *Journal of Rural Studies*, 74, 45-54.

Chambers, R. and Conway, G., 1992. *Sustainable rural livelihoods: Practical concepts for the 21st century*. Institute of Development Studies, United Kingdom.

Chanza, N., 2018. Limits to climate change adaptation in Zimbabwe: Insights, experiences and lessons. In Leal Filho and J. Nalau (eds), *Limits to climate change adaptation,* 109-127. Springer, Cham.

Chatanga, P., Kamanda, M.T., Kundhlande, A., Imbayarwo-Chikosi, V.E., Mujawo, T., Magadza, C.H.D. and Mujuru, L., 2008. Effects of *Lantana camara* (L) invasion on the native vegetation of Gonarezhou National Park, Zimbabwe. *Southern Africa Journal of Education, Science and Technology*, 3(1-2), 32-43.

Chazdon, R.L. and Guariguata, M.R., 2016. Natural regeneration as a tool for large-scale forest restoration in the tropics: Prospects and challenges. *Biotropica*, 48(6), 716-730.

Chazdon, R.L. and Uriarte, M., 2016. Natural regeneration in the context of large-scale forest and landscape restoration in the tropics. *Biotropica*, 48(6), 709-715.

Chazdon, R.L., Wilson, S.J., Brondizio, E., Guariguata, M.R. and Herbohn, J., 2021. Key challenges for governing forest and landscape restoration across different contexts. *Land Use Policy*, 104, 104854.

Chikowore, G., Martin, G.D., Chidawanyika, F., Hill, M., Neser, S., Day, M., Grice, T., Chikwenhere, G., Mangosho, E. and Sheppard, A., 2023. Weed biological control in Zimbabwe: Challenges and future prospects. *South African Journal of Botany*, 154, 336-345.

Chitakira, M. and Nyikadzino, B., 2020. Effectiveness of environmental management institutions in sustainable water resources management in the upper Pungwe River basin, Zimbabwe. *Physics and Chemistry of the Earth, Parts A/B/C,* 118-119, 102885.

Condon, L.A., Pietrasiak, N., Rosentreter, R. and Pyke, D.A., 2020. Passive restoration of vegetation and biological soil crusts following 80 years of exclusion from grazing across the Great Basin. *Restoration Ecology*, 28, 75-S85.

Convention on Biological Diversity, 2014. *Global Biodiversity Outlook 4*. CBD, Montreal, Quebec, Canada.

Coonan, E.C., Kirkby, C.A., Kirkegaard, J.A., Amidy, M.R., Strong, C.L. and Richardson, A.E., 2020. Microorganisms and nutrient stoichiometry as mediators of soil organic matter dynamics. *Nutrient Cycling in Agroecosystems*, 117(3), 273-298.

Costanza, R., Daly, L., Fioramonti, L., Giovannini, E., Kubiszewski, I., Mortensen, L.F., Pickett, K.E., Ragnarsdottir, K.V., De Vogli, R. and Wilkinson, R., 2016. Modelling and measuring sustainable wellbeing in connection with the UN Sustainable Development Goals. *Ecological Economics*, 130, 350-355.

Creswell, J.W., 2014. *Qualitative, quantitative and mixed methods approach.* Sage Publications, London, United Kingdom.

Cumming, D.H., 2008. Large scale conservation planning and priorities for the Kavango-Zambezi Transfrontier Conservation Area. Unpublished report commissioned by Conservation International.

Dauda, R.S., 2017. Poverty and economic growth in Nigeria: Issues and policies. *Journal of Poverty*, 21(1), 61-79.

Davis, M.A., 2009. *Invasion biology*. Oxford University Press on Demand. United Kingdom.

Day, M.D., Wiley, C.J., Playford, J. and Zalucki, M.P., 2003. *Lantana: Current management status and future prospects.* Australian Centre for International Agricultural Research, Australia.

De, W.A.H., Jani, V. and Webb, N.L., 2020. Disputes, relationships, and identity : A 'levels of conflict' analysis of human-wildlife conflict as human-human conflict in the mid-Zambezi valley, Northern Zimbabwe. *South African Geographical Journal,* 102(1), 59-76.

Deena, M.J. and Thoppil, J.E., 2000. Antimicrobial activity of the essential oil of *Lantana camara. Fitoterapia,* 71(4), 453-455.

Department for International Development [DFID], 1999. *Sustainable livelihoods guidance sheets.* Department for International Development, London, United Kingdom.

Diedrich, A., Benham, C., Pandihau, L. and Sheaves, M., 2019. Social capital plays a central role in transitions to sportfishing tourism in small-scale fishing communities in Papua New Guinea. *Ambio*, 48(4), 385-396.

Di Sacco, A., Hardwick, K.A., Blakesley, D., Brancalion, P.H., Breman, E., Cecilio Rebola, L., Chomba, S., Dixon, K., Elliott, S., Ruyonga, G. and Shaw, K., 2021. Ten golden rules for reforestation to optimize carbon sequestration, biodiversity recovery and livelihood benefits. *Global Change Biology*, 27(7), 1328-1348.

Dube, K., 2018. *Tourism and climate change: An investigation of the two-way linkages for the Victoria Falls resort, Zimbabwe*. Doctoral Thesis: University of South Africa, South Africa.

Dube, K. and Nhamo, G., 2020. Tourism business operators' perceptions, knowledge and attitudes towards climate change at Victoria Falls. *Transdisciplinary Research South Africa*, 16(1), 1-10.

Duncan, C.A., Jachetta, J.J., Brown, M.L., Carrithers, V.F., Clark, J.K., Ditomaso, J.M., Lym, R.G., McDaniel, K.C., Renz, M.J. and Rice, P.M., 2004. Assessing the economic, environmental, and societal losses from invasive plants on rangeland and wildlands. *Weed Technology,* 18, 1411-1416.

Dyderski, M.K. and Jagodziński, A.M., 2020. Impact of invasive tree species on natural regeneration species composition, diversity, and density. *Forests*, 11(4), 456.

Egoh, B.N., Ntshotsho, P., Maoela, M.A., Blanchard, R., Ayompe, L.M. and Rahlao, S., 2020. Setting the scene for achievable post-2020 convention on biological diversity targets: A review of the impacts of invasive alien species on ecosystem services in Africa. *Journal of Environmental Management,* 261, 110171.

Emery-Butcher, H.E., Beatty, S.J. and Robson, B.J., 2020. The impacts of invasive ecosystem engineers in freshwaters: A review. *Freshwater Biology*, 65(5), 999-1015.

Erckie, L., Adedoja, O., Geerts, S., van Wyk, E. and Boatwright, J.S., 2022. Impacts of an invasive alien Proteaceae on native plant species richness and vegetation structure. *South African Journal of Botany*, 144, 332-338.

Essl, F., Latombe, G., Lenzner, B., Pagad, S., Seebens, H., Smith, K., Wilson, J.R. and Genovesi, P., 2020. The Convention on Biological Diversity (CBD)'s post-2020 target on invasive alien species–what should it include and how should it be monitored? *NeoBiota*, 62, 99.

Fahad, S., Nguyen-Thi-Lan, H., Nguyen-Manh, D. and Tran-Duc, H., 2022. Analysing the status of multidimensional poverty of rural households by using sustainable livelihood framework: Policy implications for economic growth. *Environmental Science and Pollution Research*, 1-14.

Fall, J.J., 2021. What is an invasive alien species? Discord, dissent and denialism. In K. Barker and R.A. Francis (eds), *Routledge Handbook of Biosecurity and Invasive Species*, 40-54, Routledge, Oxfordshire, United Kingdom.

Faralli, M., Cristofolini, F., Cristofori, A., Ferretti, M. and Gottardini, E., 2022. Leaf trait plasticity and site-specific environmental variability modulate the severity of visible foliar ozone symptoms in *Viburnum lantana*. *PLoS One*, 17(7), 027020.

Ferreira-Rodríguez, N., Defeo, O., Macho, G. and Pardo, I., 2019. A social-ecological system framework to assess biological invasions: *Corbicula fluminea* in Galicia (NW Iberian Peninsula). *Biological Invasions*, 21(2), 587-602.

Francis, B., Pepler, D. and Ndagurwa, H.G., 2018. Woody species diversity and surface soil nutrients in an *Opuntia fulgida* Engelm invaded semi-arid savanna rangeland of south-west Zimbabwe. In S. Mutanga, T. Simelane, T. Gumbo and M. Mujuru (eds), *Africa at a crossroads: The continent's future 50 years after the formation of the Organization of African Unity/African Union*, 8-23. Africa Institute of South Africa, Pretoria.

French Embassy in Harare, 2021. *French presence in Zimbabwe*. Available: https://zw.ambafrance.org/CIRAD-Harare [2023, January 25].

Galatowitsch, S. and Bohnen, J., 2020. Predicting restoration outcomes based on organizational and ecological factors. *Restoration Ecology*, 28(5), 1201-1212.

Geesing, D., Al-Khawlani, A. and Abba, M.L., 2004. Management of introduced *Prosopis* species: Can economic exploitation control an invasive species? *Unasylva,* 217(55), 289-299.

Gentili, R., Schaffner, U., Martinoli, A. and Citterio, S., 2021. Invasive alien species and biodiversity: Impacts and management. *Biodiversity*, 22(1-2), 1-3.

Ginosar, A., 2021. Policy for the public without the public: Net neutrality in Israel. *Review of Policy Research,* 38(6), 661-676.

Guerbois, C., Dufour, A., Mtare, G. and Fritz, H., 2013. Insights for integrated conservation from attitudes of people toward protected areas near Hwange National Park, Zimbabwe: Immigration and integrated conservation. *Conservation Biology*, 27(4), 844-855.

Haregeweyn, N., Tsunekawa, A., Tsubo, M., Meshesha, D. and Melkie, A., 2013. Analysis of the invasion rate, impacts and control measures of *Prosopis juliflora*: A case study of Amibara District, Eastern Ethiopia. *Environmental Monitoring and Assessment*, 185(9), 7527-7542.

Haubrock, P., Cuthbert, R., Tricarico, E., Diagne, C., Courchamp, F. and Gozlan, R., 2021. The recorded economic costs of alien invasive species in Italy. *NeoBiota*, 67, 247-266.

Heck, S., Campos, H., Barker, I., Okello, J.J., Baral, A., Boy, E., Brown, L. and Birol, E., 2020. Resilient agri-food systems for nutrition amidst COVID-19: Evidence and lessons from food-based approaches to overcome micronutrient deficiency and rebuild livelihoods after crises. *Food Security*, 12(4), 823-830.

Heringer, G., Angulo, E., Ballesteros-Mejia, L., Capinha, C., Courchamp, F., Diagne, C., Duboscq-Carra, V., Nuñez, M. and Zenni, R., 2021. The economic costs of biological invasions in Central and South America: A first regional assessment. *NeoBiota*, 67, 401-426.

Heshmati, I., Khorasani, N., Shams-Esfandabad, B. and Riazi, B., 2019. Forthcoming risk of *Prosopis juliflora* global invasion triggered by climate change: Implications for environmental monitoring and risk assessment. *Environmental Monitoring and Assessment*, 191(2), 1-12.

Hill, M.P., Conlong, D., Zachariades, C., Coetzee, J.A., Paterson, I.D., Miller, B.E., Foxcroft, L. and Van Der Westhuizen, L., 2021. The role of mass-rearing in weed biological control projects in South Africa. *African Entomology,* 29(3), 1030-1044.

Hoffmann, J.H., Moran, V.C. and Hill, M.P., 2019. Conceptualizing, categorizing and recording the outcomes of biological control of invasive plant species, at a population level. *Biological Control*, 133, 134-137.

Holmes, T.P., Aukema, J.E., Von Holle, B., Liebhold, A. and Sills, E., 2009. Economic impacts of invasive species in forests. *Annals of the New York Academy of Sciences*, 1162(1), 18-38.

Hulme, P.E., 2006. Beyond control: Wider implications for the management of biological invasions. *Journal of Applied Ecology*, 43(5), 835-847.

Hulme, P.E., Brundu, G., Carboni, M., Dehnen-Schmutz, K., Dullinger, S., Early, R., Essl, F., González-Moreno, P., Groom, Q.J., Kueffer, C. and Kühn, I., 2018. Integrating invasive species policies across ornamental horticulture supply chains to prevent plant invasions. *Journal of Applied Ecology*, 55(1), 92-98.

Hussner, A., Stiers, I., Verhofstad, M.J.J.M., Bakker, E.S., Grutters, B.M.C., Haury, J., van Valkenburg, J.L.C.H., Brundu, G., Newman, J., Clayton, J.S., Anderson, L.W.J. and Hofstra, D., 2017. Management and control methods of invasive alien freshwater aquatic plants: A review. *Aquatic Botany*, 136, 112-137.

Ibrahim, M.A., Sabti, M.Z. and Mousa, S.H., 2021. In vitro accumulation potentials of heavy metals in big–sage (*Lantana camara*) plant. *DYSONA–Life Sciences*, 2(2), 2-17.

Ignatyeva, M., Yurak, V. and Logvinenko, O., 2020. A new look at the natural capital concept: approaches, structure, and evaluation procedure. *Sustainability,*12(21), 9236.

Israel, A., Einav, R., Silva, P.C., Paz, G., Chacana, M.E. and Douek, J., 2010. First report of the seaweed Codium parvulum (Chlorophyta) in Mediterranean waters: recent blooms on the northern shores of Israel. *Phycologia*, 49(2), 107-112.

James, E.H., 2000. Race-related differences in promotions and support: Underlying effects of human and social capital. *Organisation Science*, 11(5), 493-508.

Kabir, S., Kashem, M.A., Rahman, M.A., Hasan, M.M., Khan, M.A.A. and Hossain, M.Z., 2020. Variation in leaf traits of *Lantana camara* in relation to its adaptation with the soil condition of Bangladesh. *Journal of Advanced Plant Sciences*, 10(2), 37-44.

Kamwi, J.M., Endjala, J. and Siyambango, N., 2020. Dependency of rural communities on non-timber forest products in the dry lands of southern Africa: A case of Mukwe Constituency, Kavango East Region, Namibia. *Trees, Forestry and People,* 2, 100022.

Kannan, R., Shackleton, C.M., Krishnan, S. and Shaanker, R.U., 2016. Can local use assist in controlling invasive alien species in tropical forests? The case of *Lantana camara* in southern India. *Forest Ecology and Management*, 376, 166-173.

Kannan, R., Shackleton, C.M. and Shaanker, R.U., 2014. Invasive alien species as drivers in socio-ecological systems: Local adaptations towards use of *Lantana* in Southern India. *Environment, Development and Sustainability,* 16(3), 649-669.

Karki, S., 2021. Sustainable Livelihood Framework: Monitoring and evaluation. *International Journal of Social Sciences,* 8(1), 266-271.

Kasperson, J.X., Kasperson, R.E., Turner, B.L., Hsieh, W. and Schiller, A., 2022. Vulnerability to global environmental change. In R.E. Kasperson and J. Kasperson (eds), *The social contours of risk,* 245-285. Routledge, Oxfordshire, United Kingdom.

Kato-Noguchi, H. and Kurniadie, D., 2021. Allelopathy of *Lantana camara* as an invasive plant. *Plants*, 10(5), 1028.

Kent, M., 2011. *Vegetation description and data analysis: A practical approach*. Wiley and Sons, West Sussex, United Kingdom.

Keys, P.W., Galaz, V., Dyer, M., Matthews, N., Folke, C., Nyström, M. and Cornell, S.E., 2019. Anthropocene risk. *Nature Sustainability*, 2(8), 667-673.

Koech, G., Makokha, G.O. and Mundia, C.N., 2020. Climate change vulnerability assessment using a GIS modelling approach in ASAL ecosystem: A case study of Upper Ewaso Nyiro basin, Kenya. *Modeling Earth Systems and Environment*, 6(1), 479-498.

Kovács-Hostyánszki, A., Szigeti, V., Miholcsa, Z., Sándor, D., Soltész, Z., Török, E. and Fenesi, A., 2022. Threats and benefits of invasive alien plant species on pollinators. *Basic and Applied Ecology*, 64, 89-102.

Kull, C.A., Shackleton, C.M., Cunningham, P.J., Ducatillon, C., Dufour-Dror, J.-M., Esler, K.J., Friday, J.B., Gouveia, A.C., Griffin, A.R., Marchante, E., Midgley, S.J., Pauchard, A., Rangan, H., Richardson, D.M., Rinaudo, T., Tassin, J., Urgenson, L.S., von Maltitz, G.P., Zenni, R.D. and Zylstra, M.J., 2011. Adoption, use and perception of Australian acacias around the world. *Diversity and Distributions*, 17(5), 822-836.

Kumar, H., Pandey, B.W. and Anand, S., 2019. Analysing the impacts of forest ecosystem services on livelihood security and sustainability: A case study of Jim Corbett National Park in Uttarakhand. *International Journal of Geoheritage Parks*, 7(2), 45-55.

Kumar, M. and Garkoti, S.C., 2022. Allelopathy effects of invasive alien *Ageratina adenophora* on native shrub species of chir pine forest in the central Himalaya, India. *Journal of Forest Research,* 27(1), 53-62.

Kumar, T., Bishwas, A.J., Khare, P.K. and Garg, N., 2021. Invasive alien flora of tropical dry deciduous forest of Nauradehi Wildlife Sanctuary, Central India. *Indian Journal of Ecology*, 48(1), 219-225.

Lamb, N., Havens, K., Holloway, J., Steffen, J.F., Zeldin, J. and Kramer, A.T., 2022. Low passive restoration potential following invasive woody species removal in oak woodlands. *Restoration Ecology*, 30(4), 13568.

Lamsal, P., Kumar, L., Aryal, A. and Atreya, K., 2018. Invasive alien plant species dynamics in the Himalayan region under climate change. *Ambio*, 47(6), 697-710.

Lin, Z., Zengrang, X. and Wuxue, C., 2022. Ecological protection and livelihood improvement in ecologically vulnerable regions. *Journal of Resource Ecology*, 13(5), 759-762.

Lukey, P. and Hall, J., 2020. Biological invasion policy and legislation development and implementation in South Africa. In B. van Wilgen, J. Measey, D. Richardson, J. Wilson and T. Zengeya (eds), *Biological invasions in South Africa. Invading nature – Springer Series in Invasion Ecology, 14*, 515-512. Springer, Cham.

Lwanga-Ntale, C. and Owino, B.O., 2020. Understanding vulnerability and resilience in Somalia. *Jàmbá: Journal of Disaster Risk Studies*, 12(1).

Macdonald, I.A.W., Kruger, J.F. and Ferrar, A., 1986. *The ecology and management of biological invasion in South Africa*. Oxford University Press, United Kingdom.

Makoni, T. and Chikobvu, D., 2018. Modelling tourism demand volatility using a seasonal autoregressive integrated moving average autoregressive conditional heteroscedasticity model for Victoria Falls Rainforest arrivals in Zimbabwe. *Journal of Economics Finance Science*, 11(1), 1-9.

Makuvaza, S., 2012. Who owns the special area at Victoria Falls World Heritage Site? Political, organizational and governmental interests. *The Historic Environment: Policy and Practice*, 3(1), 42-63.

Makwana, N., 2019. Disaster and its impact on mental health: A narrative review. *Journal of Family Medicine and Primary Care*, 8(1), 3090.

Makwindi, N. and Ndlovu, J., 2021. Prospects and challenges of community-based tourism as a livelihood diversification strategy at Sehlabathebe National Park in Lesotho. *African Journal of Hospitality Tourism Leisure*, 10(1), 333-348.

Mamun, M.S.A. and Ahmed, M., 2011. Prospect of indigenous plant extracts in tea pest management. *International Journal of Agricultural Research, Innovation and Technology*, 1(1-2), 16-23.

Maroyi, A., 2022. Alien flora of Zimbabwe: Data derived from herbarium specimens. *Data in Brief*, 42, 108186.

Martin, S.M. and Lorenzen, K., 2016. Livelihood diversification in rural Laos. *World Development*, 83, 231-243.

Masocha, M., 2010. *Savanna aliens.* Doctoral Thesis: Wageningen University, Netherlands.

McConnachie, M.M., Cowling, R.M., van Wilgen, B.W. and McConnachie, D.A., 2012. Evaluating the cost-effectiveness of invasive alien plant clearing: A case study from South African. *Biology Conservation*, 155, 128-135.

Mehdi, A. and Chaudhry, D., 2015. Human capital potential of India's future workforce. *ICRIER Working paper 308*. Indian Council for Research on International Economic Relations, India.

Mhlanga, N. and Pisa, L.S., 2022. Impacts of the Agrarian Land Reform Policy on livelihoods of resettled communities adjust to the Hwange National Park, Zimbabwe. *International Science and Technology Journal of Namibia*, 15, 57-68.

Mhlongo, E.S., 2021. *Perceptions of the role of* Lantana camara *on human well-being and rural livelihoods in Vhembe Biosphere Reserve, South Africa.* Doctoral Thesis: Rhodes University, South Africa.

Montagnani, C., Gentili, R., Brundu, G., Caronni, S. and Citterio, S., 2022. Accidental introduction and spread of top invasive alien plants in the European Union through human-mediated agricultural pathways: What should we expect? *Agronomy*, 12(2), 423.

Moore, H.E. and Rutherfurd, I.D., 2017. Lack of maintenance is a major challenge for stream restoration projects. *River Research and Applications*, 33(9), 1387-1399.

Moyo, B. and Ravhuhali, K.E., 2022. Abandoned croplands: Drivers and secondary succession trajectories under livestock grazing in communal areas of South Africa. *Sustainability*, 14(10), 6168.

Mthethwa, S. and Wale, E., 2020. Household vulnerability to food insecurity in rural South Africa: Evidence from a nationally representative survey data. *International Journal of Environmental Research and Public Health*, 18(4), 1917.

Mugwedi, L., 2020. Harnessing opportunities provided by the invasive *Chromolaena odorata* to keep it under control. *Sustainability*, 12(16), 6505.

Mujaju, C., Mudada, N. and Chikwenhere, G.P., 2021. Invasive alien species in Zimbabwe (Southern Africa). In T. Pullaiah and M.R. Ielmini (eds), *Invasive alien species: Observations and issues from around the world*, *1*, 330-361. John Wiley and Sons.

Mukherjee, S., Banerjee, S., Basu, P., Saha, G.K. and Aditya, G., 2015. *Lantana camara* and butterfly abundance in an urban landscape: Benefits for conservation or species invasion? *Ekológia,* 34(4), 309.

Mungi, N.A., Qureshi, Q. and Jhala, Y.V., 2020. Expanding niche and degrading forests: Key to the successful global invasion of *Lantana camara* (sensu lato). *Global Ecology and Conservation*, 23, 01080.

Mushawemhuka, W.J., Fitchett, J.M. and Hoogendoorn, G., 2021. Towards quantifying climate suitability for Zimbabwean nature-based tourism. *South African Geography*, 103(4), 443-463.

Muvengwi, J. and Ndagurwa, H.G.T., 2015. Soil seed bank dynamics and fertility on a seasonal wetland invaded by *Lantana camara* in a savanna ecosystem. *South African Journal of Botany*, 100, 190-194.

Mwakiwa, E., Maparara, T., Tatsvarei, S. and Muzamhindo, N., 2018. Is community management of resources by urban households, feasible? Lessons from community gardens in Gweru, Zimbabwe. *Urban Forestry & Urban Greening*, 34, 97-104.

Natarajan, N., Newsham, A., Rigg, J. and Suhardiman, D., 2022. A sustainable livelihoods framework for the 21st century. *World Development*, 155, 105898.

Ncube, B., Shekede, M.D., Gwitira, I. and Dube, T., 2020. Spatial modelling the effects of climate change on the distribution of *Lantana camara* in Southern Zimbabwe. *Applied Geography*, 117, 102172.

Ndebele-Murisa, M. and Mubaya, C., 2015. Climate change: Impact on agriculture, livelihood options and adaptation strategies for smallholder farmers in Zimbabwe. In T. Murisa and T. Chikweche (eds), *Beyond the crises*: *Zimbabwe's prospects for transformation*, 155-198. Trust Africa and Weaver Press.

Negi, G., Sharma, S., Vishvakarma, S.C., Samant, S.S., Maikhuri, R.K., Prasad, R.C. and Palni, L., 2019. Ecology and use of *Lantana camara* in India. *The Botanical Review*, 85(2), 109-130.

Ngorima, A. and Shackleton, C.M., 2019. Livelihood benefits and costs from an invasive alien tree (*Acacia dealbata*) to rural communities in the Eastern Cape, South Africa. *Journal of Environmental Management*, 229, 158-165.

Nguyen, N.B.C., Nguyen, B.Q. and Dong, T.C.T., 2019. Antifeedant activity of essential oil *Lantana camara* against *Spodoptera litura* and *Plutella xylostella*. *University Journal of Science*, 11(1), 1-6.

Nhemachena, C., Mano, R., Mudombi, S. and Muwanigwa, V., 2014a. Climate change adaptation for rural communities dependent on agriculture and tourism in marginal farming areas of the Hwange District, Zimbabwe. *African Journal of Agricultural Research*, 9(26), 2045-2054.

Nhemachena, C., Mano, R., Mudombi, S. and Muwanigwa, V., 2014b. Perceptions on climate change and its impact on livelihoods in Hwange District, Zimbabwe. *Jàmbá: Journal of Disaster Risk Studies*, 6(1), 1-6.

Niehof, A., 2004. The significance of diversification for rural livelihood systems. *Food Policy*, 29(4), 321-338.

Nordberg, K., Mariussen, Å. and Virkkala, S., 2020. Community-driven social innovation and quadruple helix coordination in rural development. Case study on leader group Aktion Österbotten. *Journal of Rural Studies*, 79, 157-168.

Novoa, A., Kaplan, H., Wilson, J.R.U. and Richardson, D.M., 2016. Resolving a prickly situation: Involving stakeholders in invasive cactus management in South Africa. *Environment Management,* 57(5), 998-1008.

Ntalo, M., Ravhuhali, K.E., Moyo, B., Hawu, O. and Msiza, N.H., 2022. *Lantana camara*: Poisonous species and a potential browse species for goats in Southern Africa—A review. *Sustainability*, 14(2), 751.

O'Connor, T.G. and van Wilgen, B.W., 2020. The impact of invasive alien plants on rangelands in South Africa. *Biological Invasions in South Africa*, 14, 459-487.

Ojija, F., 2020. *The distribution, control techniques and impact of* Parthenium hysterophorus *on flower visitors and soil chemical properties in Meru District, Tanzania*. Doctoral Thesis: Mbeya University of Science and Technology, Tanzania.

O'Riordan, T. and Stoll-Kleemann, S. (eds), 2002. *Biodiversity, sustainability and human communities: Protecting beyond the protected*. Cambridge University Press, United Kingdom.

Pahl-Wostl, C., 2009. A conceptual framework for analysing adaptive capacity and multi-level learning processes in resource governance regimes. *Global Environmental Change,* 19(3), 354-365.

Patel, S., 2011. A weed with multiple utility*: Lantana camara*. *Reviews in Environmental Science and Biological Technology*, 10(4), 341-351.

Pergl, J., Brundu, G., Harrower, C.A., Cardoso, A.C., Genovesi, P., Katsanevakis, S., Lozano, V., Perglová, I., Rabitsch, W., Richards, G., Roques, A., Rorke, S.L., Scalera, R., Schönrogge, K., Stewart, A., Tricarico, E., Tsiamis, K., Vannini, A., Vilà, M., Zenetos, A. and Roy, H.E., 2020. Applying the convention on biological diversity pathway classification to alien species in Europe. *NeoBiota*, 62, 333-363.

Pimentel, D., Zuniga, R. and Morrison, D., 2005. Update on the environmental and economic costs associated with alien-invasive species in the United States. *Ecology and Economics in Control Bioinvasions*, 52(3), 273-288.

Potgieter, L.J, Shrestha, N. and Cadotte, M.W., 2022. Prioritising terrestrial invasive alien plant species for management in Urban ecosystems. *Journal of Applied Ecology*, 59, 872-883.

Prach, K., Šebelíková, L., Řehounková, K. and del Moral, R., 2020. Possibilities and limitations of passive restoration of heavily disturbed sites. *Landscape Research*, 45, 247-253.

Pratt, C.F., Constantine, K.L. and Murphy, S.T., 2017. Economic impacts of invasive alien species on African smallholder livelihoods. *Global Food Security*, 14, 31-37.

Pyšek, P. and Richardson, D.M., 2010. Invasive species, environmental change and management, and health. *Annual review of environment and resources*, 35, 25-55.

Pyšek, P., Hulme, P.E., Meyerson, L.A., Smith, G.F., Boatwright, J.S., Crouch, N.R., Figueiredo, E., Foxcroft, L.C., Jarošík, V., Richardson, D.M. and Suda, J., 2013. Hitting the right target: Taxonomic challenges for, and of, plant invasions. *AoB Plants*, 5, plt042.

Pyšek, P., Hulme, P.E., Simberloff, D., Bacher, S., Blackburn, T.M., Carlton, J.T., Dawson, W., Essl, F., Foxcroft, L.C., Genovesi, P. and Jeschke, J.M., 2020. Scientists' warning on invasive alien species. *Biological Reviews*, 95(6), 1511-1534.

Radić, B. and Gavrilović, S., 2021. Natural habitat loss: Causes and implications of structural and functional changes. In W. Leal Filho, A. Azul, L. Brandli, A. Lange Salvia and T. Wall (eds), *Life on land. Encyclopedia of the UN Sustainable Development Goals,* 699-712. Springer, Cham.

Rahman, M.M., 2023. Assessing the role of invasive *Lantana* (*Lantana camara*) in disrupting the native biodiversity of Bangladesh. *Social Science Research Network*, April 6, 4411276.

Rai, P.K. and Singh, J.S., 2020. Invasive alien plant species: Their impact on the environment, ecosystem services and human health. *Ecological Indicators*, 111, 106020.

Raj, R., Das, T.K., Kaur, R., Singh, R. and Shekhawat, K., 2018. Invasive noxious weed management research in India with special reference to *Cyperus rotundus*, *Eichhornia crassipes* and *Lantana camara*. *Indian Journal of Agriculture,* 88(2), 181-196.

Rakodi, C., 2002. A livelihoods approach – Conceptual issues and definitions. In T. Lloyd-Jones and C. Rakodi, *Urban livelihoods*, 26-45. Routledge, London, United Kingdom.

Rebus, A., 2019. *A wealth of life. Victoria Falls*. Weigl Publications, Calgary, Canada.

Reynolds, C., Venter, N., Cowie, B.W., Marlin, D., Mayonde, S., Tocco, C. and Byrne, M.J., 2020. Mapping the socio-ecological impacts of invasive plants in South Africa: Are poorer households with high ecosystem service use most at risk? *Ecosystem Services*, 42, 101075.

Richardson, D.M. and Rejmánek, M., 2011. Trees and shrubs as invasive alien species–a global review. *Diversity and distributions*, 17(5), 788-809.

Rogers, P., Nunan, F. and Fentie, A.A., 2017. Reimagining invasions: The social and cultural impacts of *Prosopis* on pastoralists in southern Afar, Ethiopia. *Pastoralism*, 7, 1-13.

Rojas-Sandoval, J., Ackerman, J.D., Marcano-Vega, H. and Willig, M.R., 2022. Alien species affect the abundance and richness of native species in tropical forests: The role of adaptive strategies. *Ecosphere*, 13(12), 4291.

Ruwanza, S. and Shackleton, C.M., 2016. Effects of the invasive shrub, *Lantana camara*, on soil properties in the Eastern Cape, South Africa. *Weed Biology and Management*, 16(2), 67-79.

Salerno, J., Stevens, F.R., Gaughan, A.E., Hilton, T., Bailey, K., Bowles, T., Cassidy, L., Mupeta-Muyamwa, P., Biggs, D., Pricope, N. and Mosimane, A.W., 2021. Wildlife impacts and changing climate pose compounding threats to human food security. *Current Biology*, 31(22), 5077-5085.

Schmerbeck, J. and Gupta, S., 2022. *Lantana camara* does not exclude all tree regeneration: An opportunity to help establish mixed forests in the Western Himalaya? *Forest Ecology and Management*, 520, 120366.

Schweizer, D., van Kuijk, M. and Ghazoul, J., 2021. Perceptions from non-governmental actors on forest and landscape restoration, challenges and strategies for successful implementation across Asia, Africa and Latin America. *Journal of Environmental Management*, 286, 112251.

Scoones, I. 1998. Sustainable Rural Livelihoods: A framework for analysis. *IDS Working Paper 72*. Institute of Development Studies, Brighton, United Kingdom.

Sembiring, Y.S. and Zuhra, C.F., 2021. Analysis of chemical components and antibacterial activity in essential oil of *Lantana* flowers (*Lantana camara)*. *Biomedika*, 14(1), 19-28.

Serrat, O., 2017a. *Knowledge solutions: Tools, methods, and approaches to drive organizational performance*, 1140. Springer Nature.

Serrat, O., 2017b. The sustainable livelihoods approach. In O. Serrat, *Knowledge solutions: Tools, methods, and approaches to drive organizational performance*, 21–26. Springer, Asian Development Bank, Mondaluyong, Philippines.

Shackleton, C.M., McGarry, D., Fourie, S., Gambiza, J., Shackleton, S.E. and Fabricius, C., 2007. Assessing the effects of invasive alien species on rural livelihoods: Case examples and a framework from South Africa. *Human Ecology*, 35(1), 113-127.

Shackleton, R.T., Le Maitre, D.C., Pasiecznik, N.M. and Richardson, D.M., 2014. *Prosopis*: A global assessment of the biogeography, benefits, impacts and management of one of the world's worst woody invasive plant taxa. *AoB Plants,* 6.

Shackleton, R.T., Le Maitre, D.C. and Richardson, D.M., 2015. Prosopis invasions in South Africa: Population structures and impacts on native tree population stability. *Journal of Arid Environments*, 114, 70-78.

Shackleton, R.T., Novoa, A., Shackleton, C.M. and Kull, C.A., 2020. The social dimensions of biological invasions in South Africa. In B.W. van Wilgen, J. Meassey, D.M. Richardson, J.R. Wilson and T.A. Zengeya (eds), *Biological invasions in South Africa*, 701-729. Springer, Cham.

Shackleton, R.T., Richardson, D.M., Shackleton, C.M., Bennett, B., Crowley, S.L., Dehnen-Schmutz, K., Estévez, R.A., Fischer, A., Kueffer, C., Kull, C.A. and Marchante, E., 2019. Explaining people's perceptions of invasive alien species: A conceptual framework. *Journal of Environmental Management*, 229, 10-26.

Shackleton, R.T., Shackleton, C.M. and Kull, C.A., 2019. The role of invasive alien species in shaping local livelihoods and human well-being: A review. *Journal of Environmental Management*, 229, 145-157.

Shackleton, R.T., Witt, A.B., Nunda, W. and Richardson, D.M., 2017. *Chromolaena odorata* (Siam weed) in eastern Africa: Distribution and socio-ecological impacts. *Biological Invasions*, 19, 1285-1298.

Shannon, C., Stebbing, P.D., Dunn, A.M. and Quinn, C.H., 2020. Getting on board with biosecurity: Evaluating the effectiveness of marine invasive alien species biosecurity policy for England and Wales. *Marine Policy*, 122, 104275.

Sharma, B. and Kumar, P., 2009. Bioefficacy of *Lantana camara* against some human pathogens. *Indian Journal of Pharmaceutical Science*, 71(5), 589.

Shaw, P., Jobe, J. and Gedan, K.B., 2022. Environmental limits on the spread of invasive *Phragmites australis* into upland forests with marine transgression. *Estuaries and Coasts*, 45(2), 539-550.

Sheppard, A.W., Paynter, Q., Mason, P., Murphy, S., Stoett, P., Cowan, P., Brodeur, J., Warner, K., Villegas, C., Shaw, R., Hinz, H., Hill, M. and Genovesi, P., 2019. IUCN SSC Invasive Species Specialist Group. The application of biological control for the management of established invasive alien species causing environmental impacts. *Convention on Biological Diversity Technical Series*. *No. 91*. The Secretariat of the Convention on Biological Diversity, Montreal, Canada.

Shiferaw, W., Demissew, S. and Bekele, T., 2018. Invasive alien plant species in Ethiopia: Ecological impacts on biodiversity: A review paper. *International Journal of Molecular Biology*, 3(4),171-178.

Shoeb-Ur-Rahman, M., Simmons, D.G., Shone, M. and Ratna, N., 2020. Co-management of capitals for community wellbeing and sustainable tourism development: A conceptual framework. *Tourism Planning and Development*, 17(2), 225-236.

Shrestha, U.B. and Shrestha, B.B., 2019. Climate change amplifies plant invasion hotspots in Nepal. *Diversity and Distributions*, 25(10), 1599-1612.

Shyamna, K. and Nasser, M., 2019. *Altitudinal difference in insect pollinators of* Lantana camara*, an invasive plant in southern Western Ghats*. Insect Ecology and Ethology Laboratory, Department of Zoology, University of Calicut, Thenhipalam, Kerala.

Siges, T.H., Hartemink, A.E., Hebinck, P. and Allen, B.J., 2005. The invasive shrub *Piper aduncum* and rural livelihoods in the Finschhafen area of Papua New Guinea. *Human Ecology*, 33(6), 875-893.

Sileshi, G.W., Gebeyehu, S. and Mafongoya, P.L., 2019. The threat of alien invasive insect and mite species to food security in Africa and the need for a continent-wide response. *Food Security*, 11(4), 763-775.

Simelane, D.O., Katembo, N. and Mawela, K.V., 2021. Current status of biological control of *Lantana camara* in South Africa. *African Entomology*, 29(3), 775-783.

Singh, K., Byun, C. and Bux, F., 2022. Ecological restoration of degraded ecosystems in India: Science and practices. *Ecological Engineering*, 182, 106708.

Sintayehu, D.W., Dalle, G. and Bobasa, A.F., 2020. Impacts of climate change on current and the future invasion of *Prosopis juliflora* in Ethiopia: Environmental and socio-economic implications. *Heliyon*, 6(8), 04596.

Skoufias, E., 2001. *Progresa and its impact on the human capital and welfare of household in rural Mexico: A synthesis of the results of an evaluation.* International Food Policy Research Institute, Washington, USA.

Sohrabi, S., Vila, M., Zand, E., Gherekhloo, J. and Hassanpour-Bourkheili, S., 2023. Alien plants of Iran: Impacts, distribution and managements. *Biological Invasions*, 25(1), 97-114.

Souza-Alonso, P., Rodríguez, J., González, L. and Lorenzo, P., 2017. Here to stay. Recent advances and perspectives about *Acacia* invasion in Mediterranean areas. *Annals of Forest Science,* 74(3), 1-20.

Stoldt, M., Göttert, T., Mann, C. and Zeller, U., 2020. Transfrontier conservation areas and human-wildlife conflict: The case of the Namibian component of the Kavango-Zambezi (KAZA) TFCA. *Scientific Reports*, 10(1), 1-16.

Stoops, G., Van Ranst, E. and Verbeek, K., 2001. Pedology of soils within the spray zone of the Victoria Falls (Zimbabwe). *Catena*, 46(1), 63-83.

Tambe, S., 2022. Sustainable Livelihoods Approach. In S. Tambe, *Teaching and Learning Rural Livelihoods,* 45-56. Springer, Cham.

Tambo, G., Chabikwa, G.T., Daga, O. and Dhlomo, J.L., 2021. *Livestock production system in Ward 12, Tsholotsho District, Zimbabwe: Characterization and potential productivity enhancement*. Matopos Research Institute, Zimbabwe.

Tarugara, A., Clegg, B.W. and Matuvhunye, R., 2022. Mapping the spatial distribution and canopy cover of *Lantana camara* in the Zaka District of Zimbabwe. *Scientific African,* 17, 01339.

Teixeira, J.C. and Huber, C.D., 2021. The inflated significance of neutral genetic diversity in conservation genetics. *Proceedings of the National Academy of Sciences*, 118, 10.

Tendai, M. and Chikobvu, D., 2017. Modelling international tourist arrivals and volatility to the Victoria Falls Rainforest, Zimbabwe: Application of the GARCH family of models. *African Journal Hospitality Tourism Leisure*, 6(4), 1-16.

Thomas, S.E., Evans, H.C., Cortat, G., Koutsidou, C., Day, M.D. and Ellison, C.A., 2021. Assessment of the macrocyclic rust *Puccinia lantanae* as a classical biological control agent of the pantropical weed *Lantana camara*. *Biological Control*, 160, 104688.

Thondhlana, G. and Muchapondwa, E., 2014. Dependence on environmental resources and implications for household welfare: Evidence from the Kalahari drylands, South Africa. *Ecological Economics*, 108, 59-67.

Tonello, K.C., Pereira, L.C., Balbinot, L., Nnadi, E.O., Mosleh, M.H. and Bramorski, J., 2022. Patterns of litter and nutrient return to the soil during passive restoration in Cerrado, Brazil. *Biologia*, 1-16.

Tsiamis, K., Azzurro, E., Bariche, M., Çinar, M.E., Crocetta, F., De Clerck, O., Galil, B., Gómez, F., Hoffman, R., Jensen, K.R. and Kamburska, L., 2020. Prioritizing marine invasive alien species in the European Union through horizon scanning. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 30(4),794-845.

Tsiamis, K., Gervasini, E., Deriu, I., D'amico, F., Nunes, A., Addamo, A. and Cardoso, A.C., 2017. *Baseline distribution of invasive alien species of Union concern*. Publications Office of the European Union, Ispra, Italy.

Turbelin, A. and Catford, J.A., 2021. Invasive plants and climate change. In T.M. Letcher (ed.), *Climate change*, 515-539. Elsevier.

Turbelin, A.J., Malamud, B.D. and Francis, R.A., 2017. Mapping the global state of invasive alien species: patterns of invasion and policy responses. *Global Ecology and Biogeography*, 26(1), 78-92.

Urban, B. and Mutendadzamera, K., 2022. Social capital leading to innovation: understanding moderating effects of the environment in the Zimbabwean small and medium enterprise context. *Journal of Enterprising Communities: People and Places in the Global Economy*, 16, 4, 631-652.

Uyi, O., Mukwevho, L., Ejomah, A.J. and Toews, M., 2021. Invasive alien plants in Sub-Saharan Africa: A review and synthesis of their insecticidal activities. *Frontiers in Agronomy*, 3, 725895.

van Wilgen, B.W., 2018. The management of invasive alien plants in South Africa: Strategy, progress and challenges. *Outlooks on Pest Management*, 29(1),13-17.

van Wilgen, B.W., Cowling, R.M., McConnachie, M., Marais, C., Sharp, D. and Esler, K.J., 2012. Challenges in invasive alien plant control in South Africa: news and views. *South African Journal of Science*, 108(11), 1-3.

van Wilgen, B.W., Raghu, S., Sheppard, A.W. and Schaffner, U., 2020. Quantifying the social and economic benefits of the biological control of invasive alien plants in natural ecosystems. *Current Opinion in Insect Science*, 38, 1-5.

van Wilgen, B.W., Zengeya, T.A. and Richardson, D.M., 2022. A review of the impacts of biological invasions in South Africa. *Biological Invasions*, 24(1), 27-50.

van Wilgen, N.J., Faulkner, K.T. and Robinson, T.B., South, J., Beckett, H., Janion-Scheepers, C., Measey, G.F. and Richardson, D.M., 2023. Climate change and biological invasions in South Africa. In L. Ziska (ed.) *Invasive species and global climate change*, 158-187. 2nd edition. CAB International.

Vardien, W., Richardson, D.M., Foxcroft, L.C., Thompson, G.D., Wilson, J.R.U. and Le Roux, J.J., 2012. Invasion dynamics of *Lantana camara* in South Africa. *South African Journal of Botany,* 81, 81-94.

Vera, I., Goosen, N., Batidzirai, B., Hoefnagels, R. and van der Hilst, F., 2022. Bioenergy potential from invasive alien plants: Environmental and socio-economic impacts in Eastern Cape, South Africa. *Biomass and Bioenergy*, 15*,* 106340.

Vimercati, G., Kumschick, S., Probert, A.F., Volery, L. and Bacher, S., 2020. The importance of assessing positive and beneficial impacts of alien species. *NeoBiota*, 62, 525.

Vorlaufer, K., 2002. CAMPFIRE - The political ecology of poverty alleviation, wildlife utilisation and biodiversity conservation in Zimbabwe. *Erdkunde,* 56(2), 184-206.

Vujanović, D., Losapio, G., Milić, S. and Milić, D., 2022. The impact of multiple species invasion on soil and plant communities increases with invasive species co-occurrence. *Frontiers in Plant Science*, 13, 875824.

Waller, L.P., Allen, W.J., Barratt, B.I.P., Condron, L.M., França, F.M., Hunt, J.E., Koele, N., Orwin, K.H., Steel, G.S., Tylianakis, J.M., Wakelin, S.A. and Dickie, I.A., 2020. Biotic interactions drive ecosystem responses to exotic plant invaders. *Science,* 368(6494), 967-972.

Wang, R., Cresswell, T., Johansen, M.P., Harrison, J.J., Jiang, Y., Keitel, C., Cavagnaro, T.R. and Dijkstra, F.A., 2021. Reallocation of nitrogen and phosphorus from roots drives regrowth of grasses and sedges after defoliation under deficit irrigation and nitrogen enrichment. *Journal of Ecology*, 109(12), 4071-4080.

Wekhanya, M.N., Mbugua, P.K. and Mworia, J.K., 2020. The effect of invasive species *Lantana camara* L. on soil nutrients at Ol-Donyo Sabuk National Park, Kenya. *African Journal of Pure and Applied Sciences*, 1(1), 25-32.

White, J.D., Stevens, N., Fisher, J.T., Archibald, S. and Reynolds, C., 2022. Nature-reliant, low-income households face the highest rates of woody-plant encroachment in South Africa. *People and Nature*, 4(4), 1020-103.

Williams-Linera, G., Bonilla-Moheno, M., López-Barrera, F. and Tolome, J., 2021. Litterfall, vegetation structure and tree composition as indicators of functional recovery in passive and active tropical cloud forest restoration. *Forest Ecology and Management,* 493, 119260.

Yiridomoh, G.Y., Sullo, C. and Bonye, S.Z., 2021. Climate variability and rural livelihood sustainability: Evidence from communities along the Black Volta River in Ghana. *Geography Journal*, 86(4), 1527-1543.

Yletyinen, J., Perry, G.L., Burge, O.R., Mason, N.W. and Stahlmann-Brown, P., 2021. Invasion landscapes as social-ecological systems: Role of social factors in invasive plant species control. *People and Nature*, 3(4), 795-810.

Zhang, D., Wang, W., Zhou, W., Zhang, X. and Zuo, J., 2020. The effect on poverty alleviation and income increase of rural land consolidation in different models: A China study. *Land Use Policy,* 99, 104989.

Zhang, Y., Leng, Z., Wu, Y., Jia, H., Yan, C., Wang, X., Ren, G., Wu, G. and Li, J., 2022. Interaction between nitrogen, phosphorus, and invasive alien plants. *Sustainability*, 14(2), 746.

Zhao, X., Chen, H., Zhao, H. and Xue, B., 2022. Farmer households' livelihood resilience in ecological-function areas: The case of the Yellow River water source area of China. *Environment, Development and Sustainability*, 24(7), 9665-9686.

Zhou, Z. 2018. A survey of visitor satisfaction at Victoria Falls Rainforest. *African Journal of Hospitality Tourism Leisure*, 7(1), 1-21.

Zimbabwe Vulnerability Assessment Committee [ZimVAC], 2020. *Zimbabwe Vulnerability Assessment Committee (ZimVAC): 2020 Rural Livelihoods Assessment Report*. Technical Report. Food and Nutrition Council, Harare, Zimbabwe.

Zivenge, E., Shiferaw-Mitiku, T., Thomas, J. and Ushadevi, K.N., 2013. Economic performance of community garden in Zimbabwe. *Russian Journal of Agricultural and Socio-Economic Sciences*, 21(9), 18-22.

Chapter 2: Impacts of *Lantana camara* invasion on native woody species and soil nutrients in Kavango Zambezi Transfrontier Conservation Area

Abstract

Invasive alien species such as *Lantana camara* L. have ecological impacts that affect native species and soil conditions in different areas. The context specific ecological effects of *L. camara* to native woody species communities remain poorly understood yet such explorations are essential to maintain biodiversity. Therefore, a study was conducted in the Kavango Zambezi Transfrontier Conservation Area (KAZA TFCA), located in Zimbabwe, which assessed the ecological impacts of *L. camara* on native woody species and soil nutrients. Stratified random sampling was used to select 60 plots in the Communal and Park areas in invaded and uninvaded sites based on *L. camara* invasion. In each plot, vegetation was assessed with respect to stem counts, species identification, mean canopy diameter, diameter at breast height, and height. Soil samples were collected and analysed for nitrogen, organic carbon, phosphorus, potassium, and pH. The results showed that *L. camara* negatively affected the native species, with a significant decrease in species richness, density, height, and canopy cover in the invaded as compared to the uninvaded plots. However, invaded plots had significantly higher pH (6.1) and soil nutrients such as phosphorus and organic carbon than uninvaded plots. Canonical Correspondence Analysis revealed that soil variables influenced native woody species composition. The study concludes that *L. camara* invasion affected native vegetation diversity and growth. The findings have implications for the management strategies of *L. camara* and other invasive alien plants in KAZA TFCA. The study recommends a need to tailor-make a management strategy for *L. camara* that is suitable for the area, and considers the land-use in the area. The study also recommends further research into specific mechanisms of how *L. camara* impacts native vegetation and soil properties.

Keywords: Communal area; Invaded; Native woody species; Park; Soil nutrients; Uninvaded

2.1. Introduction

One of the major drivers affecting biodiversity globally is Invasive Alien Species (IAS) (Holmes et al., 2019; Tererai et al., 2015). Among IAS are Invasive Alien Plants (IAPs) which are a major biodiversity threat undermining conservation status of some habitats (Lazzaro et al., 2020). One such IAP is *Lantana camara* L., an evergreen, aromatic shrub, belonging to the Verbenaceae family (Patel, 2011). It thrives in disturbed environments (Dube et al., 2020) and can dominate and affect the growth of the native vegetation.

L. camara occurs in multiple land covers including forests and woodland areas (Niphadkar et al., 2017). However, *L. camara* rarely invades intact forests and cannot survive in forests with dense canopies unless there has been some form of disturbance that has created gaps (Negi et al., 2019). Although *L. camara* often grows in pure stands, to a height of between one to two metres, it is also capable of growing among a diversity of other plant species (Muvengwi and Ndagurwa, 2015; Ruwanza, 2016). *Lantana camara* can easily naturalise easily in different parts of the world such as Australia, India and parts of Africa, with attendant impacts (Bhagwat et al., 2012; Goncalves et al., 2014; Paul et al., 2021).

The impacts of *L. camara* include allelopathic effects, thereby hindering the regeneration and the growth of neighbouring plants (Fan et al., 2010). *Lantana camara* invasion often decreases native species richness and diversity, plant density, and seedling recruitment (Aravindhan and Rajendran, 2014; Gooden et al., 2009; Sundaram and Hiremath, 2012). Ruwanza (2020) also showed that *L. camara* contributes to major shifts in vegetation structure of trees and shrubs as compared to herbs and grasses that are native to the area. For example, *L. camara* is a threat to native species through reduction in species richness of ferns, herb, tree and vine species (Ruwanza, 2020), and a rapid decline in native species richness of these major life forms was observed when there was a dense *L. camara* infestation in southeastern Australia (i.e., above 75% cover) (Gooden et al. 2009).

The impact of *L. camara* on native vegetation has secondary effects on other species, such as sable antelope (*Hippotragus niger*) in Kenya, by threatening their habitat (Nanjappa et al., 2005). Also, *L. camara* infestation resulted in a decline in insectivorous birds and canopy birds due to habitat alterations in India (Aravind et al., 2010). The impact on native vegetation by *L. camara* is also increased by the greater risk of fires in invaded sites, as *L. camara* is highly flammable (Berry et al., 2011). Berry et al. (2011) reported that in the dry forests densely invaded by *L. camara*, there was an increase in leaf litter depths, fuel bed depths, and percentage cover by fuels, which increases the ignition potential in Australia. Similarly, Hiremath and Sundaram (2005) suggested that there was positive feedback between *L. camara* establishment and present-day fires that impacted the forest composition and functionality in India.

In the case of soil ecosystems, *L. camara* typically alters soil properties such as pH and mineral composition which might favour its growth (Osunkoya and Perrett, 2011; Simba et al., 2013; Tererai et al., 2015). Ruwanza and Shackleton (2016) showed how *L. camara* increased soil properties, such as total carbon (C), resulting in changes in soil physico-chemical properties which are likely to affect native species growth, yet favour spread of *L. camara*. An increase in soil nutrients such as calcium (Ca), magnesium (Mg), potassium (K), as well as pH, have been attributed to favour the growth of *L. camara* (Simba et al., 2013). Mahla and Mlambo (2019) also found that the presence of *L. camara* was associated with increase in soil C, nitrogen (N) and pH. Contrary to the above assertions, Kisaakye (2021) reported that *L. camara* invasion does not modify the soil pH, but invades places with favourable soil pH.

Despite several studies on the impacts of *L. camara* on native vegetation and soil properties, Dassonville et al. (2008) state that it is difficult to make generalisations about IAS because impacts often appear to be site- and species-specific. Native woody species form a significant component of biodiversity and offer various ecosystem services that contribute to local livelihoods (Nacoulma et al., 2011). In the Western part of Zimbabwe, the Hwange District is part of the Kavango Zambezi Transfrontier Conservation Area (KAZA TFCA) where concerns about the spread of *L. camara* have been raised by local leadership and Rural District Council officials. The spread of *L. camara* in the KAZA TFCA potentially poses a threat to some native woody species, with implications for both the ecosystem services and the ecological functioning and resilience.

To the best of my knowledge, there is limited research (approximately less than five studies in different parts of Zimbabwe, and none within the KAZA TFCA) on the direct impacts of *L. camara* invasion on native woody species composition and structure in Zimbabwe. This study provides additional research on this issue. Additionally, this study offers an opportunity to examine the ability of native woody species to resist invasion by assessing the diversity and richness in the invaded and uninvaded areas. The study, therefore, focused on determining the impacts of *L. camara* invasion on the native woody species composition and structure, and on soil nutrients, at KAZA TFCA. The study addressed two main research questions which were: (i) What is the impact of *L. camara* invasion on native woody species composition and structure in the KAZA TFCA, and (ii) What is the impact of *L. camara* invasion on soil nutrients in KAZA TFCA?

2.2. Methodology

2.2.1. Study site

To avoid repetition, the study site is described in detail in Chapter 1 (refer to section 1.6).

2.2.2. Data collection

A reconnaissance survey was conducted to determine areas with *L. camara* in the Park and the Communal area. Both the Park and the Communal area were then stratified based on invasion, namely "invaded" or "uninvaded". Grid squares of the sampled area were established using Universal Transverse Mercator (UTM) grid squares. The 60 plots measuring 25 m x 25 m each were then selected randomly using computer generated numbers, following Chatanga et al. (2018). A total of 18 plots were selected in the Park, with 12 in invaded areas and six in uninvaded ones. In the Communal area, a total of 42 plots were sampled, comprising 34 invaded sites and eight uninvaded ones. For each plot, GPS coordinates were taken in the middle of the plot.

2.2.3. Vegetation attributes

Data collection was done at the end of the rainy season (April-May 2018) to make identification easier. Native woody species for this study were defined as hard stemmed, self-supporting plants with a diameter at breast height (DBH) of ≥ 5 cm. In each plot, species were identified using appropriate field guides (Van Wyk, 2013), and stem counts per species were done and recorded. Vegetation structure attributes such as canopy cover, DBH, and height were also measured. Mean Canopy Diameter was estimated using the crown diameter method developed by Ellenberg and Mueller-Dombois (1974) as seen in Chatanga et al. (2018). The longest diameter of the canopy and the diameter perpendicular to it were measured to get the mean diameter. The mean crown diameter was then used to estimate the mean canopy diameter using the formula below and the figures were converted to a percentage:

$$\textbf{Mean canopy diameter} = \pi\,(D/2)^2$$

Where D is the mean crown diameter

The height of all trees and shrubs in each plot was measured using a ranging rod. For plants with multiple stems, the height of the tallest stem were measured (Chatanga et al., 2018; Ngarakana and Kativu, 2018). The DBH for all woody species in the plot was measured using a diameter tape. For multi-stemmed individuals, a mean DBH was derived through the use of a quadratic formula (Ganamé et al. 2019) shown below:

$$\textbf{DBH}= \sqrt{\sum \boldsymbol{di^2}}$$

Where d_i is the diameter of the different stems and measurements were in metres (m).

2.2.4. Soil attributes

Soil samples were collected at a depth range of 10-15 cm. This range ensures consistency in results on testing and is similar to what many scholars have applied such as Chatanga et al. (2018); Panter and Ruwanza (2019). A sample was collected at each of the four corners of the plot and one sample was collected in the middle. The five samples per plot were mixed thoroughly before analysis (Ngarakana and Kativu, 2018). The soil samples were sieved using a 2 mm mesh sieve to discard pebbles and roots larger than 2 mm. Although 60 composite soil samples were collected, due to the high cost of analysis, only 36 were sent for analysis (20 Communal area and 16 Park).

The soil nutrients were analysed for N, phosphorus (P), K, pH, and organic carbon (OC). Nitrate nitrogen (NO_3^--N) was analysed calorimetrically using a Shimadzu UV 1800, after extraction from the soil with a 1:10 soil:1M KCl solution. Available P was determined with a calorimetric test using a Shimadzu UV 1800 after extraction from the soil by the Mehlic 3 procedure (Gentili et al., 2018). To determine exchangeable K, flame emission spectrometry was conducted using a Varian AA 200 after extraction from the soil using the Mehlic procedure. The OC was determined using the Walkley-Black procedure (Sato et al., 2014). The analyses were carried out by the Department of Applied Chemistry Laboratory at the National University of Science and Technology (NUST) in Zimbabwe.

2.2.5. Data analysis

Species diversity was calculated using Shannon-Wiener (H) index in PAST software (2018). The formula for the Shannon-Wiener index is shown below:

$$\boldsymbol{H = -\sum p_i \ln p_i}$$

Where p_i represents the proportion of each species based on counts in the sample.

Density was calculated, per species and plot, as stems per hectare. The Frequency (F) was taken as the proportion of sample plots in which a given species was noted.

The normality and homogeneity of variance of the variables were determined using Shapiro-Wilks and Levene's tests, respectively. Two-way permutational multivariate analysis of variance (PERMANOVA) was used to examine the effects of site and invasion conditions on woody species diversity and vegetation attributes. The two-way analysis of variance was used to test the differences between sites (Communal area and Park) and invasion conditions for factors that were non-normally distributed after relevant transformation, including soil variables. The non-parametric Mann-Whitney U test was used to test whether there were differences between Communal and invaded Park sites in terms of *L. camara* stem height, DBH, canopy cover, Shannon-Wiener index, and richness. PERMANOVA was conducted using PRIMER statistical software, version 6, and its add-on package PERMANOVA+, whereas an Analysis of Variance (ANOVA), a Mann-Whitney U test, and Shapiro-Wilks and Levene's tests were performed in STATISTICA version 13.

A two-way analysis of similarities (ANOSIM) (Clarke and Warwick, 1994), a non-parametric permutation procedure similar to ANOVA, was conducted to examine the effects of sites and invasion status on woody species composition. The non-metric multidimensional scaling (NMDS) ordination (Clarke and Warwick, 1994) based on Bray–Curtis distances was used to evaluate the distribution patterns of woody species between Communal and Park sites. Further, a cluster analysis was used to illustrate the similarities and differences between the Communal area and Park. ANOSIM and cluster ordinations were performed in PRIMER version 6, whereas NMDS was performed using the Vegan: Community Ecology Package in R (Oksanen et al., 2020).

A Detrended Correspondence Analysis (DCA) on the woody species data was conducted to select an appropriate model (Chen et al., 2016; Lepš and Šmilauer, 2003). Canonical Correspondence Analysis (CCA) was used to evaluate the relationship between woody species composition and soil variables. The gradient length of the first ordination axis in this was greater than 3.5; hence CCA ordination, which is based on the unimodal method, was used. Furthermore, forward selection with Monte-Carlo permutation tests (999 unrestricted permutations, $p < 0.05$) was conducted to assess whether the CCA model was significant and to define important soil variables that explain the distribution and abundance of woody communities. A Pearson correlation analysis was used to test the correlations between soil variables and species diversities. The CCA and DCA were undertaken using the Vegan: Community Ecology Package in R (Oksanen et al., 2020), whereas the Pearson analysis was performed in STATISTICA software, version 13.

2.3. Results

2.3.1. Vegetation structure and diversity

The invaded plots were markedly different from the uninvaded plots in terms of all the variables investigated; this pattern was evident across the Park and Communal sites (Table 2.1). Plots invaded with *L. camara* had lower woody plant stem height, stem diameter, density, canopy cover, diversity, and species richness than uninvaded plots (Table 2.1). Whilst all these measures were less in the invaded sites, the most marked differences were the native species canopy cover that decreased from above 70% in the uninvaded sites to less than 10% in the invaded sites. Stem density also showed a marked difference due to invasion ($p < 0.002$), with uninvaded plots in both the Park and Communal area having a higher stem density compared to the invaded plots (Table 2.1).

Species richness also differed significantly among all the sites, with the highest richness in the uninvaded plots in the Park, followed by the uninvaded plots in the Communal area, and then invaded sites (Table 2.1). Consequently, the Shannon-Wiener index showed diversity to be higher in the uninvaded plots than in invaded ones. There were, however, no significant differences when comparing species diversity in both sites in the same invasion category (Table 2.1). Comparing *L. camara* in the Communal area and Park showed a seven times higher density ($Z = 4.5$; $p = 0.001$) and more than four times greater canopy cover ($Z = 4.2$; $p = 0.001$) of *L. camara* in the Communal area than in the Park. The mean height of *L. camara* was also approximately double in the Communal site than what it was in the Park ($Z= 3.4$; $p = 0.001$; Table 2.1.).

Table 2.1: Mean (±SD) stem height, diameter, density and canopy cover for woody species and *L. camara* in invaded and uninvaded the Communal and Park areas. (Means with different letters (superscripts) indicate significant differences between invasions).

Variables	**Communal area**		**Park area**		**Test Statistic**	
	Invaded	**Uninvaded**	**Invaded**	**Uninvaded**	**value**	**p-value**
Native Woody Species						
Stem height (m)	5.1±1.6^{c}	9.5±0.6^{ab}	8.5±2.0^{b}	10.7±0.8^{a}	31.52	0.001
Stem diameter (m)	0.4±0.1^{b}	0.6±0.02^{a}	0.7±0.4^{a}	0.8±0.1^{a}	24.68	0.001
Density/ha	228±308^{b}	1 308±432^{a}	479±284^{a}	1 561±706^{a}	18.59	0.002
Canopy cover (%)	6.6±5.9^{b}	81±14.7^{a}	9±10.5^{b}	72.5±9.4^{a}	7.26	0.01
Shannon-Wiener index	0.4±0.6^{c}	2.8±0.4^{a}	1.6±0.5^{b}	3.2±0.4^{a}	74.97	0.001
Species richness	5.4±3.3^{d}	21.3±5.1^{b}	12.7±4.1^{c}	33.5±0.3^{a}	51.02	0.001
L. camara					**Z-value**	**p-value**
Stem height (m)	2.2±0.8^{a}	N/A	1.8±0.1^{b}	N/A	3.4	0.001
Stem diameter (m)	0.04±0.01^{b}	N/A	0.07±0.10^{a}	N/A	1.1	0.04
Density/ha	5 542±4 747^{a}	N/A	747±567^{b}	N/A	4.5	0.001
Canopy (%)	47.4±27.8^{a}	N/A	10.3 ±2.0^{b}	N/A	4.2	0.001

2.3.2. Species composition and its association with the invasion of *L. camara*

In this study, fifty native woody species from 21 families were recorded (Table 2.2). Uninvaded plots in the Park had the most species (48), whereas the invaded Communal plots had the lowest number (25). Generally, woody species were more abundant in the uninvaded sites; this was particularly so in Communal areas (ANOSIM R = 0.8; p = 0.001) (Table 2.3). The same distribution pattern was evident in the Park, except for species such as *Diospyros mespiliformis* and *Hyphaene coriacea* that had equal or greater frequency in invaded plots than in uninvaded plots. Apart from the plots in the Park having more frequently-occurring species, the ANOSIM revealed that the differentials between invaded and uninvaded conditions were greater in the Communal plots (Table 2.3).

The most frequently occurring species, with an occupancy frequency greater than 70% across uninvaded sites were *Afzelia quanzensis, Albizia harveyi, Bauhinia petersiana, Colophospermum mopane, Combretum imberbe, Dichrostachys cinerea, Diplorhynchus* *condylocarpon, Euclea divinorum, Grewia flavescens* and *Terminalia sericea*. Only, *Combretum imberbe* had high frequencies (greater than 50%) in the invaded sites (Table 2.2).

Table 2.2: Fifty species that frequently occur in sites invaded and uninvaded by *L. camara*. Values show frequencies of occupancy for each species (%)

Species Name	**Communal area**		**Park area**	
	Invaded	**Uninvaded**	**Invaded**	**Uninvaded**
Adansonia digitata	0	0	9	100
Afzelia quanzensis	3	75	17	83
Albizia harveyi	12	100	25	100
Antidesma venosum	0	0	9	50
Azanza garkeana	0	0	0	50
Baikiaea plurijuga	3	63	33	33
Bauhinia petersiana	30	100	68	83
Berchemia discolor	0	0	33	83
Brachystegia spiciformis	12	75	9	50
Bridelia mollis	0	0	25	67
Burkea africana	6	75	0	0
Colophospermum mopane	15	75	50	83
Combretum apiculatum	0	63	9	17
Combretum hereroense	0	0	17	83
Combretum imberbe	53	88	92	100
Combretum molle	44	75	17	33
Corymbia gummifera	0	0	0	67
Dichrostachys cinerea	9	75	50	100
Diospyros lycioides	0	0	17	83
Diospyros mespiliformis	0	0	50	50
Diospyros quiloensis	0	0	42	33
Diplorhynchus condylocarpon	21	100	17	100
Euclea divinorum	38	88	33	100
Ficus natalensis	6	88	17	33
Ficus petersii	0	0	9	50
Grewia flavescens	24	100	75	100
Gymosporia senegalensis	9	75	25	50
Hyphaene coriacea	0	0	9	8
Kigelia africana	0	0	9	33
Lantana camara	100	0	100	0
Manilkara mochisia	0	0	9	50
Ozoroa reticulata	0	0	33	67
Peltophorum africanum	0	0	25	83
Philenoptera violacea	0	0	17	50
Phoenix reclinata	0	0	9	83

Species Name	Communal area		Park area	
	Invaded	Uninvaded	Invaded	Uninvaded
Piliostigma thonningii	0	0	9	100
Pseudolachnostylis maprouneifolia	0	0	17	50
Pterocarpus rotundifolius	3	100	25	33
Sclerocarya birrea subsp. *caffra*	18	75	25	33
Senegalia galpinii	32	63	33	67
Senegalia nigrescens	12	88	9	17
Strychnos potatorum	0	0	0	50
Terminalia sericea	21	88	67	100
Terminalia stuhlmanii	0	0	42	67
Trichilia emetica	0	0	17	100
Vachellia erioloba	15	50	25	83
Vachellia karroo	12	63	33	67
Vangueria infausta	24	50	8	100
Ximenia caffra	0	0	17	33
Ziziphus mucronata	15	88	25	83

Table 2.3: ANOSIM results showing the significant difference between sample pairs.

Groups	Global R: 0.62; p = 0.001	
	R-Statistic	**Significance Level (p-value)**
Invaded Communal, Uninvaded Communal	0.8	0.001
Invaded Communal, Invaded Park	0.4	0.002
Invaded Communal, Uninvaded Park	0.9	0.001
Invaded Communal, Invaded Park	0.7	0.001
Uninvaded Communal, Uninvaded Park	0.5	0.001
Invaded Park, Uninvaded Park	0.6	0.002

2.3.3. Species composition

The ordination plot revealed three distinct clusters based on invasion status and land-use (Figure 2.1). Invaded Communal formed Cluster A, whilst invaded Park formed Cluster B; Cluster C consisted of uninvaded sites in both locations (Figure 2.1). Uninvaded plots in Cluster A favoured the assemblage of species such as *C. imberbe, C. molle, G. flavescens, G. senegalensis, P. africanum, V. infausta* and *S. birrea* among others, whereas Cluster B was positively associated with *B. petersiana, C. hereroense, C. mopane, D. cinerea, F. petersii* and

T. emetica among others. The last Cluster was associated with *B. discolor*, *B. spiciformis*, *K. africana*, *S. potatorium* and *T. stuhlmanii*.

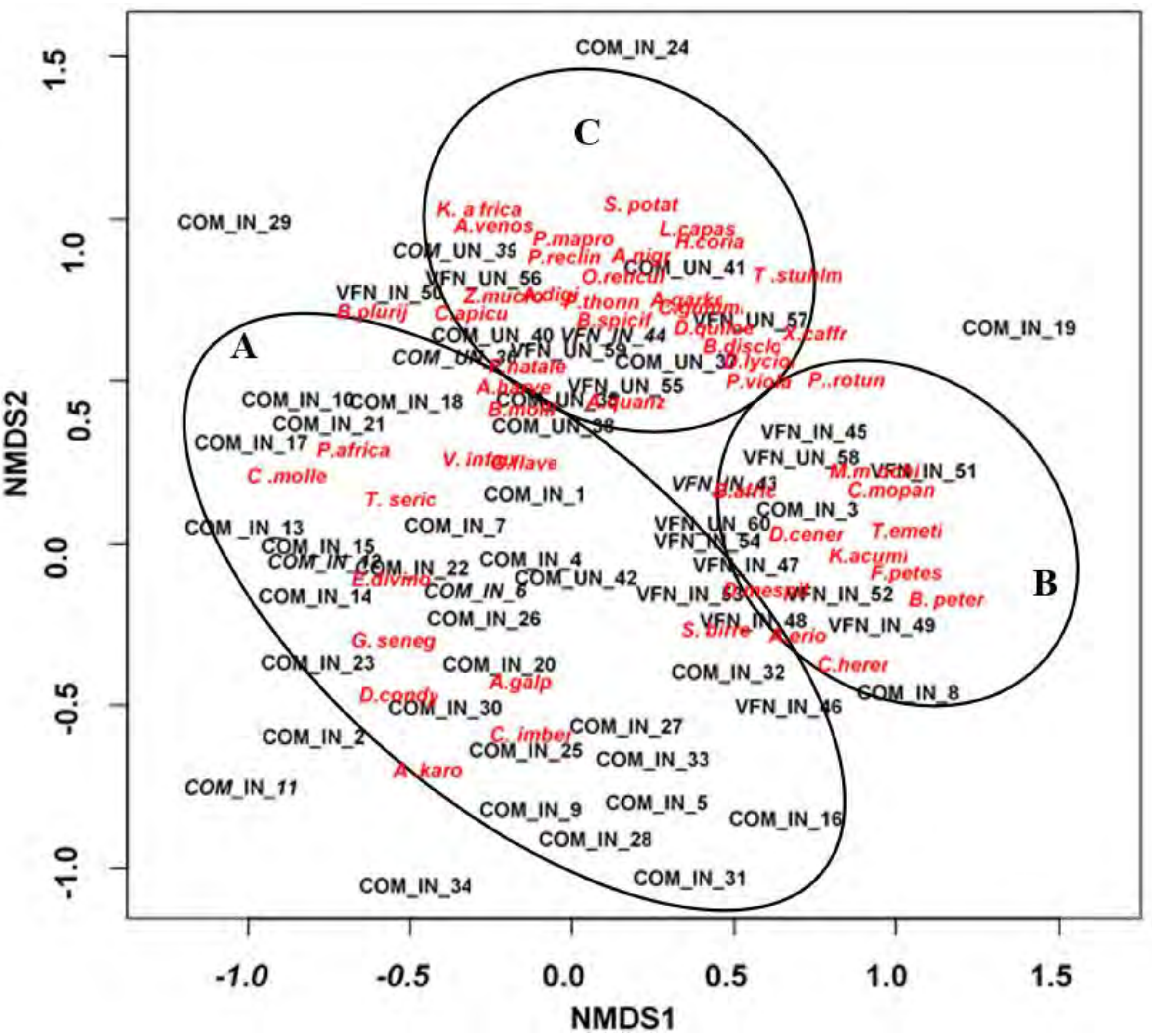

Figure 2.1: NMDS ordination plot showing the relationship between the invasion conditions and native woody species.
Abbreviations: Communal invaded (COM_IN), Communal uninvaded (COM_UN), Victoria Falls National Park invaded (VFN_IN) Victoria Falls Park uninvaded, (VFN_UN). Full species names are in Table 2.2.

Assemblages in the cluster analysis revealed four distinct clusters (Figure 2.2). Like the NMDS ordination, the first cluster, Cluster A, consisted of invaded Communal sites, whereas Cluster B comprised mainly invaded Park sites. Uninvaded Park and Communal areas formed Cluster C, whereas Cluster D consisted of a combination of invaded Park and Communal sites (Figure 2.2).

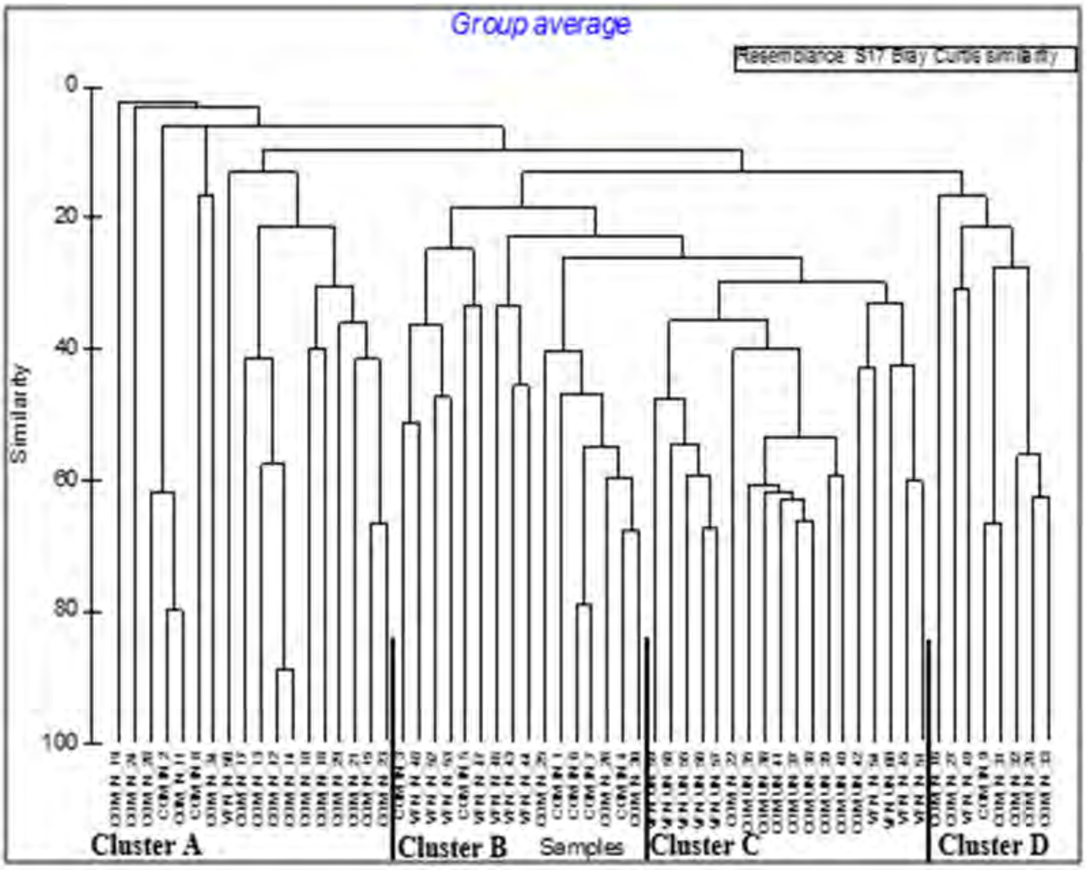

Figure 2.2: Cluster Analysis of woody species from samples in Communal invaded, Communal uninvaded, Victoria Falls National Park invaded and Victoria Falls National Park uninvaded.

2.3.4. Effects of *L. camara* on soil nutrients, species composition and diversity

There were significant differences in soil nutrients between invaded and uninvaded areas (Table 2.4). The most significantly affected variables were pH, P and OC, which were higher in invaded than uninvaded sites. For example, OC was approximately two times higher in the invaded sites than the uninvaded sites in the Communal area, whereas P was 15 to 100 times less in the uninvaded sites compared to invaded sites. Generally, differences between invaded and uninvaded sites were more pronounced in the Communal area (Table 2.4).

The CCA results revealed that the first two axes explained 54% of the total variation between species composition and soil nutrient variables (Table 2.5). Axes 1 and 2 with eigenvalues 0.11 and 0.098 accounted for 30% and 24% of the variance, respectively.

Table 2.4: Comparison of soil pH, N, P, K and OC in invaded and uninvaded Communal and Park areas. Data are mean ± SD and ANOVA results are shown.

Variable	Communal area		Park area		F-value	p-value
	Invaded	**Uninvaded**	**Invaded**	**Uninvaded**		
pH	6.8±0.5^{bc}	7.6±0.6^{a}	6.1±0.3^{c}	7.0±0.8^{ac}	6.3	0.001
NO_3^--N (mg/kg)	2.3±1.5^{a}	1.1± 0.7^{a}	6.7±6.2^{a}	7.5±13^{a}	1.4	0.3
P (mg/kg)	3.8±5.6^{ab}	0.2±0.4^{b}	10.7±8.8^{a}	10.5±0.001^{a}	3.5	0.03
K (mg/kg)	4 040±3 453^{a}	2 050±1 629^{a}	1 660±523^{a}	1 087±554^{a}	1.8	0.2
OC (%)	1.8± 1.1^{ab}	0.95±0.93^{b}	1.9± 0.84^{ab}	2.9± 0.6^{a}	2.6	0.04

The Monte-Carlo permutation test revealed that the CCA model was statistically significant ($p<0.05$), indicating that the soil variables had a significant effect on the community composition of woody species in both locations. The CCA ordination plot revealed that pH and OC were the most important soil variables influencing the distribution of native woody species at the two sites (Table 2.5). The above-mentioned two variables indicated a strong negative association with the invaded Communal sites (Figure 2.3). The variables were positively associated with *D. condylocarpon, G. flavescens, P. maprouneifolia, V. infausta* and *Z. mucronata*. Conversely, pH and OC indicated positive associations with invaded and uninvaded Park sites, and favoured the assemblage of species such as *C. mopane, H. coriacea, O. reticulata, P. thonningii, S. potatorium* and *T. emetica*.

Table 2.5: Summary statistics of CCA ordination on nutrient variables and woody species vegetation in the Communal and Park sites

CCA properties	**CCA1**	**CCA2**
Eigen value	0.21	0.24
Variance explained	0.21	0.24
Cumulative variance	0.21	0.54
Soil variables	**F-value**	**p-value**
pH	1.52	**0.03**
NO_3^-	1.65	0.11
P	1.53	**0.01**
K	1.02	0.37
C	1.74	**0.03**

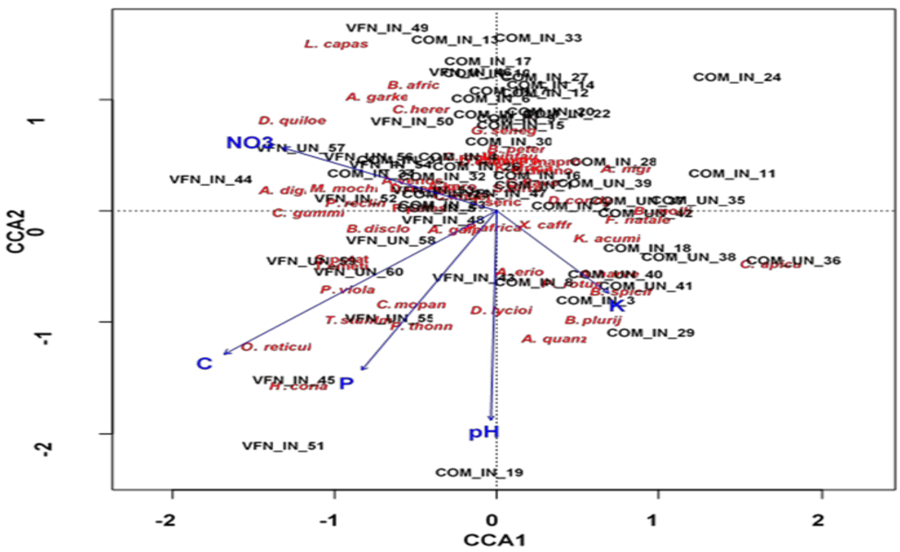

Figure 2.3: CCA ordination plot of the relationships between soil variables and native vegetation

Of the ten measures of vegetation attributes considered, only three were significantly associated with soil nutrients ($P < 0.05$; Table 2.6). Native woody species DBH significantly decreased with increasing pH and OC, whereas *L. camara* height increased with increasing K. Conversely K negatively affected the canopy of *L. camara* (Table 2.6).

Table 2.6: Correlation between species diversity, vegetation and soil nutrient variables. The positive and negative values indicate positive and negative correlations respectively. The asterisk (*) represents level of significance (p<0.05 *) and p<0.001**)

Species diversity/vegetation attributes	**pH**	**NO_3^-**	**P**	**K**	**OC**
Shannon index	0.20	0.15	-0.06	-0.09	0.14
Species richness	0.21	0.28	-0.09	-0.09	0.18
Native woody species density/ha	-0.19	-0.10	-0.10	-0.13	-0.19
Native woody species height (m)	0.13	0.21	-0.04	-0.21	0.12
Native woody species diameter (m)	-0.008**	0.09	-0.07	-0.18	-0.04*
Native woody species cover (%)	0.28	0.10	-0.19	-0.01	0.17
L. camara density/ha	-0.20	-0.12	-0.08	-0.11	-0.19
L. camara height (m)	-0.27	-0.11	0.15	0.043*	-0.17
L. camara canopy (%)	-0.17	-0.06	-0.02	-0.03*	-0.12
L. camara diameter (m)	-0.03	0.02	-0.06	0.04	0.24

2.4. Discussion

2.4.1. Vegetation structure and diversity

This study showed that *L. camara* invasion negatively affected native woody species diversity and composition and is associated with changes in soil nutrients. The ability of *L. camara* to alter ecosystem structure, function and resource availability is well documented. For example, studies by Chatanga et al. (2018), Gooden et al. (2009), Goyal et al. (2018), Negi et al. (2019) and Ruwanza (2020) have all shown the threat posed by *L. camara* to native plant communities, through alteration of their structure and biodiversity. In this study, *L. camara* contributed to a decline in species richness and diversity of native woody species in both the Communal and Park areas. This is in line with Ruwanza (2020) who showed how areas invaded with *L. camara* had reduced species plant diversity and composition. Other vegetation structural attributes such as stem height, DBH, density and canopy cover were also negatively impacted by the presence of *L. camara*. Chatanga et al. (2018) support this by suggesting that *L. camara* invasion contributes to the reduction of native species by reducing native plant recruitment and growth potential. This explains the observed low woody plant density, stem height, and diameter of native plants in invaded areas on both sites.

L. camara's allelopathic effects can also explain the low densities of native plants in invaded sites. Shackleton et al. (2017) and Negi et al. (2019) reported that *L. camara'*s allelopathic chemicals affect the recruitment and growth of native species. Kumar et al. (2020) also assert that the ability of *L. camara* to release phytotoxic chemicals from roots suppresses the growth and recruitment of native plants in surrounding areas. The marked difference in richness and diversity of species among *L. camara* invaded sites was consistent in both sites in this study. Shackleton et al. (2017), in their study on *L. camara* in the Eastern Cape, state that an increase in density of *L. camara* has a negative correlation with native species richness. Jevon and Shackleton (2015) attributed the differences in density and richness among invaded and uninvaded areas to *L. camara* possibly slowing down successional processes through suppression of native vegetation recruitment, density and richness. However, Chatanga et al. (2018) and Kumar et al. (2020) reported that *L. camara* increases fire frequency and intensity, resulting in some native plants being lost during fires, thus reducing species richness and abundance.

Gooden et al. (2009) express the need to exercise caution in attributing species loss to *L. camara* as it is also possible for changes in vegetation structure to happen simply by chance, and, in some instances, native species loss precedes *L. camara* invasion. Factors such as fire and grazing may also have facilitated the decline of native vegetation and consequently improved *L. camara* persistence. This is in line with results from this study, which show high *L. camara* densities in Communal areas as compared to the Park. The need for land by an increasing human population has resulted in extensive clearing of land in the Communal areas for agriculture and settlement, creating patches of disturbance that offer pathways for establishment of *L. camara*. Negi et al. (2019) assert that disturbances caused by anthropogenic factors may modify microclimates, thus leading to an invasion by species such as *L. camara*.

However, Goyal et al. (2018) assert that the longer a species spends in an area, the higher the probability of it performing better. It is difficult to ascertain whether *L. camara* has been resident in the Communal area longer than in the protected area as there are no invasive alien species monitoring programs in the Communal area. However, based on personal communication with the residents in the area, an assumption was made that *L. camara*'s residence in the Communal area has been longer than in Park. As a result, the negative effects of *L. camara* are more pronounced in Communal areas than the Park. *Lantana camara* is poor at invading intact native vegetation (Gooden et al., 2009). The intact nature of the Park might explain the low density of *L. camara* and the relatively low effects on vegetation composition and structure. Furthermore, the invasion could be a new occurrence as the site showed disturbance of native vegetation by elephants. As the elephants move, they destroy some of the vegetation, making the Park less intact and more vulnerable to IAPs that thrive in disturbed environments.

2.4.2. Species composition

Species composition varied greatly between invaded and uninvaded areas, with higher species richness and diversity in uninvaded areas compared to invaded ones (Table 2.2). The uninvaded areas in the Park had the highest species richness (48), while invaded Communal areas had the lowest (25). This is in line with Gooden et al. (2009) who found that *L. camara* significantly alters native species composition. Gooden et al. (2009) also reported that trees that contribute to compositional differences have low abundance in heavily invaded *L. camara* sites, in contrast to uninvaded sites which have high species richness.

The ANOSIM results revealed that species composition was highly different in invaded sites across land uses. Invaded Communal areas had a lower species richness (25) as compared to Park invaded areas (40). The difference in species richness can be explained by the relatively high native species canopy cover in the invaded sites of the Park, which probably limits *L. camara* invasion. This is supported by Gooden et al. (2009), who emphasised *L. camara'*s poor invasion capabilities in intact vegetation. The study further reported that *L. camara* impacts native vegetation over large areas by hindering native species regeneration. This partly explains the difference between invaded Communal and Park sites. However, modification of soil physio-chemical properties may also be a reason for the difference between the Park and Communal invaded sites. Mahla and Mlambo (2019) provided evidence that invasive alien species modify soils to increase their ability for better growth relative to that of native species.

2.4.3. Effects of *L. camara* on soil nutrients, species composition and diversity

In line with Kumar et al. (2016) the results (Table 2.4) showed that *L. camara* invasion greatly influenced soil nutrient concentrations on both sites. Levels of OC varied with specific site invasions, with Communal areas having high OC in invaded sites as compared to uninvaded; this is in contrast with results from the Park which show higher OC levels in uninvaded sites. The result that OC concentrations are elevated in invaded areas concurs with previous studies that concluded that *L. camara* invasion increases OC levels due to accumulation of high quality and quantity litterfall under *L. camara* thickets (Kumar et al., 2020; Mahla and Mlambo, 2019; Ruwanza and Shackleton, 2016).

Ruwanza and Shackleton (2016) suggest that litter accumulation and its subsequent decomposition improves total soil P and extractable K. This is in line with the results which showed that P and K concentrations were higher in invaded areas as compared to the uninvaded areas. Studies have recorded that *L. camara*'s deep root system enables it to take up nutrients in deep soil layers in a process called "nutrient uplift" (Ruwanza and Shackleton, 2016). It is this process that is probably responsible for the increased soil P on the soil surface in invaded sites.

In contrast to other soil nutrients, NO_3^--N were low in invaded areas compared to uninvaded areas. Muvengwi and Ndagurwa (2015) suggested that IAPs are associated with low moisture levels which can affect nitrification, and this could explain the low NO_3^- concentrations in the soil of invaded areas as compared to non-invaded areas. Muvengwi and Ndagurwa (2015) and

Ruwanza and Shackleton (2016) have reported the importance of soil pH in controlling soil nutrients. In this study, soil pH was significantly different in the Communal invaded sites and Park invaded sites as compared to Communal uninvaded and Park uninvaded sites. The decrease in pH may be attributed to *L. camara* invasion which results in high litterfall. As the soil becomes less alkaline, this compromises the bacterial and fungal activity in sites invaded by *L. camara*, leading to a low decomposition rate with confounding effects on nutrient cycling. The study findings are similar to Muvengwi and Ndagurwa (2015) who assert that the low NO_3^- could be attributed to slow nitrification caused by the decline in pH.

The CCA ordination (Figure 2.3) showed that pH and OC had a strong negative effect on species composition particularly in Communal invaded areas. This shows the influence of *L. camara* in soil modification and its eventual negative impact on native species composition. The study findings are similar to Kumar et al. (2020) who suggested that invaded sites experience changes in species composition and structure. They further explain that this change in species composition affects nutrient dynamics and helps the growth and development of invasive species. In contrast, pH and OC indicated a positive association with invaded and uninvaded Park sites. This shows that soil nutrient modification by *L. camara* has a limited effect on species composition in the Park. Gooden et al. (2009) attempted to explain such results as *L. camara* impact thresholds; the study suggested that there is a threshold impact cover at which *L. camara* starts showing changes in native vegetation composition and structure.

2.5. Conclusion

This study shows that *L. camara* invasion in both locations contributes to changes in native woody species composition in the KAZA TFCA. Since the study focused on both Communal and Park areas, the results give insight into *L. camara* negatively affecting woody species composition despite the land use. This study also showed that *L. camara* contributed to the alteration of soil properties. This study suggests that *L. camara* invasion is likely to have serious negative implications for the ecological functioning of the area, and consequently for sustaining the provision of key ecosystem services. This has implications for the management strategies put in place to manage the invasive alien plants.

References

Aravind, N.A., Rao, D., Ganeshaiah, K.N., Shaanker, R.U. and Poulsen, J.G., 2010. Impact of the invasive plant, *Lantana camara*, on bird assemblages at Male Mahadeshwara Reserve Forest, South India. *Tropical Ecology*, 51(2S), 325-338.

Aravindhan, V. and Rajendran, A., 2014. Diversity of invasive plant species in Boluvampatti forest Range, Southern Western Ghats, India. *American-Eurasian Journal of Agricultural and Environmental Sciences*, 14(8), 724-731.

Berry, Z.C., Wevill, K. and Curran, T.J., 2011. The invasive weed *Lantana camara* increases fire risk in dry rainforest by altering fuel beds: *Lantana camara* increases fuel beds. *Weed Research*, 51(5), 525-533.

Bhagwat, S.A., Breman, E., Thekaekara, T., Thornton, T.F. and Willis, K.J., 2012. A battle lost? Report on two centuries of invasion and management of *Lantana camara* L. in Australia, India and South Africa. *PLoS One*, 7(3), 32407.

Chatanga, P., Kamanda, M.T., Kundhlande, A., Imbayarwo-Chikosi, V.E., Mujawo, T., Magadza, C.H.D. and Mujuru, L., 2018. Effects of *Lantana camara* (L.) invasion on the native vegetation of Gonarezhou National Park, Zimbabwe. *Southern Africa Journal of Education, Science and Technology*, 3(1-2), 32.

Chen, Z., Ding, S.X., Zhang, K., Li, Z. and Hu, Z., 2016. Canonical correlation analysis-based fault detection methods with application to alumina evaporation process. *Control Engineering Practice*, 46, 51-58.

Clarke, K.R. and Warwick, R.M., 1994. Similarity-based testing for community pattern: The two-way layout with no replication. *Marine Biology*, 118(1), 167-176.

Dassonville, N., Vanderhoeven, S., Vanparys, V., Hayez, M., Gruber, W. and Meerts, P., 2008. Impacts of alien invasive plants on soil nutrients are correlated with initial site conditions in NW Europe. *Oecologia,* 157(1), 131-140.

Dube, T., Shoko, C., Sibanda, M., Madileng, P., Maluleke, X.G., Mokwatedi, V.R., Tibane, L. and Tshebesebe, T., 2020. Remote Sensing of invasive *Lantana camara* (Verbenaceae) in semiarid savanna rangeland ecosystems of South Africa. *Rangeland Ecology and Management*, 73(3), 411-419.

Fan, L., Chen, Y., Yuan, J. and Yang, Z., 2010. The effect of *Lantana camara* Linn. invasion on soil chemical and microbiological properties and plant biomass accumulation in southern China. *Geoderma*, 154(3-4), 370-378.

Ganamé, M., Bayen, P., Ouédraogo, I., Dimobe, K. and Thiombiano, A., 2019. Woody species composition, diversity and vegetation structure of two protected areas along a climatic gradient in Burkina Faso (West Africa). *Folia Geobotanica*, 54(3), 163-175.

Gentili, R., Ambrosini, R., Montagnani, C., Caronni, S. and Citterio, S., 2018. Effect of soil pH on the growth, reproductive investment and pollen allergenicity of *Ambrosia artemisiifolia* L. *Frontiers in Plant Science,* 9, 1335.

Goncalves, E., Herrera, I., Duarte, M., Bustamante, R.O., Lampo, M., Velasquez, G., Sharma, G.P. and García-Rangel, S., 2014. Global invasion of Lantana camara: Has the climatic niche been conserved across continents*? PLoS One*, 9(10),111468.

Gooden, B., French, K., Turner, P.J. and Downey, P.O., 2009. Impact threshold for an alien plant invader*, Lantana camara* L., on native plant communities. *Biological Conservation*, 142(11), 2631-2641.

Goyal, N., Esler, K.J. and Sharma, G.P., 2018. What drives performance potential of *Lantana camara* L. (sensu lato) in the invaded range? *Tropical Ecology*, 59, 57-68.

Hiremath, A.J. and Sundaram, B., 2005. The fire-lantana cycle hypothesis in Indian forests. *Conservation and Society*, 26-42.

Holmes, N.D., Spatz, D.R., Oppel, S., Tershy, B., Croll, D.A., Keitt, B., Genovesi, P., Burfield, I.J., Will, D.J., Bond, A.L., Wegmann, A., Aguirre-Muñoz, A., Raine, A.F., Knapp, C.R., Hung, C., Wingate, D., Hagen, E., Méndez-Sánchez, F., Rocamora, G., Yuan, H., Fric, J., Millett, J., Russell, J., Liske-Clark, J., Vidal, E., Jourdan, H., Campbell, K., Springer, K., Swinnerton, K., Gibbons-Decherong, L., Langrand, O., Brooke, M., McMinn, M., Bunbury, N., Oliveira, N., Sposimo, P., Geraldes, P., McClelland, P., Hodum, P., Ryan, P.G., Borroto-Páez, R., Pierce, R., Griffiths, R., Fisher, R.N., Wanless, R., Pasachnik, S.A., Cranwell, S., Micol, T. and Butchart, S.H.M, 2019. Globally important islands where eradicating invasive mammals will benefit highly threatened vertebrates. *PLoS One*, 14(3), 0212128.

Jevon, T. and Shackleton, C.M., 2015. Integrating local knowledge and forest surveys to assess *Lantana camara* impacts on indigenous species recruitment in Mazeppa Bay, South Africa. *Human Ecology*, 43(2), 247-254.

Karidozo, M., La Grange, M. and Osborn, F.V., 2016. Assessment of the human wildlife conflict mitigation measures being implemented by the Kavango-Zambezi Transfrontier Conservation Area (KAZA TFCA) partner countries. Report to the KAZA TFCA Secretariat (BMZ No.: 2009, 66, 788 and BMZ No.: 2006, 65, 646), Kasane, Botswana.

Kisaakye, P., 2021. *The effect of* Lantana camara *on soil chemical properties and germination of maize seeds in Mawu Village, Makulubita Subcounty, Luweero District*. Doctoral Thesis: Makerere University, Uganda.

Kumar, A., Jhariya, M.K. and Yadav, D.K., 2016. Vegetation dynamics in plantation sites of collieries. *Nature Environment & Pollution Technology*, 15, 4.

Kumar, M., Verma, A.K. and Garkoti, S.C., 2020. *Lantana camara* and *Ageratina adenophora* invasion alter the understory species composition and diversity of chir pine forest in central Himalaya, India. *Acta Oecologica*, 109, 103642.

Lazzaro, L., Bolpagni, R., Buffa, G., Gentili, R., Lonati, M., Stinca, A., Acosta, A.T.R., Adorni, M., Aleffi, M., Allegrezza, M. and Angiolini, C., 2020. Impact of invasive alien plants on native plant communities and Natura 2000 habitats: State of the art, gap analysis and perspectives in Italy. *Journal of Environmental Management*, 274, 111140.

Lepš, J. and Šmilauer, P., 2003. *Multivariate analysis of ecological data using CANOCO*. Cambridge University Press.

Mahla, N. and Mlambo, D., 2019. Influence of two co-occurring invasive plant species on resident woody species and surface soil properties in Chipinge Safari Area, Zimbabwe. *Tropical Ecology*, 60(1), 129-139.

Muvengwi, J. and Ndagurwa, H.G.T., 2015. Soil seed bank dynamics and fertility on a seasonal wetland invaded by *Lantana camara* in a savanna ecosystem. *South African Journal of Botany*, 100, 190-194.

Nacoulma, B., Salifou, T., Hahn, K. and Thiombiano, A., 2011. Impact of land use types on population structure and extent of bark and foliage harvest of *Afzelia africana* and *Pterocarpus erinaceus* in Eastern Burkina Faso. *International Journal of Biodiversity and Conservation*, 3(3), 62-72.

Nanjappa, H.V., Saravanane, P. and Ramachandrappa, B.K., 2005. Biology and management of *Lantana camara* L. A review. *Agricultural Reviews,* 26(4), 272-280.

Negi, G.C.S., Sharma, S., Vishvakarma, S.C.R., Samant, S.S., Maikhuri, R.K., Prasad, R.C. and Palni, L.M.S., 2019. Ecology and Use of *Lantana camara* in India. *The Botanical Review*, 85(5), 109-130.

Ngarakana, E. and Kativu, S., 2018. Soil based assessment of the invasive species *Vernonanthura phosphorica* (Vell.) H. Rob. (Asteraceae) in Burma Valley, Zimbabwe. *Transactions of the Royal Society of South Africa*, 73(1), 16-19.

Niphadkar, M., Nagendra, H., Tarantino, C., Adamo, M. and Blonda, P., 2017. Comparing pixel and object-based approaches to map an understorey invasive shrub in tropical mixed forests. *Frontiers in Plant Science*, 8, 892.

Oksanen, J., Blanchet, F.G., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., Minchin, P.R., O'Hara, R.B., Simpson, G.L., Solymos, P., Stevens, M.H.H., Szoecs, E. and Wagner, H., 2020. *Vegan: Community Ecology Package.* R package version 2.5-6. 2019.

Osunkoya, O.O. and Perrett, C., 2011. *Lantana camara* L. (Verbenaceae) invasion effects on soil physicochemical properties. *Biology and Fertility of Soils*, 47(3), 349-355.

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

Patel, S., 2011. A weed with multiple utility: *Lantana camara*. *Reviews in Environmental Science and Bio/Technology*, 10(4), 341-351.

Paul, R., Subudhi, D.K., Sahoo, C.K. and Banerjee, K., 2021. Invasion of *Lantana camara* L. and its response to climate change in the mountains of Eastern Ghats. *Biologia*, 76(5), 1391-1408.

Ruwanza, S., 2016. Soil seed bank depletion as a mechanism of *Lantana camara* L. invasion. *South African Journal of Plant and Soil*, 33(4), 303-308.

Ruwanza, S., 2020. Effects of *Lantana camara* invasion on vegetation diversity and composition in the Vhembe Biosphere Reserve, Limpopo Province of South Africa. *Scientific African,* 10, 00610.

Ruwanza, S. and Shackleton, C.M, 2016. Effects of the invasive shrub, *Lantana camara*, on soil properties in the Eastern Cape, South Africa. *Weed Biology and Management*, 16(2), 67-79.

Sato, J.H., de Figueiredo, C.C., Marchão, R.L., Madari, B.E., Benedito, L.E.C., Busato, J.G. and de Souza, D.M., 2014. Methods of soil organic carbon determination in Brazilian savannah soils. *Scientia Agricola*, 71, 302-308.

Shackleton, R.T., Witt, A.B., Aool, W. and Pratt, C.F., 2017. Distribution of the invasive alien weed, *Lantana camara*, and its ecological and livelihood impacts in eastern Africa. *African Journal of Range and Forage Science*, 34(1), 1-11.

Simba, Y.R., Kamweya, A.M., Mwangi, P.N. and Ochora, J.M., 2013. Impact of the invasive shrub, *Lantana camara* L. on soil properties in Nairobi National Park, Kenya. *International Journal of Science and Research,* 2(12), 294-300.

Sundaram, B. and Hiremath, A.J., 2012. *Lantana camara* invasion in a heterogeneous landscape: Patterns of spread and correlation with changes in native vegetation. *Biological Invasions*, 14(1), 1127-1141.

Tererai, F., Gaertner, M., Jacobs, S.M. and Richardson, D.M., 2015. *Eucalyptus Camaldulensis* invasion in riparian zones reveals few significant effects on soil physico-chemical properties. *River Research and Applications*, 31(5), 590-601.

Van Wyk, B., 2013. *Field guide to trees of southern Africa*. Penguin Random House, South Africa.

Wekhanya, M.N., Mbugua, P.K. and Mworia, J.K., 2020. The effect of invasive species *Lantana camara* L. on soil nutrients at Ol-Donyo Sabuk National Park, Kenya. *African Journal of Pure and Applied Sciences,* 1(1), 25-32.

Chapter 3: Cost-benefit analysis and local perceptions of the socio-economic effects of *Lantana camara* on rural livelihoods in the Kavango Zambezi Transfrontier Conservation Area

Abstract

Invasive alien plants impact both positively and negatively on people's well-being and livelihoods, yet there is a dearth of studies examining the contexts in which costs associated with their presence might outweigh the benefits. To address this gap, a cost-benefit analysis of the socio-economic effects of *Lantana camara* on rural livelihoods was conducted in the Kavango Zambezi Transfrontier Conservation Area in Hwange District, Zimbabwe. A total of 300 questionnaires were administered to households and two focus group discussions were held with elders in four wards invaded by *L. camara*. Also, the vulnerability status of households to *L. camara* invasion was assessed using a vulnerability score. The elderly participants reported that *L. camara* invaded the first ward in 1942, and spread to the other three wards within 16 years. It encroached into grazing areas and arable lands, reducing the quality and quantity both of livestock forage and of crop productivity, which affected local livelihoods. Simultaneously, it provided ecosystem services, such as material for live fences around homesteads and fields, mosquito repellent, fruits, fire kindling, ornaments, and protection of fowls from raptors. Costs borne by locals relate to the eradication of *L. camara*, displacement of valued indigenous plants, shortage of forage for livestock, and the harbouring of rodents and snakes by *L. camara*. Combined, the mean costs were US\$ 483 household^{-1} year^{-1}, while the mean benefits derived from *L. camara* were valued at US\$ 716 household^{-1} year^{-1}, thereby exceeding the costs by 48%. Although the mean benefit value was higher, most households (62%) experienced net negatives, indicating that mean benefits outweigh costs because a minority of households experience unusually high net benefits. This shows a complex situation because some households gain and some lose. Most households (58%) were at moderate risk from the invasion of *L. camara,* while a few (2%) indicated extreme risk. Management by use is recommended considering the complex of gain and loss experienced by households, which makes it difficult to suggest a method that may benefit one group disadvantaging the other group.

Keywords: Invasive alien plants; Livelihood assets; Perceptions; Vulnerability

3.1. Introduction

When invasive alien plants (IAPs) occupy new areas, they often impact the native vegetation, soil and ecological processes (Weidlich et al., 2020). Depending on the effects resulting from the IAPs, this may impact positively or negatively on any rural communities that depend on the environment for survival. Some effects caused by IAPs include modification of the environment to suit the IAPs and displacement of native plant species, which may eventually impact negatively on local livelihoods (Corlett, 2020). Despite IAPs being generally associated with negative effects on the environment and people (Reynolds et al., 2020), some also provide ecosystem services such as timber, firewood, food and environmental services such as erosion control, soil nutrient enrichment and carbon sequestration (Shackleton et al., 2019).

Lantana camara is an IAP that has spread in many tropical countries and has been associated with causing a reduced abundance and diversity of native forest species (Mungi et al., 2020; Ruwanza, 2020), reduced herbaceous species diversity and plant productivity (Zende et al., 2017), and modification of soil conditions (Ruwanza and Shackleton, 2016). This has ripple effects on rural communities that directly depend on natural resources for their livelihoods. For example, they experience reductions in forage for livestock, crop yields and availability of native plants used for medicine (Shackleton et al., 2017). The disturbance of natural ecosystems through *L. camara* encroachment is widely associated with the disruption of a subsistence way of life. For example, in the Sokoke forest in Kenya, the loss of indigenous species resulted in communities being unable to access medicinal plants and other traditional benefits derived from the natural forest (Maunguja, 2016).

In South Africa, the invasion of *L. camara* resulted in more costs than benefits in the Vembe Biosphere Reserve (Mhlongo, 2021). *Lantana camara* was found to change the environment, impacting native species growth and diversity therefore negatively affecting food resources and other ecosystem services. In India, the invasion of *L. camara* made it difficult to gather firewood and harvest edible tubers due to the thick undergrowth of *L. camara,* and caused the reduction in trees providing non-timber forest products (NTFPs) (Sundaram et al., 2012). Similarly, most respondents in Tanzania highlighted that *L. camara* was harmful to their livelihoods (Mushi, 2019), due to the loss of livestock when they stray into the dense thickets. The thickets formed by *L. camara* may also pose a danger to life, by reducing visibility and harbouring dangerous wild animals such as bears and elephants in the case of India (Sundaram et al., 2012) and snakes in South Africa (Jevon and Shackleton, 2015).

The adverse effects of *L. camara* invasion in communal areas may result in environmental shocks and stresses that in certain instances affect the ability of people to continue accessing resources that are key to their daily activities (Shackleton et al., 2017; Tarugara et al., 2022). It has also been found that some traditional animal migration routes in the Serengeti have been impacted due to the proliferation of *L. camara* (van Wilgen et al., 2017). The invasion of *L. camara* may reduce the ability of people to access land for cultivation or grazing for livestock. Agricultural activities, especially crop production and livestock, are key livelihood activities for many rural people, hence their loss or decrease may result in heightened vulnerability. In east Africa, respondents indicated that *L. camara* reduced the amount of land available for livestock to access forage and it also resulted in the reduction of crop yields (Shackleton et al., 2017). The loss of productive land for crops and grazing may force people to seek alternative means, such as the purchase of supplementary feed during what would have been a normal farming season. In other instances, it may necessitate people resorting to other means, such as moving their livestock to faraway places in search of better pastures (Shrestha, 2021).

IAPs frequently decrease or displace indigenous tree species, some of which are a source of NTFPs. In cases where *L. camara* has invaded the ecosystem, some fruits and plant species may cease to exist, and hence make communities vulnerable to shocks and stresses. In some instances where *L. camara* has invaded grazing and crop fields, communities have been forced to vacate and seek areas where *L. camara* has not invaded (Ranjan, 2019). In some cases, the vulnerability has extended to people being unable to access areas of cultural importance (Jevon and Shackleton, 2015), thereby restricting them from carrying out rituals. The invasion of *L. camara* has made households consider vacating areas where they have lived for generations as vulnerability increases and as costs to eradicate it escalate (Shrestha, 2021). Such decisions mean that households must incur huge costs to establish new residential areas and fields, and ultimately settle in unfamiliar areas that lack social capital support.

While the costs associated with IAPs are well studied and documented, there has been a dearth of information regarding the benefits of IAPs, they are however recognised as a great ecological threat (Rana and Akhter, 2010; Mukul et al., 2006). The limited information reveals that there are benefits from some IAPs in particular contexts. For example, in South Africa, *L. camara* is used as a hedge to substitute fencing and its twigs are used for kindling (Jevon and Shackleton, 2015; Patel, 2011). *Lantana camara* is used as a fuel for portable stoves (Sharma et al., 2020).

Most rural communities rely on fuelwood for cooking and sometimes they travel a distance to find fuelwood.

Therefore, the integration of *L. camara* as a source of energy not only reduces the time spent by local people finding fuelwood but may also mitigate the firewood shortage in areas where *L. camara* is abundant, particularly when indigenous trees for fuelwood are few or scarce (Nawaz et al., 2016).

L. camara is widely used as an insecticide on vegetables, such as rape (*Brassica napus*) (Culver and Precious, 2018; Patel, 2011). Additionally, its leaves are used as mulch in vegetable production (Negi et al., 2019). The leaves of *L. camara* are high in nitrogen and have low lignin and therefore decompose quickly (Negi et al., 2019). Production of vegetables is a common livelihood activity in Zimbabwe, with most households cultivating home gardens (Maroyi, 2009) in their backyards or near watercourses. Therefore, the use of *L. camara* as an insecticide as well as a mulch eliminates the cost of purchasing insecticides and fertilisers, while on the other hand, it promotes healthy home gardens that offer resilience against food insecurity (Routray et al., 2021). Communities can use leaves from any tree species; in this respect, *L. camara* becomes more valuable in the sense that it is termed an "undesirable species" where it is not wanted. Patel (2011) notes its benefit as a fumigant against weevils for grain storage. Safeguarding grain is a critical aspect of livelihood sustenance in rural areas, as it enables communities to have access to food throughout the year.

The ripe berries of *L. camara* are eaten by both people and birds (Negi et al., 2019; Turner and Downey, 2008). Birds play an important part in the ecosystem in which they disperse seeds and act as pollinators (Turner and Downey, 2008). Also, birds provide a source of protein for people, and fruits from *L. camara* have been fermented to provide wine and blended to provide juice (Tsegay and Lemma, 2020). Most rural communities in Africa to some extent rely on natural capital for their livelihoods, and have ways and means of processing and preserving food using local knowledge; this is crucial for food security as it assists households in having food at all times throughout the year (Ibnouf, 2012). Hence, the passing of knowledge of how to preserve food from IAPs like *L. camara* becomes an important aspect of human capital.

In some parts of Ethiopia and South Africa, the leaves of *L. camara* are heavily browsed by livestock, especially goats (Ntalo et al., 2022). Ntalo et al. (2022) note that in Southern Africa, mainly soya and groundnuts are used in the production of livestock feed. Therefore, the use of

L. camara as an alternative reduces the cost of purchasing soya and groundnuts, thus resulting in some cash savings that could be diverted toward other household needs.

Communities have used *L. camara* for medicinal purposes to treat various ailments and also as a mosquito repellent (Alghamdi et al., 2020; Culver and Precious, 2018). Similar to savings realised from using the leaves of *L. camara* as fodder, using *L. camara* for medicinal purposes mitigates against the need to purchase medication or mosquito repellents, which may be expensive and thus diminish the household's financial capital. Some communities have found ways to convert IAPs such as *L. camara* into durable household goods such as furniture, for sale (Priyanka and Joshi, 2013).

At the household scale, the costs and benefits are dependent on several factors such as the type and social status of that household, its needs and its ability to withstand shocks and stresses imposed by IAPs. To adapt to the adverse effects of IAPs, some households derive uses for *L. camara*. For example, the use of the pesticidal properties of plants such as *L. camara* and *Tithonia diversifolia* has been known to encourage weed control in cultivated plots for rural households in Tanzania (Mkindi et al., 2021). This is important because a conventional pharmaceutical pesticide may not be readily available or may be expensive for some households. The benefits of using plants such as *L. camara* in this regard is that they are readily available, and households are easily able to experiment and adapt them to uses that might be beneficial and supplement conventional pharmaceutical products which are costlier. In Southern India, studies have revealed that in certain instances some households realised a 7% to 46% overall income increase due to the adoptive use of *L. camara* in furniture making, and this was when it impacted traditional sources of income as opposed to non-users (Kannan et al., 2014).

It is agreed that such perceived benefits cannot be viewed as static, as the uncontrolled proliferation of IAPs is likely to bring unforeseen challenges and potentially become problematic to households. As IAPs dominate lands, and households lose control over such invasive plants, the perceived benefits tend to diminish (Westwood, 2020). This situation will require management approaches in which households and associated communities participate in cost-benefit analyses of the use of IAPs, for example through adaptation of their pesticide capabilities (Mkindi et al., 2021; Shackleton et al., 2019). The Shackleton et al. (2007) conceptual framework for the interpretation of the impacts of IAS on rural livelihoods, described in Chapter one, shows that neither costs nor benefits are static in time. Each aspect (i.e., benefits, costs, abundance and vulnerability) changes in response to a host of social, economic and ecological dynamics through time. Thus, whilst a cost-benefit analysis is useful, it cannot project too far into the future, especially if the invasions are still strong in terms of increasing the spatial extent or density of the IAS. In accord with this statement, Ntalo et al. (2022) and Vardien et al. (2012) posit that the *L. camara* invasion in Southern Africa will significantly increase in the coming decades, its control will present major challenges to households, and the benefits derived will be outweighed by the negative effects.

Different perceptions highlighted from these many studies show that, although there are general socio-economic effects that may be common, some effects may vary depending on the context. Consideration of the benefits and costs of IAPs for livelihoods and human well-being, is critical for the establishment of policy (Shrestha, 2021). Furthermore, establishing the perceptions of those affected by the IAPs such as *L. camara* is critical for giving informed recommendations for management interventions. The socio-economic effects of *L. camara* invasions on rural livelihoods are poorly understood in many countries, including Zimbabwe. It is against this background that the present study was conducted to establish the perceptions of local people on the socio-economic effects of *L. camara* on their livelihoods. The main research questions addressed by the study were: (i) What is the invasion history of *L. camara* in the Hwange District, (ii) What are the effects of *L. camara* on livelihood assets, (iii) What are the benefits of *L. camara* to livelihood assets, and (iv) What is the vulnerability status of local people in relation to exposure to *L. camara*?

3.2. Methods

3.2.1. Study site

To avoid repetition, the study site is written in detail in Chapter 1 (refer to 1.6).

3.2.2. Data collection

The research approach included the use of a semi-structured questionnaire and focus group discussions (FGDs) (Creswell and Creswell, 2018), conducted between May and September 2018. The research tools received ethics clearance from the Rhodes University Ethics Committee (Ethics number ES17/39). Furthermore, permission was also granted by the Hwange Rural District council and the traditional leadership (chief, headman and village heads). Both the questionnaire and FGDs were carried out in Ndebele, which is the common local language. This was to ensure the respondents understood the questions and could articulate their responses fully.

3.2.2.1. Focus group discussions

Two focus group discussions (FGDs) were conducted. The FGDs involve structured, in-depth conversations with a group of individuals with the help of a facilitator (Gundumogula, 2020). Following Kannan et al. (2013), the participants for the FGDs were purposively selected from all four wards, guided by the traditional leadership (chief, headman and village heads) in the area. The recommended group size for FGDs ranges from four to 12 individuals (Muijeen et al., 2020), and thus one FGD consisted of ten females and the other of eight males to enable the free expression of individuals in each group. Each FGD lasted for 60 to 90 minutes, which was long enough to hold in-depth discussions (Breen, 2006). Each participating member was allowed to voice their opinion in a safe environment to enable freedom of expression. Focus groups were carried out before the household interviews to provide a baseline understanding. The FGDs focused on an array of topics (refer to Appendix 6 for details) including:

- The history and knowledge of the invasion of *L. camara*, establishing when the invasion first occurred, how it was brought to the area, and what prompted the spread from one ward to another.

- Views on the effects of *L. camara* on livelihood assets – establishing how *L. camara* impacted local livelihoods, either positively or negatively.

3.2.2.2. Household survey

Households were randomly selected across the four study area wards selected in the Ndlovu Communal area. Databases of households residing in these wards were obtained from the respective ward councillors. This was used to generate a list in Microsoft Excel and numbers were assigned to each household. Random selection was then done using the random number function (RAND) in Microsoft Excel to select 300 households (75 from each ward). At each selected household, the head was interviewed, or if the head was absent, the eldest individual present was interviewed. Where there was no person present, or no one willing to participate, the nearest neighbouring household was approached. Each interview lasted approximately 45 to 60 minutes. The questionnaire was divided into subtopics including (see Appendix 5 for details):

- The invasion history – this was established based on answers by elderly respondents (n=19) who were already residing in the area between 1940 and 1970; they were asked when they first saw *L. camara* in the area. Additionally, these respondents were asked how *L. camara* was introduced in the area.

- The effects of *L. camara* on livelihood assets – this was established based on all sampled respondents (n=300), where for each of the five livelihood capitals, respondents were asked specific questions about whether the presence of *L. camara* had impacted negatively or positively on their livelihood.

- The vulnerability status of respondents was established based on all sampled respondents (n=300). The questions were asked in such a way as to ascertain if the presence of *L. camara* caused a decline in assets and capabilities which resulted in a reduced capacity to withstand shocks. The main questions included vulnerability to loss of crops, loss of fodder, and snake bites among other questions. Furthermore, households were asked to score vulnerability for each variable, assessed as the degree of risk to rural household assets caused by *L. camara* invasion (vulnerability status). The vulnerability scores were set as follows: 0 = none, where a respondent indicated their household not to be at risk; 1 = medium, where a respondent considered to be at minor risk; 2 = high, where a respondent considered a household to be at high risk; and 3 = very high, where a respondent considered a household to be at very high risk.

3.2.2.3. Cost-benefit analysis

Establishing the net effects of *L. camara* on livelihood assets requires the inclusion of all the costs and benefits of the affected households (Reyns et al., 2018). Each household was asked to establish if there were any costs incurred associated with the invasion of *L. camara*. The questions covered the five livelihood assets (financial, human, natural, physical and social). For example, the days spent on eradication of *L. camara*, the number of tubes of cream used to treat skin irritations resulting from physical contact with *L. camara*, the amount of supplementary livestock feed purchased, the number of packets purchased in a month to treat unwanted pests, time spent on eradication of *L. camara*, and displacement of valuable native tree sources of firewood. Also, they were asked if the invasion prevented them from accessing cultural sites, the number of tons and hectarage of fodder lost due to the invasion, and if there was any relationship between the costs incurred due to the presence of *L. camara* and the size of household land holding.

A similar method was applied to establish the positive effects of *L. camara.* This included asking questions, e.g., if they used *L. camara* and, if so, how many tubes of repellent would they ordinarily buy in a year which were now being offset by using *L. camara*; the number of *L. camara* plants used as ornaments; the length of *L. camara* hedging instead of using wire fencing; the amount of fruit derived from *L. camara*; the amount of fuelwood/kindling harvested; and the number of chickens protected against raptors. Where households indicated that *L. camara* was used as a hedge, the length was measured. Also taken into consideration was the durability of a fence. In general, a good fence is durable for a minimum of 20 years (Worley, 2009). Therefore, costing the hedging benefit would require collecting information on the price of a diamond mesh wire fence, the total length needed and factoring in the durability of the fence to determine value of annual cost savings. This was done for all the positive effects listed. Household data were extrapolated to ward level, based on the total number of households per ward.

3.2.3. Data analysis

Both quantitative and qualitative data were entered into MS Excel spreadsheets prior to analysis in R (version 3.5.3). For qualitative data, text analysis was carried out; this involved identifying emergent keywords from the survey responses (Atyosi et al., 2019; Nyumba et al., 2018). Text analysis enables identification of key messages and ideas in qualitative surveys in order to develop a central theme to support analysis. This was coupled with a descriptive summary of how *L. camara* was brought into the area, and its invasion history.

Descriptive statistics were used to summarise the perceptions of respondents on the effects of *L. camara* on their livelihoods. Furthermore, the costs and benefits for each related capital were calculated for each household based on the formulas in Table 3.1. For example, if a respondent indicated that they had engaged in the clearing of *L. camara*, the respondent indicated the number of days they had done so, and this was used to estimate the labour cost by factoring in the prevailing average payment rate for a general labourer per day and the number of workdays per month. In instances where the respondent indicated incurring a cost of buying ointments or creams due to skin irritation from *L. camara*, respondents indicated the number of tubes used and this was used to calculate the cost, factoring in the number of tubes bought per month and the cost of tubes as shown in Table 3.1. The same trend was followed for other costs such as supplementary livestock feed cost, rodenticide cost, displacement of native vegetation cost and reduced crop cost, as shown in Table 3.1.

Table 3.1: Equations used to calculate costs (US$)) related to labour, skin irritation, supplementary cow feed and rodenticides (Cost reference year: 2022)

Equation	Definition
Labour cost: $Household\ Labour\ Cost = PR * WD * T$	LC = monthly labour costs PR = payment rate day^{-1} (\$5.34) WD = work days month^{-1}(22.5) T = time spent on *L. camara* eradication yr^{-1}
Skin irritation cost: $SIC = (Hydr_{price}\ x\ Hydr_{tubes)} + Cala_{price}\ x\ Cala_{bottles)*T)}$	CH = cost of hydrocortisone cream CC = cost of calamine lotion $Hydr_{price}$ = price of hydrocortisone tube (\$1.60) $Cala_{price}$ = price of calamine bottle (\$1.60) $Hydr_{tubes}$ = hydrocortisone tubes household^{-1} month^{-1} $Cala_{bottles}$ = Calamine bottles household^{-1} month^{-1}
Supplementary cow feed cost: $FC = F_{price}\ x\ F_{bags}$ $F_{bags} = F_{3months}/50kg$ $F_{3months} = F_{day}\ x\ FP\ x\ C$	FC = feed cost yr^{-1} F_{price} = feed price of 50 kg bag (\$15) F_{bags} = number of 50 kg bags household^{-1} yr^{-1} $F_{3months}$ = 3 months of feed intake F_{day} = daily feed intake (10.5 kg) FP = feeding period yr^{-1} (3 months) C = cows household^{-1}
Rodenticide cost: $RC = P_{price}\ x\ P_{qty}$	RC = cost of rodenticide household^{-1} yr^{-1} P_{price} = price of an 85 g rodenticide packet (\$3) P_{qty} = number of 85 g packets household^{-1} yr^{-1}
Displacement of valuable native vegetation cost: $DNV = AP_A + AP_B$	DNV = Total cost of wood provided by native vegetation AP_A = annual cost of hoe or axe handles (\$10) AP_B = annual cost of yokes (\$10)
Reduced crop productivity cost: $CP = Y\ x\ A\ x\ M$	CP = cost of reduced maize yield due to invasion $Y_{hectare}$ = Maize yield per hectare $A_{hh/annum}$ = Area lost per household per year M_{cost} = Price of a tonne of maize (\$382)

The value of benefits derived from *L. camara* for each individual was calculated based on formulas presented in Table 3.2. For example, if a respondent indicated that they used *L. camara* as a repellent, the respondent was asked how many bottles of insect repellent would they ordinarily use if they had no substitute. Based on this, the number of bottles was multiplied by the cost of the insect repellent to get a value of the cost they are saving by using *L. camara* as an alternative. The same method was also followed for ornamental benefits where the respondents indicated the number of ornaments they had per household and the average price of an ornamental plant, thereby indicating the saving made by using *L. camara* as an ornament, as shown in Table 3.2. *Lantana camara* is also used as a hedge in place of fencing, so the cost of fencing material was estimated per household, based on the formula presented, and the estimated benefit value was derived (Table 3.2; $ are US$).

Table 3.2: Equations used to calculate benefits derived from the use of ecosystem services provided by *L. camara*

Equation	Definition
Repellent benefit: $R_{benefit} = R_{qty}\ x\ R_{price}$	$R_{benefit}$ = value of *L. camara* as a repellent household^{-1} yr^{-1} R_{qty} = No. of bottles/tubes used household^{-1} R_{price} = average price of insect repellent ($15)
Ornamental benefit: $O_{benefit} = O_{qty}\ x\ O_{price}$	$O_{benefit}$ = value of *L. camara* as an ornamental household^{-1} yr^{-1} O_{qty} = number of ornamental plants household^{-1} O_{price} = average price of ornamental plant^{-1}
Annual Hedge benefit: $H_{benefit} = ((F_{length}\ x\ F_{price}) + (P_{quantity}\ x\ P_{price}) + LC)/D$	$H_{benefit}$ = value substituted by the use of *L. camara* as a boundary per household^{-1} yr^{-1} F_{length} = length of diamond mesh wire fence at a crop field and homesteads and crop fields F_{price} = price of diamond mesh wire fence per metre ($3) $P_{quantity}$= number of fencing poles needed P_{price} = price of each fencing pole ($9) LC = Labour cost D = durability in years of diamond mesh wire fencing (20)
Fruit benefit: $F_{benefit} = F_{qty}\ x\ F_{price}\ x\ H_{period}$	$F_{benefit}$ = value of *L. camara* fruits picked household^{-1} yr^{-1} F_{qty} = amount of fruits picked household^{-1} yr^{-1} F_{price} = price of *L. camara* fruits per kg ($10)
Kindling benefit: $K_{benefit} = K_{qty}\ x\ K_{price}$	$K_{benefit}$ = value of *L. camara* kindling used as fuel K_{qty} = amount of fire kindlers required as substitutes household^{-1} yr^{-1} K_{price} = price of fire kindling ($5)
Fowl protection benefit: $P_{benefit} = No_{birds}\ x\ B_{price}$	$P_{benefit}$ = use of *L. camara* by chickens as cover against raptors No_{birds} = number of chickens protected household^{-1} B_{price} = price of a bird ($6)

The vulnerability indicators were formulated from the focus group discussions and information derived from text analysis of the qualitative data. The degree of risk to household assets caused by *L. camara* invasion (vulnerability status) was based on a subjective self-rating score of vulnerability for each asset. Vulnerability scores were as follows; 0 = none; 1 = medium; 2 = high; and 3 = very high. The magnitude of vulnerability was established by summing each score per household and was categorised as follows; 0 = no risk; 1-5 = low risk (acceptable); 6-10 = moderate risk (tolerable); 11-15 = high risk (intolerable) and 15 = extreme risk (unacceptable).

Frequency counts, graphs and tables were used to present the effects of *L. camara* on livelihood assets, the cost-benefits of *L. camara* presence, as well as vulnerability to invasion, in the Ndlovu Communal area. A regression analysis was conducted to establish if there was a relationship between financial loss and land holding size. Principal component analysis (PCA) was conducted to examine the association of the variables in relation to vulnerability for both male – and female-headed households.

3.3. Results

3.3.1. Respondent profile

Of the 300 respondents, 219 were male (73%) and 81 were female (27%) (Figure 3.1). Adults were the majority (49%) compared to a few youths (10%) and the elderly (41%) (Figure 3.1). Most of the respondents were literate, having attained either a primary school or secondary school qualification, as compared to only 16% that had no formal education (Figure 3.1).

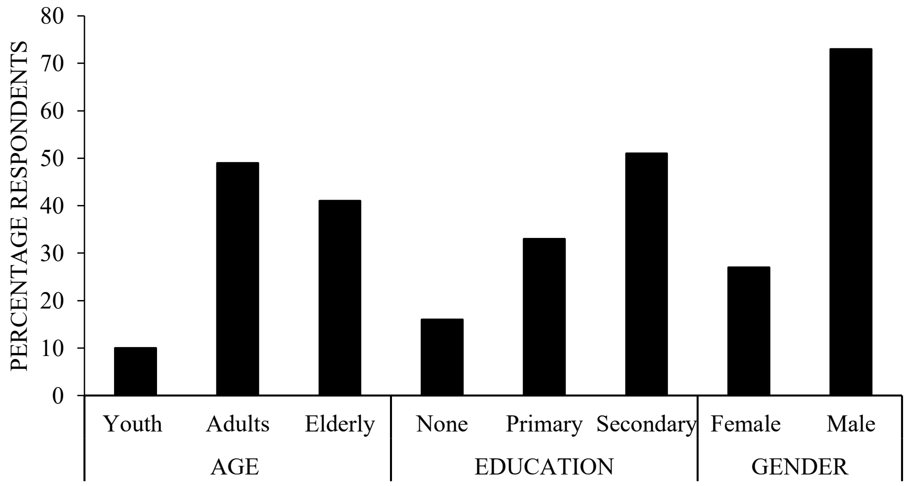

Figure 3.1: Socio-demographic profile of respondents

3.3.2. Introduction of *L. camara* into the area

3.3.2.1. First sightings of L. camara and invasion into different wards

Data from the 19 elderly respondents born between 1925 and 1935, showed that the invasion of *L. camara* in the Ndlovu Communal area had occurred in all four wards by 1968. The first observations of *L. camara* were in the homestead areas and crop fields and eventually it was increasingly observed in the grazing lands and along riparian fringes. Invasion of *L. camara* in the Ndlovu Communal area first occurred in Ward 2, then moved to Wards 3, 4 and 7. The first observation in Ward 2 was in 1942 (Figure 3.2). Later, in the 1950s, it invaded Ward 3 and Ward 4 (1951 and 1954, respectively). The invasion in Ward 7 was 17 years after the first record in Ward 3 (Figure 3.2).

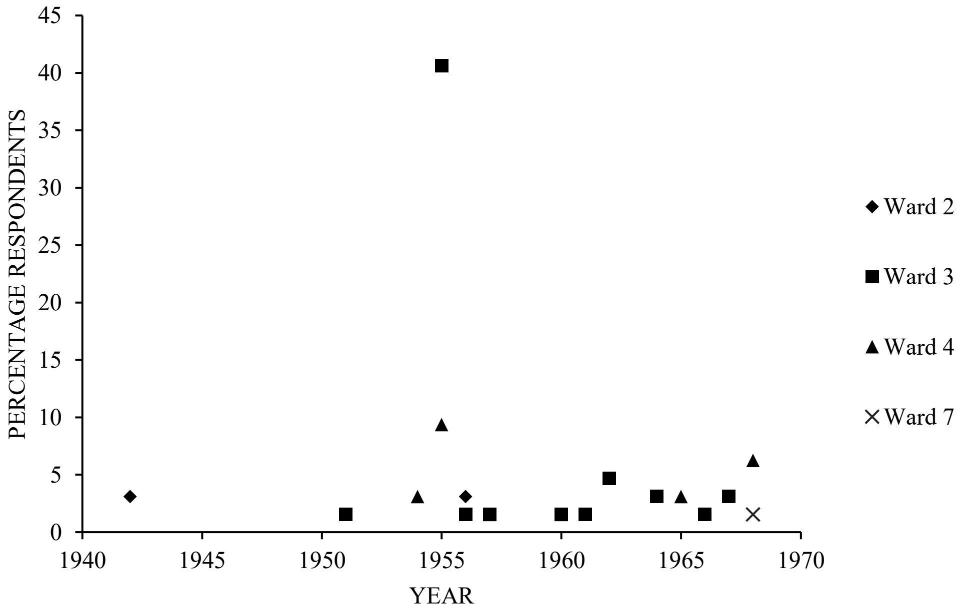

Figure 3.2: Invasion trend of *L. camara* in four wards based on elderly respondents' first sighting (n = 19)

3.3.2.2. Mode of the introduction of L. camara

Approximately half of the respondents (52%) indicated they knew how *L. camara* was established in the area. Among those who said they knew, 36% indicated that the plant was brought in for use as an ornamental plant, 34% said it was brought in for use as a hedge, and 30% indicated that it had self-established. Participants from the FGDs stated that the spread was accelerated by its use as a hedge. Local people would share it for use as a hedge around their homes, vegetable gardens and crop fields. Eventually, *L. camara* was self-establishing in grazing lands, especially along the river.

3.3.3. Effects of *L. camara* on rural livelihoods

3.3.3.1. Negative effects (Costs)

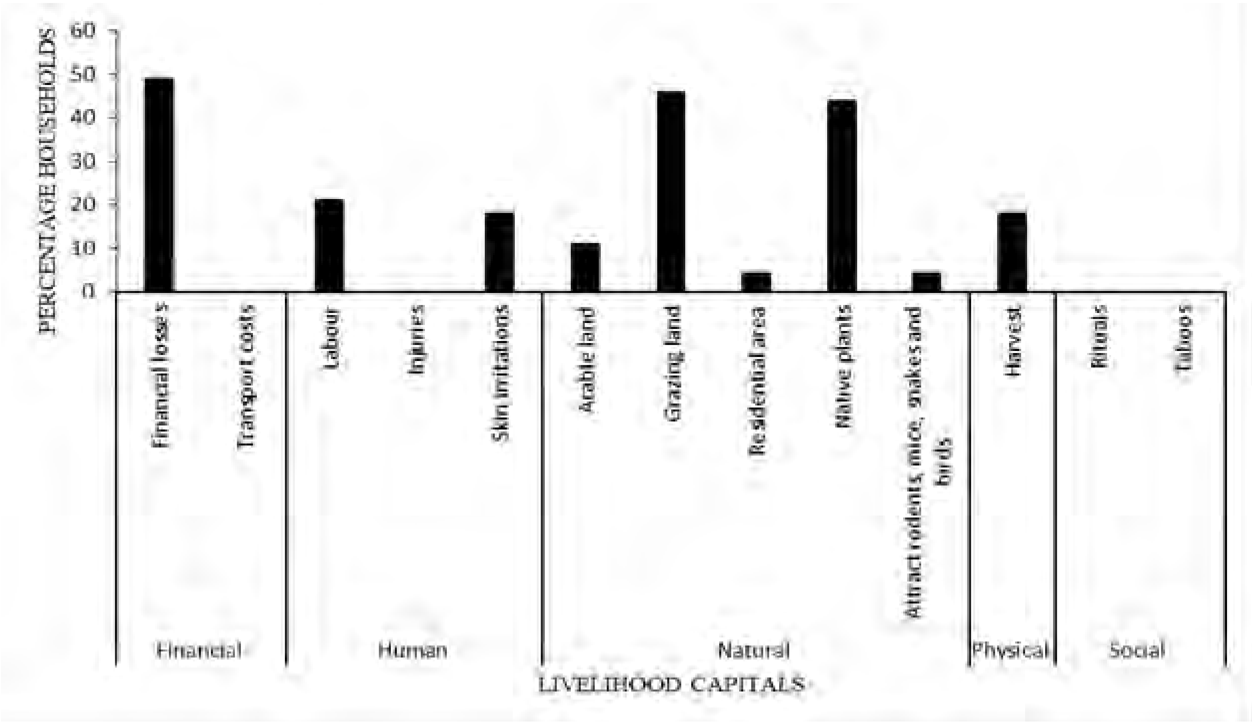

Figure 3. 3: The proportion of households reporting negative effects of *L. camara* on livelihood assets

L. camara impacted some aspects of the financial, natural, social, physical and human capitals (Figure 3.3). Respondents indicated *L. camara* was present in the arable lands, grazing lands and residential areas. Its presence in the grazing lands was associated with negative effects such as a reduction of area occupied by forage for livestock. On the natural capital, *L. camara* was also associated with the reduction in the abundance of some useful tree species such as *Colophospermum mopane*, and preferred forage grasses such as *Panicum maximum* and *Urochloa panicoides.* The invasion of *L. camara* in the residential areas provided a dense cover that reputedly attracts rats, mice and snakes that can be a menace to humans. Concerning crop productivity, only 18% of the respondents indicated that their harvests were negatively affected by the invasion of *L. camara* (Figure 3.3). *Lantana camara* was indicated to have negative effects as it was labour-intensive to remove, as well as causing skin irritations which inhibited locals from carrying on their day-to-day livelihood chores. Although there were no direct negative effects associated with financial capital, 49% of the respondents indicated that the loss in their crops and impact on grazing land translated to financial losses, as agricultural sales were part of their livelihood options (Figure 3.3).

The losses attributed to *L. camara* may be translated to cash values. The mean total loss estimated per household from all assets was US$ 483 per year (Figure 3.4). However, 46% of the respondents experienced annual costs of US$ 0-300. The cost distribution is more or less skewed to the left, showing the percentage of households in a cost category reducing with increased calculated cost (Figure 3.4).

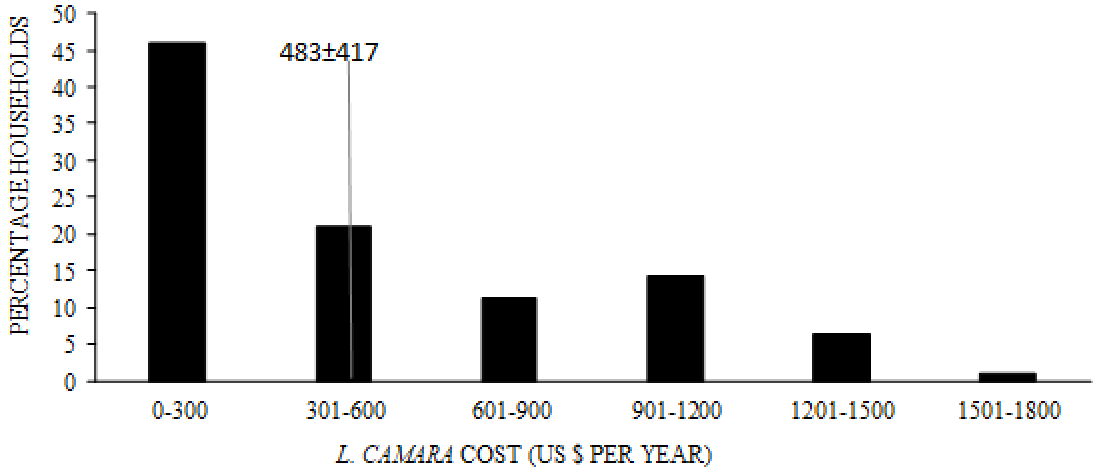

Figure 3.4: Distribution of annual costs experienced by households resulting from *L. camara*

The regression analysis conducted on costs (financial loss) and land holding size gave an R^2 value of 0.602 showing that 60.2 % of the variation of the costs is explained by the land holding size. The p-value for the regression analysis was found to be significant ($p < 2e-16$ ***).

3.3.3.2. Positive effects (Benefits)

With respect to natural capital, 16% of the households rated *L. camara* as beneficial for use as browse for livestock, especially goats and donkeys (Figure 3.5). The nutritional value of *L. camara* has been proven , with crude protein values of between 30 and 70 g/kg which gives a protein level of 28.8% and this compares well with other species such as *Vigna unguiculata* (26%) *Cajanus cajan* (24.6%) and *Phaseolus lunatus* (27.2%) (Ntalo et al., 2022). In Zimbabwe livestock has been observed browsing on *L. camara* leaves during drier periods (Gusha et al., 2016).

Additionally, most respondents (93%) indicated that *L. camara* was a good plant to have in residential areas, especially for use as a live fence or hedge (66%) and as an ornamental in the home (78%). Responses from the FGDs also confirmed these benefits. Respondents indicated that *L. camara* canes are useful for the production of specific products, including portable fowl cages. Some of the respondents (48%) stated that *L. camara* is used in traditional medicine as well as an insect repellent because of the distinct smell of the leaves.

In physical capital, some respondents mentioned that accessing equipment and tools from the Environmental Management Agency to clear the land improved the way they worked and also improved the infrastructure of the area where the roads became accessible and clear. Also, though lumped under products and services, respondents (12%) indicated that there are products such as hoes, axe handles and fowl cages which were made out of *L. camara* (Figure 3.5). Financial savings were made through the use of *L. camara*, e.g., as a useful alternative for fencing. The benefits associated with social capital as shown in Figure 3.5 included bridging and bonding of locals during working together in the groups or clubs called "Amalima" (Isindebele word for working together). Amalima, was viewed similarly to community members paying a subscription for membership of bodies like the Zimbabwe Farmers Union (ZFU) or any other social club.

Responses from the FGDs also highlighted these benefits, with participants indicating that most households use *L. camara* as a hedge around their homes, vegetable gardens and crop fields, thus reducing the cost of fencing that uses purchased wire and poles. The FGDs revealed that *L. camara* hedges around homesteads offer protection for poultry against birds of prey (raptors) such as *Polyboroides typus* and *Circaetus cinereus*, among others. Respondents also indicated that the dry twigs from *L. camara* come in handy for kindling for cooking fires. Thin sticks pruned from *L. camara* were said to be beneficial in making portable fowl cages which are used when a hen is brooding or when transporting chickens for sale.

The value of the mean benefits for the households were found to be US$ 716 ± 470 per year. The high variability is because a small proportion of household obtain very high benefit values, whereas most do not. The high benefit range of US$ 1501-1800 had the least percentage of respondents followed by the benefit value range of US$ 301-600, while the rest of the ranges had a percentage of households ranging between 18 and 30 (Figure 3.6).

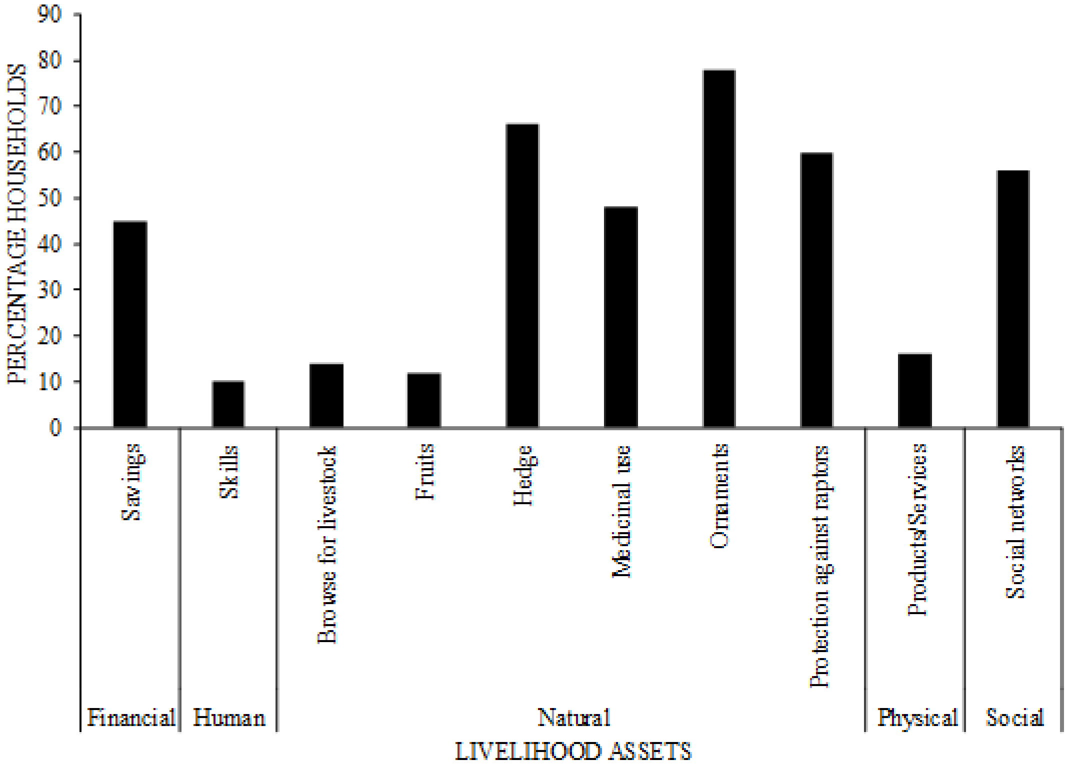

Figure 3.5: The proportion of households reporting positive effects of *L. camara* on rural livelihood assets

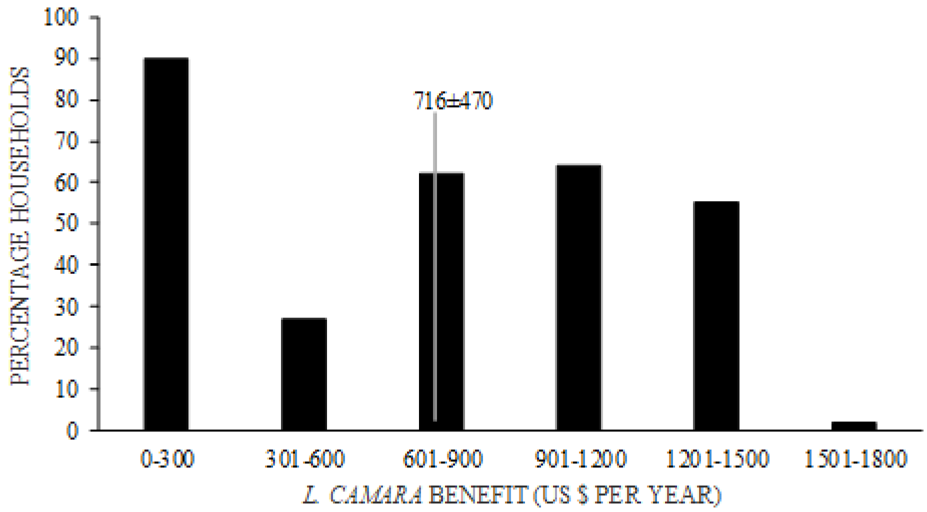

Figure 3.6: The proportion of households reporting benefits associated with a *L. camara* invasion. The line on the graph represents the benefits mean ± standard deviation

3.3.3.3. Net cost-benefit analysis

Overall, the benefits outweighed the cost with the deduction of benefits from cost resulting in a negative mean net cost of US$ 234 per year (Figure 3.7). Cumulatively, the percentage of households with a negative net cost-benefit analysis was higher (62%) than those with a positive net cost-benefit analysis (38%).

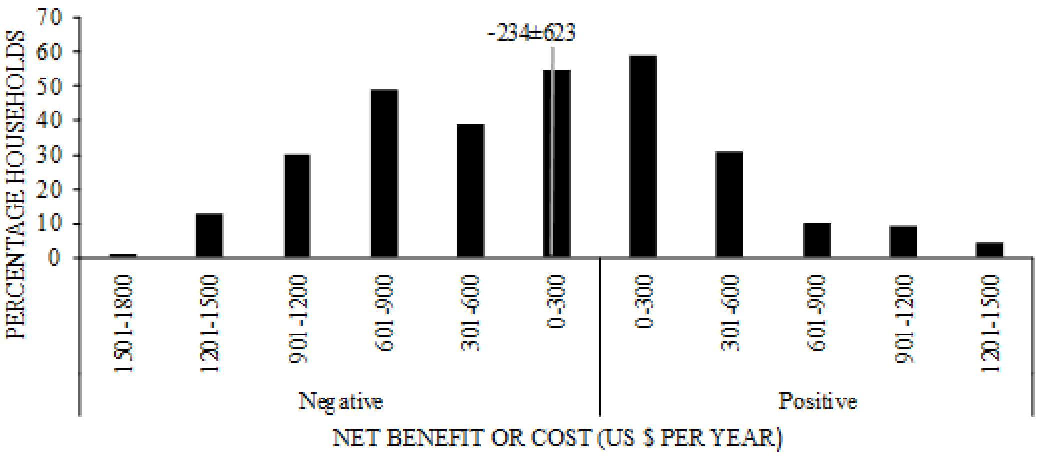

Figure 3.7: Distribution of household net cost-benefits associated with the *L. camara* invasion

3.3.4. Vulnerability

L. camara invasion makes people vulnerable to skin irritation, unforeseen financial loss, reduced land for crops, shortage of forage (pasture), displacement of valuable native plants, increased management costs, transport costs to attend training, and rodent crop damage, which would erode savings. Additionally, *L. camara* invasion contributes to the shortage of native wood for making various implements such as hoe handles and yokes. Overall, skin irritations, financial losses, and reduced cropping land and pasture shortages had the highest means for the capitals, showing that respondents' vulnerability was more linked to those three as shown in Table 3.3. Family displacements had the lowest mean, showing that respondents were less likely to vacate their original homesteads due to effects of *L. camara* (Table 3.3).

Table 3.3: Proportion of households in each vulnerability class (0 = Lowest; 3 = Highest)

Losses	**Percentage of households per vulnerability class/score**				
	0	**1**	**2**	**3**	**Mean ± SD**
Skin irritations	38	60	2	1	0.66 ± 0.55
Financial losses	41	44	14	1	0.65 ± 0.66
Reduced cropping land	42	55	2	1	0.62 ± 0.59
Pasture shortage	48	46	2	4	0.62 ± 0.72
Displacement of valuable native trees	50	47	3	0	0.53 ± 0.55
Increased management cost	50	47	2	0	0.52 ± 0.54
Transport costs	57	22	20	1	0.50 ± 0.74
Rodent crop damage	66	31	3	1	0.39 ± 0.59
Wood shortage	69	28	1	1	0.34 ± 0.57
Education disruption	84	7	6	4	0.29 ± 0.74
Loss of knowledge-sharing time	74	25	0	0	0.26 ± 0.46
Labour costs	77	21	2	0	0.25 ± 0.49
Disruption of bonding time	90	2	8	0	0.19 ± 0.57
Snake attacks	89	10	0	0	0.11 ± 0.32
Lack of access to burial sites	90	9	0	1	0.11 ± 0.37
Injuries	90	7	3	0	0.10 ± 0.37
Lack of access to cultural sites	94	5	1	0	0.07 ± 0.29
Reduced access to ritual sites	94	5	1	0	0.06 ± 0.27
Family displacements	99	1	0	0	0.01 ± 0.08

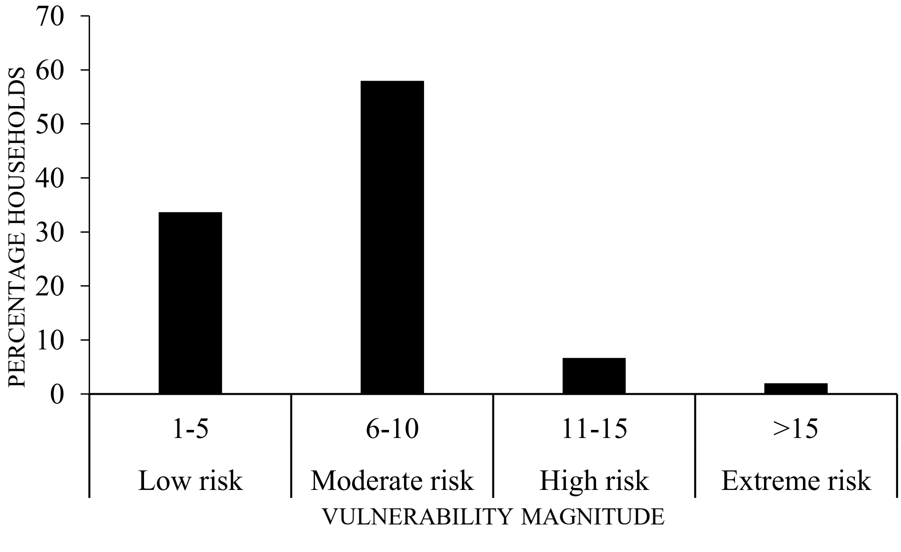

Figure 3.8: Distribution of vulnerability scores across 300 households

Most households (58%) were at moderate risk with a total vulnerability score of between 6-10, while a few (2%) were at extreme risk of vulnerability to the negative impacts resulting from the presence of *L. camara* (Figure 3.8).

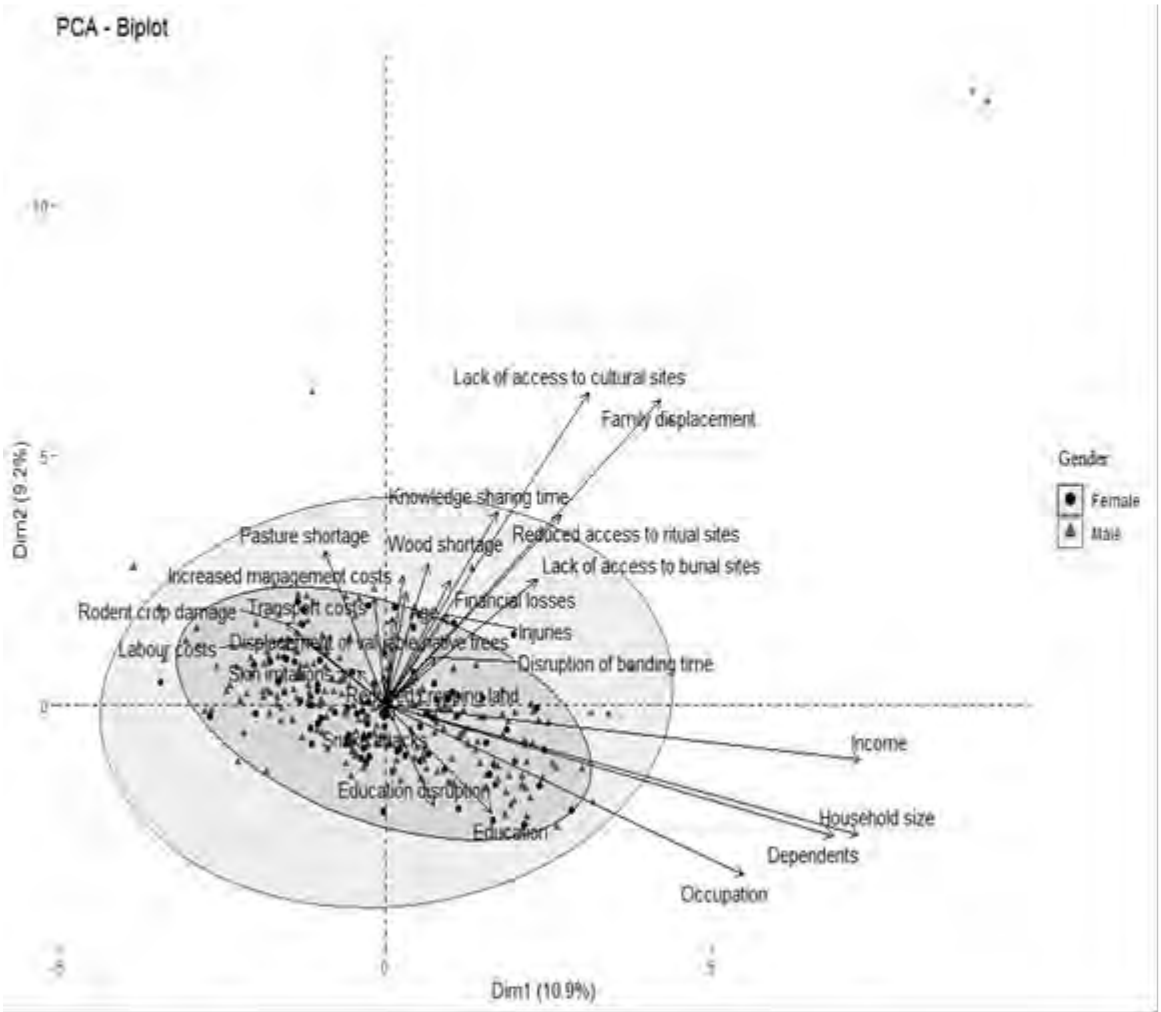

Figure 3.9: PCA showing the association of all variables in vulnerability to the presence of *L. camara*

Males were associated with increased vulnerability to reduced access to ritual and cultural sites, and to family displacements, pasture shortages and displacement of valuable NTFPs, compared to females, as shown in Figure 3.9. From the PCA plot (Figure 3.9) it is evident that the vulnerability of females was similar to the vulnerability of some males as shown by the ellipse of females housed within the one for males. Age was associated with loss of knowledge-sharing time, lack of access to cultural sites, family displacements, lack of access to burial sites, and pasture shortages. This association was shown by the proximity of the arrows, with the increase in age associated with the increase in vulnerability of these variables. The higher the education level, the more respondents indicated the increase in vulnerability to education disruptions (Figure 3.9). Other variables such as level of income, household size, number of dependents and occupation did not seem to have many relationships with the other vulnerability scores. Overall, the level of variance accounted for by the PCA was low.

3.4. Discussion

3.4.1. Invasion history of *L. camara*

The respondents in the Ndlovu community indicated that the start of the *L. camara* invasion was 1942 in the first ward right up to 1968 in the last ward sampled (Figure 3.2). To the best of our knowledge, there are no studies that have dealt with understanding the invasion history of *L. camara* in Zimbabwe or part of the Kavango Zambezi (KAZA) Transfrontier Conservation Area (TFCA). Establishing an understanding of the invasion trajectory of invasive plants has been recommended as an important way of contributing to developing adaptive management strategies (Bhagwat et al., 2012).

Literature on the history of invasion shows there are other countries where invasion happened around the same time (within about 20 years of either the first recording or last recording in this study), and these include the Island of Rodrigues (Kueffer et al., 2004) where initial records of invasion date back to 1930, Cooks Island in 1969 (Fosberg, 1972) and Fiji in 1971 (Thaman, 1974). For other countries, invasions date back to the 19th century, and this includes South Africa with a record of 1858, India in 1807 and Australia in 1841. These invasions were related to the pattern of colonisation as L. camara was spread due to its appeal as an ornamental (Kannan et al., 2013). There is also poor recording of this aspect of the history of the invasion of *L. camara* in several countries of note such as Indonesia, Kenya, Nicaragua, Philippines, Rwanda, Sri Lanka, Tanzania, Trinidad, Turkey and Uganda, where, despite the records of associated impacts and extent of invasion, there is no record of when *L. camara* was first seen (Vardien et al., 2012). These indications of invasion periods ought not be compared to the invasion period of a particular study site in Zimbabwe as such a comparison may result in a broad extrapolation and generalisation.

The FGDs held with 19 elders older than 80 years show a generational gap in terms of understanding the invasion history of *L. camara* in the study site. The elders represent an undocumented source of important information. While they could have been accurate with the events leading to the invasion and the period of invasion establishment, there is a possibility of erosion of knowledge over the years – knowledge which could have been supported by photographs had they been available. Early records of *L. camara* literature specific to Zimbabwe are documented by Shone and Drummond (1965) and Wild (1955). These years are close to the ones indicated by the elderly respondents on first sightings in the Hwange District, probably implying that *L. camara* could have been brought to Zimbabwe close to those years.

This study further revealed that the introduction of *L. camara* in the area was by local residents for their personal use as ornamentals and as a hedge. This is in line with Ghisalberti (2000), who states that the introduction of *L. camara* in many countries was by the locals to benefit from its various uses as a medicinal plant, hedge plant and ornamental plant, and for mulch. In some Asian countries, such as India, *L. camara* was initially solely introduced for ornamental purposes, which was recorded in 1807 (Kannan et al., 2013; Negi et al., 2019). While the locals in the Ndlovu area did not sell it, they shared it through cuttings, seeds and seedlings from one household to the other, hence it continued to spread to other wards and ecosystems within the area.

While the locals are responsible for introducing *L. camara* in the area (mainly around their homesteads, crop fields and garden fields), it soon self-established by spreading to the grazing areas and riverine sites. Akin-Fajiye and Akomolafe (2021) state that some of the factors contributing to the self-establishment of *L. camara* include disturbance, and appropriate climatic conditions. This may be the case with the Ndlovu community as it experiences various disturbances such as overgrazing and abandonment of crop fields, as well as climatic conditions which are favourable for *L. camara*, i.e., high temperatures with an average rainfall of less than 600 mm per annum (Moyo et al., 2012). According to Shackleton et al. (2020) the two main pathways of IAPs introductions are firstly, deliberately for perceived uses, and, secondly, accidentally if it finds its way to the surroundings. My study supports both pathways because the locals intentionally introduced *L. camara* for its uses, and it escaped into the surroundings. Eventually, it probably self-established and spread to other areas such as the riverine and grazing areas where it was not intentionally introduced.

3.4.2. Costs and benefits of *L. camara*

The IAPs are known to have negative and positive impacts in the areas where they have invaded (Reynolds et al., 2020). Similarly, this study shows negative and positive impacts on rural livelihoods. Rural livelihoods are dependent on a variety of sources as there are few or no formal employment opportunities in rural areas. The lack of formal employment in rural areas makes it important for rural people to rely on agricultural production (mainly crop cultivation and livestock production) as well as the use of natural resources (Islam and Quli, 2017; Kamwi et al., 2020). The five livelihood asset classes of the Sustainable Livelihoods Framework (financial, human, natural, physical and social) were examined to reveal a critical role in how rural people survive.

In parts of Africa and Southeast Asia, *L. camara* has been associated with the loss of crop yields (Barik et al., 2020). The main loss realised in this study was through the reduced crop yields and grazing land, as indicated by 49% of respondents (Figure 3.3). The reduction of crop yields came as a result of the invasion by *L. camara* of land under cultivation in a traditional farming area, thereby, without constant clearing efforts, rendering the land unproductive and inaccessible. The same applies to grazing land where the encroachment of *L. camara* reduced the availability of grazing area and sometimes impacted grass species required for fodder. In rural areas many farmers rely on natural grazing to feed the livestock, therefore having to purchase supplementary fodder worsens and erodes the financial well-being of the households.

In East Africa, it was noted that the invasion also impacted the availability of natural resources, such as traditional medicinal plants, and the general ability to move animals and people (Shackleton et al., 2017). In the KAZA area, in communities affected by the invasion of *L. camara,* 20% of respondents indicated that it caused skin irritations and injuries (Figure 3.3). Similarly, respondents in a study in Tanzania indicated that contact with *L. camara* leaves caused skin irritations and injuries (Mushi, 2019). The greater impact of IAPs such as *L. camara* is noticeable in its negative impacts on the natural capital of households. *Lantana camara* is generally recognised as a major disrupter which can have social-ecological effects where it invades. In some instances, communities may have to relocate from their traditional areas as well as to abandon fields. About 45% of respondents indicated that the *L. camara* impacted their grazing land negatively, thereby reducing the much-needed grazing land which provides fodder for their livestock (Figure 3.3).

About 40% of respondents further indicated that the invasion of *L. camara* harmed the survival and diversity of native plants. This is in line with results from Pothohar in northeastern Pakistan, where *L. camara* was found to impact natural biomes by displacing native species

(Qureshi et al., 2020). This reduction in biodiversity disrupts system dynamics and, as a result, attracts rodents, mice and snakes.

The issue of physical capital is core to the livelihoods of rural communities, and the ability to harness and utilise such assets is key to improving the rural livelihoods. In this regard, more than 30% of respondents indicated that their harvest and productive capacity were negatively impacted by the invasion of *L. camara* (Figure 3.3). Lack of access to the land resource negatively impacts productivity and reduces potential harvest per household. It is noted that the agricultural sector is very important in maintaining rural livelihoods, as approximately 80% of jobs in the poorest African, Asian and Latin American communities rely on agricultural land to sustain livelihoods (Ayana et al., 2021). Furthermore, results from the regression analysis (section 3.4.) showed that there is a relationship between the costs of the presence of *L. camara* and the land holding size of each respondent. Individuals who owned larger pieces of land for their homestead and cropfield, felt the negative impacts of the presence of *L. camara* more than those with smaller pieces of land. This may be due to the higher probability of occurrence of *L. camara* in the larger pieces than the smaller pieces.

When invasive species occupy an area, community members may start using the invasive species since they depend on natural resources for their livelihoods. This also brings about positive benefits in terms of adding to knowledge and skills in the affected communities. When local communities are empowered with knowledge of the invasion, they can control and, as a result reduce, the negative effects on their livelihoods. Community education is considered important in the control of the IAPs (Shrestha et al., 2019). Such knowledge, once imparted to households, remains valuable knowledge for the future management of IAPs. In the KAZA study area, community interaction in the passing and understanding of the invasive *L. camara* is viewed as a positive aspect in assisting the affected communities to understand the behaviour and mechanisms of controlling IAPs.

It should be appreciated that rural livelihoods are significantly improved by any small savings derived from the use of available natural resources as a substitute to conventional alternatives, as shown in Figure 3.5. The use of *L. camara* twigs for fuel as fire kindlers and the adaptive use of the same for medicinal use and aromatic flavouring of drinking water are some examples. Similarly, the blueberries of *L. camara* are consumed (Tsegay and Lemma, 2020); however, studies show that they have adverse effects when consumed in large quantities especially when consumed while they are still not ripe (Tull, 2021). Also, *L. camara* produces chemicals that are harmful to both humans and animals despite its recorded antimicrobial attributes (Shah et al., 2020).

In the study area, it is estimated that the presence of *L. camara* costs each household an average of US$ 483 per annum. This is a high cost considering that the poverty datum line (PDL) in Zimbabwe stands at an average of US$ 2 per person per day (Zhanda et al., 2022). This means that the proportion of the cost of *L. camara* cost relative to the PDL is more than 50%. This resonates well with a study carried out in East Africa which showed that an average household incurred a cost between US$ 400-500 per annum towards managing the negative effects of *L. camara* (Shackleton et al., 2017). Over 40% (Figure 3.5) of respondents indicated that having *L. camara* within their premises as a hedge around crop fields or vegetable gardens provided a less expensive alternative as opposed to the use of wire fence. Each household accrued benefits of an average of US$ 716 by using *L. camara* (Figure 3.6). It is further noted that livelihoods in the area are dependent on income and savings accrued from working on the pieces of land in their possession; any minimal amount of savings goes a long way to improve overall livelihoods.

In the study area, fewer than 15% of respondents (Figure 3.5) saw *L. camara* as positively impacting their livelihoods. From this, it is clear that there are no major inherent benefits derived from the use of *L. camara* in the conversion of raw materials into finished products, which is usually a key aspect in the adoption of the IAP as a useful physical capital. The people in the study area are not impacted in a significant way that influences the conversion of the IAP to benefit their livelihoods in the study area. The main use of *L. camara* is the production of fowl cages, and as a hedge with inherent savings, as is highlighted in Table 3.2.

In comparison to the experiences in other countries, such as India where a whole industry has sprouted to enable the conversion of *L. camara* into products such as baskets, couches and other items that are sold on the open market and improve the lives of the impacted people (Negi et al., 2019), this has not been the case in the study area. Either the invasion in the four wards has not grown to an extent where it can support such a community industry, or the skills and expertise have not been developed to a level that enables such a vibrant community-driven industry.

Results from this study show that respondents did not see any noticeable negative impact on their social capital (Figure 3.3). This relates to affecting their rituals, taboos and day-to-day social activities. In some countries, *L. camara* impacted the ability of communities to carry out ritual activities and access areas of cultural significance where important traditional functions have been carried out generations (Souza et al., 2018). In some instances, people have been forced to sell part of their properties due to the adverse effects of *L. camara*, with some respondents interviewed in Uganda indicating they had to relocate from their traditional land (Kagonji village) and find new land to settle after being pushed out by the adverse effects of *L. camara* (Shackleton et al., 2017).

Such adverse effects have not been recorded or are not yet visible in the study area. This may be interpreted two ways. The first possibility is that *L. camara* has not reached a phase where it has become aggressive, as shown by Shackleton et al. (2007) in the conceptual framework for the interpretation of the impacts of IAS on rural livelihoods. The second possibility may be that the conditions are not conducive enough for *L. camara* to spread aggressively, so it may not ever reach that phase where it is extremely aggressive. It is over 75 years since *L. camara* invaded the study site, but the communities still consider its benefits as well as its costs.

Over half (55%) of respondents in the study area indicated that *L. camara* had positively impacted their ability to develop social networking systems (Figure 3.5). This was due to the improved relationships created between the local District management initiatives represented by the Environmental Management Agency, and resulted in community cohesion as the ward members interacted, shared knowledge and improved their understanding of the impact of *L. camara*. Such networking initiatives would benefit other community social improvement initiatives such as teamwork (Amalima) in tackling problems affecting the ward, such as those listed in Figure 3.5. It should be noted that the social benefits derived through the presence of *L. camara* in the study area do not seem to create any significant financial benefits, as shown by the low financial value derived from its presence in the ward.

3.4.3. Vulnerability

There are anticipated gender-related differences in vulnerability to invasions in rural communities, based on differences that exist in ownership and access to assets, resources, education, cultural practices in an area, and decision making, among other issues (Fish, 2010). However, results from this research showed that there were no significant differences between genders on the level of vulnerability to shocks and stresses brought about by the invasion of *L. camara*. This means both genders are more or less equally vulnerable. In line with findings from this study, Rai et al. (2012) found no significant differences between perceived impacts of IAPs between males and females, and attributed this to the effects being viewed at the household level rather than at the individual level. This may be a similar case to this study. Additionally, this may be due to the social dynamic changes occurring in most communities over the years where gender emancipation has been part of most community development initiatives; gender emancipation has made women and men equally responsible for their livelihoods, hence they will suffer similar shocks and stresses.

The analysis showed that those with a higher level of education were associated with perceptions of higher vulnerability to shocks and stresses caused by *L. camara*. This may be associated with the assertion by Marshall et al. (2011) that they associated education and level of experience with better environmental awareness and vulnerability to change. This shows that knowledge plays a major role in respondents accepting that *L. camara* encroachment into the land where it is not required makes them more vulnerable to shocks and stresses that impact their livelihoods. According to Shackleton et al. (2019), the harm to livelihoods and increase in vulnerability is generally through the intrusion of the invader species on productive land, and this limits movement or access to such areas and also limits the use of the valuable land resource. This is in line with these results, where differences in vulnerability are found in the physical resource which is associated with crop productivity. Most respondents are farmers who depend on growing crops and the level of education of an individual may make them more aware of the costs associated with loss of crop productivity due to *L. camara* occupying the areas where they would harvest more crops from. Nguyen et al. (2020) states that those with a tertiary education level within communities associate the presence of *L. camara* with a higher economic risk. This may be due to the negative impacts brought about by *L. camara* where it can invade the grazing land or crop field, thereby reducing the land value.

The proliferation of *L. camara* in areas that are adjacent to homesteads, such as communal lands traditionally designated for livestock foraging, has the effect of reducing the cyclical rejuvenation of grasses that is important in providing a source of natural forage. This is because the invasion of such areas by *L. camara* and other woody species tends to promote the growth of woody species and hinder the growth of grass species important for fodder (Hare et al., 2021; White et al., 2022). Such a reduction of natural grass fodder affects the livelihoods of communities as there is a need to look for other means for providing supplementary fodder (Hare et al., 2021). In Southern African ecosystems, such as in South Africa, the most vulnerable affected groups are the poor and the low-income groups whose livelihoods are interlinked to an undisturbed ecosystem that promotes biodiversity (White et al., 2022). Globally the increased coverage by shrub species in traditionally grass rangeland systems negatively impacts species diversity, and subsequently negatively impacts both the ecology and the livelihoods of communities (O'Connor et al., 2020; White et al., 2022). Apart from reducing soil water retention, reduced grass species composition has the downstream effect of impacting the carbon sequestration in rangeland ecosystems, and this is ecologically important in savanna rangelands (Ghosh and Maiti, 2021; White et al., 2022).

The encroachment of woody species such as *L. camara,* therefore, tends to negatively affect vulnerable and poor rural communities as they find themselves having to allocate resources towards offsetting the impacts of such invasions. Such communities are more reliant on the extraction and use of resources from the natural environment (Reid, 2016; White et al., 2022) than investing in resources. With the resultant increase in woody species in rangelands there has been a need to strike a balance in management between livelihood sustenance and biodiversity (White et al., 2022). It is further noted that the gradual encroachment of woody species is associated with overgrazing which increases ecosystem stress and promotes proliferation of IAPs to the detriment of grasses (Malan et al., 2021; White et al., 2022; Yan and Liu, 2021).

In this study, older respondents were not associated with vulnerability to the shocks and stresses of most of the impacts on livelihood assets. This is contrary to findings by Nguyen et al. (2020); their findings showed older respondents' perceptions were of them being at higher risk of invasive plants. In a study carried out in Hawaii on understanding who was vulnerable to IAPs, it was found that long-term connections to a particular environment, generally referred to as natural bonding, made age a great determinant in understanding the long-term vulnerability effects of IAPs (Niemiec et al., 2017). This means that perceptions of vulnerability will vary depending on the value placed on the species in the environment over the years. In the case of my study, older respondents may have had more years benefiting from the use of *L. camara* such that they consider it naturalised in the area and consider it more beneficial than younger respondents do. It was understandable for youths to express vulnerability to the presence of *L. camara* in grazing areas as they are the ones who take cattle out for grazing. Considering its patchy distribution in the wards, some grazing areas will not have *L. camara* and the youths who herd notice how easy it is to move with cattle in areas without *L. camara* thicket, and that there is more space occupied by forage, too.

3.5. Conclusion and recommendations

This study provides information necessary for designing tailor-made strategies for managing *L. camara* in the KAZA TFCA. An understanding of the history of invasion gives insight into how long *L. camara* has been in the area, its trajectory of spread and, depending on the period of invasion, how communities are feeling the effect of the invasion. The occurrence was deliberate, brought in by community members for their benefit, and this has implications for them viewing the plant as a nuisance or seeing the possible detrimental effects of the invasion. Most households experience nett negatives, but the overall mean value across the sample indicate benefits outweigh costs, because a minority of households are experiencing significant benefit with high numbers, whereby the mean value is skewed. The common factor shared by the households experiencing nett negatives is their proximity to areas with high cover or densities of *L. camara*. The situation is complex and, even at local levels, some households gain and some lose, and therefore it is difficult to generalise. The social dynamics associated with vulnerability also show which groups express more vulnerability to the shocks and stresses associated with *L. camara*, and this is essential as it will assist in identifying where efforts should be directed to raising awareness of IAPs. Perceptions may be due to a lack of forecast of the future detriment caused by continued invasion by *L. camara*. Notice must also be taken that community members may not attribute their loss of crop productivity only to the invasion of *L. camara.* In an area like Hwange District, where there is limited crop productivity, and where there are other pressures such as human-wildlife conflict, community members not may attribute their loss of crop productivity not only to the presence of *L. camara.* This makes these community members view *L. camara* more for its benefits than for its negative impacts. Recommendations of this study are to focus further on the negative impacts, coupled with raising the awareness of the invasion by IAPs and its future implications, to enable the mapping of management interventions with community members that are aware of the consequences, and factor in holistic considerations. Management by use should also be something to consider since communities are already benefiting from the presence. Overall, a careful approach is required. This approach should involve further long-term studies on the ecology of *L. camara*, balanced by adopted community use of *L. camara*. While these recommendations are yet to be discussed and possibly adopted, it may be rewarding to manage by use.

References

Akin-Fajiye, M. and Akomolafe, G.F., 2021. Disturbance is an important predictor of the distribution of *Lantana camara* and *Chromolaena odorata* in Africa. *Vegetos*, 34(1), 42-49.

Alghamdi, A., Basher, N. and Hamed, N., 2020. Efficacy of leaves and flowers ethanol extracts of the invasive species *Lantana camara* Linn as a mosquito larvicidal. *International Journal of Mosquito Research* 7, 43-47.

Atyosi, Z., Ramarumo, L.J. and Maroyi, A., 2019. Alien plants in the Eastern Cape Province in South Africa: Perceptions of their contributions to livelihoods of local communities. *Sustainability*, 11(18), 5043.

Ayana, G.F., Megento, T.L. and Kussa, F.G., 2021. The extent of livelihood diversification on the determinants of livelihood diversification in Assosa Wereda, Western Ethiopia. *Geo Journal*, 87(4), 2525-2549.

Barik, S.S., Sahoo, R.P. and Barik, S.S., 2020. *Lantana camara* L.: An emerging threat to native flora and livestock: A review. *Journal of Pharmacognosy and Phytochemistry*, 9(5), 2363-2366.

Bhagwat, S.A., Breman, E., Thekaekara, T., Thornton, T.F. and Willis, K.J., 2012. A battle lost? Report on two centuries of invasion and management of *Lantana camara* L. in Australia, India and South Africa. *PLoS One*, 7(3), 32407.

Breen, R.L., 2006. A practical guide to focus-group research. *Journal of Geography in Higher Education*, 30(3), 463-475.

Corlett, R.T., 2020. Safeguarding our future by protecting biodiversity. *Plant Diversity*, 42(4), 221-228.

Creswell, J.W. and Creswell, J.D., 2018. *Research design: Qualitative, quantitative, and mixed methods approaches*. 5th edition. Sage Publications.

Culver, M. and Precious, R.M., 2018. Efficacy of *Lantana* (*Lantana camara*) extract application against aphids (*Brevicoryne brassicae*) in rape (*Brassica napus*) over varied periods of time. *African Journal of Biotechnology*,17(8), 249-254.

Fish, J., 2010. *Mainstreaming gender into prevention and management*. Global invasive Species Programme (GSIP), Nairobi, Kenya.

Fosberg, F.R., 1972. List of vascular plants (reef islands of Ratoronga). *Atoll Research Bulletin*, 160, 9-14.

Ghisalberti, E.L., 2000. *Lantana camara* L. (Verbenaceae). *Fitoterapia*, 71(5), 467-486.

Ghosh, D. and Maiti, S.K., 2021. Biochar-assisted eco-restoration of coal mine degraded land to meet United Nation Sustainable Development Goals. *Land Degradation & Development*, 32,16, 4494-4508.

Gusha, J., Masocha, M., Muchaya, K. and Ncube, S., 2016. Chemical analysis of the potential contribution of *Lantana camara* to the nutrition of browsing livestock. *Tropical and Subtropical Agroecosystems*, 19(3).

Gundumogula, M., 2020. Importance of focus groups in qualitative research. *International Journal of Humanities and Social Science*, 8(11), 299-302.

Hare, M.L., Xu, X.W., Wang, Y.D., Yuan, Y. and Gedda, A.E., 2021. Do woody tree thinning and season have effect on grass species' composition and biomass in a semi-arid savanna? The case of a semi-arid savanna, southern Ethiopia. *Frontiers in Environmental Science*, 9, 692239.

Ibnouf, F., 2012. The value of women's indigenous knowledge in food processing and preservation for achieving household food security in rural Sudan. *Journal of Food Research*, 1(1), 238.

Islam, M.A. and Quli, S.M., 2017. The role of non-timber forest products (NTFPs) in tribal economy of Jharkhand, India. *International Journal of Current Microbiology and Applied Sciences*, 6(10), 2184-2195.

Jevon, T. and Shackleton, C.M., 2015. Integrating local knowledge and forest surveys to assess *Lantana camara* impacts on indigenous species recruitment in Mazeppa Bay, South Africa. *Human Ecology*, 43(2), 247-254.

Kamwi, J.M., Endjala, J. and Siyambango, N., 2020. Dependency of rural communities on non-timber forest products in the dry lands of southern Africa: A case of Mukwe Constituency, Kavango East Region, Namibia. *Trees, Forestry and People,* 2, 100022.

Kannan, R., Shackleton, C.M. and Shaanker, R.U., 2013. Playing with the forest: Invasive alien plants, policy and protected areas in India. *Current Science*, 104, 1159-1165.

Kannan, R., Shackleton, C.M. and Shaanker, R.U., 2014. Invasive alien species as drivers in socio-ecological systems: local adaptations towards use of *Lantana* in Southern India. *Environment, Development and Sustainability*, 16(3), 649-669.

Kueffer, C., Vos, P., Lavergne, C. and Mauremootoo, J., 2004. Case studies on the status of invasive woody plant species in the Western Indian Ocean. 1. Synthesis. *Forest Health and Biosecurity Working Paper FBS/4-1E*. Food and Agriculture Organization of the United Nations.

Malan, P.W., Tiawoun, M.A., Molatlhegi, K.S. and Materechera, S.A., 2021. Effect of encroaching woody plant species on soil nutrients and selected soil chemical properties in communally managed semiarid savanna grazing lands in the North West Province, South Africa. *South African Journal of Plant and Soil*, 38(1), 27-35.

Maroyi, A., 2009. Traditional homegardens and rural livelihoods in Nhema, Zimbabwe: A sustainable agroforestry system. *International Journal of Sustainable Development World Ecology*, 16(1), 1-8.

Marshall, N.A., Friedel, M., van Klinken, R.D. and Grice, A.C., 2011. Considering the social dimension of invasive species: The case of buffel grass. *Environmental Science and Policy*, 14(3), 327-338.

Maunguja, A.B., 2016. *Assessment of plant diversity and utilization of wild medicinal species by households proximate to Arabuko Sokoke Forest in Kilifi County of Kenya.* Doctoral Thesis: University of Nairobi, Kenya.

Mhlongo, E.S., 2021. *Perceptions of the role of* Lantana camara *on human well-being and rural livelihoods in Vhembe Biosphere Reserve, South Africa*. Doctoral Thesis: Rhodes University, South Africa.

Mkindi, A.G., Coe, R., Stevenson, P.C., Ndakidemi, P.A. and Belmain, S.R., 2021. Qualitative cost-benefit analysis of using pesticidal plants in smallholder crop protection. *Agriculture*, 11(10), 1007.

Moyo, M., Mvumi, B.M., Kunzekweguta, M., Mazvimavi, K., Craufurd, P. and Dorward, P., 2012. Farmer perceptions on climate chance and variability in semi-arid Zimbabwe in relation to climatology evidence. *African Crop Science Journal*, 20, 217-335.

Muijeen, K., Kongvattananon, P. and Somprasert, C., 2020. The key success factors in focus group discussions with the elderly for novice researchers: A review. *Journal of Health Research*, 34, 359-371.

Mukul, S.A., Uddin, M.B. and Tito, M.R., 2006. Study on the status and various uses of invasive alien plant species in and around Satchari National Park, Sylhet, Bangladesh. *Tigerpaper*, 33(4), 28-32.

Mungi, N.A., Qureshi, Q. and Jhala, Y.V., 2020. Expanding niche and degrading forests: Key to the successful global invasion of *Lantana camara* (sensu lato). *Global Ecology and Conservation*, 23, 01080.

Mushi, G.V., 2019. *Local communities' perceptions on* Lantana camara *and management responses in East Usambara, Tanzania*. Doctoral Thesis: Sokoine University of Agriculture, Tanzania.

Nawaz, A., Ayub, M.A., Nadeem, F. and Al-Sabahi, J.N., 2016. *Lantana* (*Lantana camara*): A medicinal plant having high therapeutic potentials–a comprehensive review. *International Journal of Chemical and Biochemical Sciences*, 10, 52-59.

Negi, G.C.S., Sharma, S., Vishvakarma, S.C.R., Samant, S.S., Maikhuri, R.K., Prasad, R.C. and Palni, L.M.S., 2019. Ecology and use of *Lantana camara* in India. *Botanical Review*, 85(2), 109-130.

Nguyen, N.A., Eskelson, B.N.I., Meitner, M.J. and Murray, T., 2020. People's knowledge and risk perceptions of invasive plants in metro Vancouver, British Columbia, Canada. *Environmental Management*, 66(6), 985-986.

Niemiec, R.M., Ardoin, N.M., Wharton, C.B. and Brewer, F.K., 2017. Civic and natural place attachment as correlates of resident invasive species control behavior in Hawaii. *Biological Conservation*, 209, 415-422.

Ntalo, M., Ravhuhali, K.E., Moyo, B., Hawu, O. and Msiza, N.H., 2022. *Lantana camara*: Poisonous species and a potential browse species for goats in Southern Africa—A review. *Sustainability*, 14(2), 751.

Nyumba, O.T., Wilson, K., Derrick, C.J. and Mukherjee, N., 2018. The use of focus group discussion methodology: Insights from two decades of application in conservation. *Methods in Ecological Evolution*, 9(1), 20-32.

O'Connor, R.C., Taylor, J.H. and Nippert, J.B., 2020. Browsing and fire decreases dominance of a resprouting shrub in woody encroached grassland. *Ecology*, 101(2), e02935.

Patel, S., 2011. A weed with multiple utility: *Lantana camara*. *Review in Environmental Science Bio/Technology*. 10(4), 341-351.

Priyanka, N. and Joshi, P.K., 2013. A review of *Lantana camara* studies in India. *International Journal of Science and Research Publications*, 3(10), 1-11.

Qureshi, H., Anwar, T., Habib, N., Ali, Q., Haider, M.Z., Yasmin, S., Munazir, M., Basit, Z. and Waseem, M., 2020. Multiple comparisons of diversity indices invaded by *Lantana camara*. *Brazilian Journal of Biology*, 81, 83-91.

Rai, R.K., Scarborough, H., Subedi, N. and Lamichhane, B., 2012. Invasive plants – Do they devastate or diversify rural livelihoods? Rural farmers' perception of three invasive plants in Nepal. *Journal of Nature Conservation*, 20(3), 170-176.

Rana, M.P. and Akhter, F., 2010. Uses of invasive alien plant species in Rema-Kalenga Wildlife Sanctuary of Bangladesh. *Journal of Mountain Science*, 7, 380-385.

Ranjan, R., 2019. Deriving double dividends through linking payments for ecosystem services to environmental entrepreneurship: The case of the invasive weed *Lantana camara*. *Ecological Economics*, 164, 106380.

Reid, H., 2016. Ecosystem- and community-based adaptation: Learning from community-based natural resource management. *Climate and Development*, 8(1), 4-9.

Reynolds, C., Venter, N., Cowie, B.W., Marlin, D., Mayonde, S., Tocco, C. and Byrne, M.J., 2020. Mapping the socio-ecological impacts of invasive plants in South Africa: Are poorer households with high ecosystem service use most at risk? *Ecosystem Services*, 42, 101075.

Reyns, N., Casaer, J., De Smet, L., Devos, K., Huysentruyt, F., Robertson, P.A., Verbeke, T. and Adriaens, T., 2018. Cost-benefit analysis for invasive species control: The case of greater Canada goose *Branta canadensis* in Flanders (northern Belgium). *Peer Journal*, 6, 4283.

Routray, S., Kabi, M., Debnath, D. and Palei, S., 2021. Potential of Lantana camara L. extracts as biopesticide against insect pests. *Journal of Entomological Research*, 4(3), 461-469.

Ruwanza, S., 2020. Effects of *Lantana camara* invasion on vegetation diversity and composition in the Vhembe Biosphere Reserve, Limpopo Province of South Africa. *Scientific African*, 10, 00610.

Ruwanza, S. and Shackleton, C.M., 2016. Effects of the invasive shrub, *Lantana camara*, on soil properties in the Eastern Cape, South Africa. *Weed Biology and Management*, 16, 67-79.

Shackleton, C.M., McGarry, D., Fourie, S., Gambiza, J., Shackleton, S.E. and Fabricius, C., 2007. Assessing the effects of invasive alien species on rural livelihoods: Case examples and a framework from South Africa. *Human Ecology,* 35(1), 113-127.

Shackleton, R.T., Bertzky, B., Wood, L.E., Bunbury, N., Jäger, H., van Merm, R., Sevilla, C., Smith, K., Wilson, J.R., Witt, A.B. and Richardson, D.M., 2020. Biological invasions in World Heritage Sites: Current status and a proposed monitoring and reporting framework. *Biodiversity and Conservation,* 29(11), 3327-3347.

Shackleton, R.T., Shackleton, C.M. and Kull, C.A., 2019. The role of invasive alien species in shaping local livelihoods and human well-being: A review. *Journal of Environmental Management*, 229, 145-157.

Shackleton, R.T., Witt, A.B., Aool, W. and Pratt, C.F., 2017. Distribution of the invasive alien weed, *Lantana camara*, and its ecological and livelihood impacts in eastern Africa. *African Journal of Range and Forage Science*, 34(1), 1-11.

Shah, M., Alharby, H.F., Hakeem, K.R., Ali, N., Rahman, I.U., Munawar, M. and Anwar, Y., 2020. De novo transcriptome analysis of *Lantana camara* L. revealed candidate genes involved in phenylpropanoid biosynthesis pathway. *Scientific Reports*, 10(1), 1-14.

Sharma, P.K., Sharma, A.K., Pulla, R.H. and Sahoo, P.K., 2020. Performance analysis of a medium-scale downdraft gasifier using *Lantana camara* biomass as feeding material. *Energy Sources*, *Part A: Recovery*, *Utilization*, *and Environmental Effects,* 1-15.

Shone, D.K. and Drummond, R.B., 1965. Poisonous plants of Rhodesia. *Rhodesia Agricultural Journal*, 62(4),1-64.

Shrestha, B.B., Shrestha, U.B., Sharma, K.P., Thapa-Parajuli, R.B., Devote, A. and Siwakoti, M., 2019. Community perception and prioritization of invasive alien plants in Chitwan-Annapurna Landscape, Nepal. *Journal of Environmental Management*, 229, 38-47.

Shrestha, S., 2021. Impact of invasive alien species and gender. *International Journal of Research*, 9, 78-84.

Souza, A.O., Chaves, M. D. P.S.R., Barbosa, R.I. and Clement, C.R., 2018. Local ecological knowledge concerning the invasion of Amerindian lands in the northern Brazilian Amazon by *Acacia mangium* (Willd.). *Journal of Ethnobiology and Ethnomedicine*, 14(1), 1-14.

Sundaram, B., Krishnan, S., Hiremath, A.J. and Joseph, G., 2012. Ecology and impacts of the invasive species, *Lantana camara*, in a social-ecological system in South India: Perspectives from local knowledge. *Human Ecology*, 40(6), 931-942.

Tarugara, A., Clegg, B.W. and Matuvhunye, R., 2022. Mapping the spatial distribution and canopy cover of Lantana camara in the Zaka District of Zimbabwe. *Scientific African*, 17, 01339.

Thaman, R.R., 1974. *Lantana camara*: Its introduction, dispersal and impact on islands of the tropical Pacific Ocean. *Micronesica Journal of University of Guam*, 10(1), 17-39.

Tsegay, Z.T. and Lemma, S.M., 2020. Response surface optimization of cactus pear (*Opuntia ficus-indica)* with *Lantana camara* (*L. camara*) fruit fermentation process for quality wine production. *International Journal of Food Science*, 2020.

Tull, D., 2021. 4. Edible and poisonous berries and other fleshy fruits. In D. Tull, *Edible and Useful Plants of the Southwest*, 172-225. University of Texas Press.

Turner, P.J. and Downey, P.O., 2008. The role of native birds in weed invasion, species decline, revegetation and reinvasion: Consequences for *Lantana* management. In R.D. van Klinken, V.A.Osten, F.D. Panetta and J.C. Scanlan (eds), *Proceedings of the 16th Australian Weeds Conference*, 30-2. Queensland Weeds Society, Brisbane, Queensland.

van Wilgen, B.W., Witt, A.B., Beale, T. and Kiambi, S., 2017. A preliminary assessment of the extent and potential impacts of alien plant invasions in the Serengeti-Mara ecosystem, East Africa. Koedoe: *African Protected Area Conservation and Science*, 59,1,1-16.

Vardien, W., Richardson, D.M., Foxcroft, L.C., Thompson, G.D., Wilson, J.R.U. and Le Roux, J.J., 2012. Invasion dynamics of *Lantana camara* L. (sensu lato) in South Africa. *South African Journal of Botany*, 81, 81-94.

Weidlich, E.W.A., Flórido, F.G., Sorrini, T.B. and Brancalion, P.H.S., 2020. Controlling invasive plant species in ecological restoration: A global review. *Journal of Applied Ecology*, 57(9), 1806-1817.

Westwood, T., 2020. *The potential conflict of interest associated with the management of* Rosa rubiginosa *L. (rosehip) in South Africa*. Doctoral Thesis: Rhodes University, South Africa.

White, J.D., Stevens, N., Fisher, J.T., Archibald, S. and Reynolds, C., 2022. Nature-reliant, low-income households face the highest rates of woody-plant encroachment in South Africa. *People and Nature*, 4(4), 1020-1031.

Wild, H., 1955. *Common Rhodesian weeds*, 220. Government Printers, Salisbury.

Worley, J.W., 2009. *Fences for the farm*. University of Georgia.

Yan, H. and Liu, G., 2021. Fire's Effects on Grassland Restoration and Biodiversity Conservation. *Sustainability*, 13(21), 12016.

Zende, M., Kativu, S. and Mhlanga, L., 2017. Impact of *Lantana camara* on vegetation, primary productivity and soils at Imire Ranch, Zimbabwe. *Kirkia*, 19(1), 101-109.

Zhanda, K., Garutsa, N., Dzvimbo, M.A. and Mawonde, A., 2022. Women in the informal sector amid COVID-19: Implications for household peace and economic stability in urban Zimbabwe. *Cities Health*, 6(1), 37-501-14.

Chapter 4: Passive restoration of native woody species and soil properties degraded by *L. camara* invasion in the KAZA TFCA

Abstract

Ecosystem restoration is a growing need in sites affected by invasive alien plants (IAPs). However, limited context-specific studies have been done to assess passive restoration following removal of IAPs. This study focused on the impacts of clearing *Lantana camara* on the composition of native woody species and soil properties in the Kavango Zambezi Transfrontier Conservation Area, specifically the Ndlovu Communal area and Victoria Falls Rainforest area. Clearing of *L. camara* was done by mechanical removal in two sites in 2010, and the cleared areas were left to recover unaided. Forty-two plots, were surveyed at both sites, including: (i) plots cleared of *L. camara,* (ii) plots still invaded with *L. camara*, and (iii) control plots that were never invaded. In each plot, native woody species were identified, stems counted, diameter at breast height measured, and canopy cover estimated. Soil samples were collected for analysis. One-way ANOVA was used to compare species richness and diversity among the Cleared, Invaded and Control plots in both sites. Non-Metric Dimensional System (NMDS) and cluster analyses were used to show the variation in species composition based on the treatments. There was a significantly higher woody species richness and diversity in the cleared and control plots compared to the invaded plots. NMDS and cluster dendrograms showed dissimilarity in species composition for invaded plots in comparison with the cleared and uninvaded plots. Most trees found in the cleared plots had a significantly lower DBH and canopy cover than those in the invaded and control plots, indicating their younger age. Passive restoration did not contribute to changes in soil attributes, based on no significant differences being observed in cleared and invaded plots for all soil properties. The study showed the regeneration of native vegetation in plots that were cleared of *L. camara*, demonstrating the effectiveness of clearing. However, the re-emergence of *L. camara* in previously cleared plots was noted, indicating the need for follow-up management.

Keywords: Invasive alien plants; Native woody species; Passive restoration; Soil properties

4.1. Introduction

Many ecosystems globally have been disturbed by the presence of invasive alien plants (IAPs) introduced either deliberately or accidentally (Mollot et al., 2017; Pyšek et al., 2020). Once establish, many IAPs alter the conditions to suit their proliferation (Balintulo, 2021; Lamsal et al., 2018). One of the ways they modify the environment may be by exuding chemicals into the soil, causing changes to the chemical composition of the soil and compromising the survival of native species (Latif et al., 2017; Souza-Alonso et al., 2017). Areas infested with IAPs, therefore, tend to experience major changes in plant composition, and vegetation structure and function, as well as the quality and quantity of forage (DiTomaso et al., 2017). An example of where this has occurred is in the Great Basin of North America. After the introduction of cheatgrass (*Bromus tectorum*) for grazing, it spread throughout the area, altered the environment, and eventually replaced the native vegetation (Mack, 1981). In Pakistan, *Parthenium hysterophorus* occupies cropping areas, impacting the agricultural productivity which is the main livelihood of most communities (Bajwa et al., 2019). In response to these and other related negative impacts of IAPs, efforts to reverse this damage by IAPs have been ongoing, with ecological restoration as one of the strategies.

Ecological restoration has been defined as a strategy that assists the recovery of ecosystems after they have been subjected to different forms of environmental degradation, including IAPs (Holmes et al., 2020; Martin, 2017). It is being practiced globally with proven ecological benefits, and has huge implications for socio-economic well-being, population health and preservation of indigenous knowledge (Aronson et al., 2020). The main types of restoration are 'passive' and 'active'. Passive restoration allows an ecosystem to recover without interference after the driving causes of degradation are removed (McIver and Starr, 2001; Zahawi et al., 2014). In contrast, active restoration involves human involvement through management, whereby interventions such as tree planting, burning and weeding are intensively applied to reach a desired outcome of recovery within a shorter period of time (Shono et al., 2007).

The control of IAPs is one of the most common and important ecological restoration interventions in many ecosystems due to the role that IAPs play in hindering native vegetation growth (Brancalion et al., 2019; D'Antonio et al., 2016; Prior et al., 2018). Therefore, a primary step in ecological restoration is clearing the IAP, and the success of the intervention is determined by the recovery of the native vegetation composition and structure (Blanchard and Holmes, 2008). The main physical methods of IAP removal include cutting the plants, uprooting them, mowing, or stripping the bark, among others (Hoare, 2021). Other methods of removal encompass chemical means, biological control and, in some instances depending on the species and level of infestation, a combination of these methods (Harvey-Samuel et al., 2017; Hussner et al., 2017; Sher et al., 2018). For example, physical and chemical removal of IAPs such as *Cylindropuntia fulgida* was used for years with limited success in South Africa until a host-specific insect was introduced in 2011, which yielded success (Zachariades et al., 2017; Zozo, 2022).

The choice of whether to use passive or active restoration to manage a specific IAP is based on a critical analysis of the best method that will result in effectiveness and efficiency, relative to the available human and financial resources (Mostert et al., 2018). Generally, passive restoration of IAPs is preferred because it is cheaper than active restoration (Prach et al., 2020). However, there are instances of re-emergence of IAPs after some time (Froeschlin et al., 2022). This was observed by Froeschlin et al. (2022) where *Acacia saligna* was found to re-establish in sites where it was previously removed in South Africa's Cape Floristic Region, highlighting the need for further interventions.

A review by Prior et al. (2018) focused on 151 studies in which IAPs were removed, with subsequent tracking of the ecological recovery. From this review, 51% of the studies showed positive ecological outcomes after the removal of IAPs, while 20% of the studies showed negative changes such as increased soil erosion and reduced retention of nutrients, 18% showed mixed outcomes, and the smallest percentage of 11% showed no significant changes. Removal of *Tamarix* species from riparian ecosystems in Western Colorado, United States of America (USA), was found to be positively correlated with the recovery of the native plant community (Sher et al., 2018). Similarly, Zimmerman et al. (2018) in New York, USA, showed the recovery of native plant communities in plots that were monitored after the removal of *Phragmites australis*. The benefits of restoration efforts have also been recorded in Southern Africa. In particular, extensive studies conducted in South Africa showed recovery of vegetation and soil properties, as evidenced by an increase in native species, and by enhanced soil nutrients (Kerr and Ruwanza, 2016; Mhlongo, 2021; Ndou and Ruwanza, 2016; Ravhuhali et al., 2021; Ruwanza and Dondofema, 2020; Ruwanza and Tshillilo, 2019). In South Africa, research on restored plots after the removal of a heterogenous mixture of invaders, namely *Acacia saligna, Eucalyptus conferruminata, E. cladocalys, E. gomphocephala* and *Paraserianthes lophantha*, also showed good re-establishment of native vegetation after passive restoration efforts (Froeschlin et al., 2022).

Lantana camara is an example of an IAP that displaces indigenous flora, and it has been shown to do this by altering the soil chemical composition where it has invaded (Kumar et al., 2021; Ruwanza and Shackleton, 2016). A study in Nepal also showed removal of *L. camara* contributed to improving water quality and flow, increased microbial activity, and supported a more diverse biota (Khaniya and Shrestha, 2020). The clearing of *L. camara* can also be beneficial in improving access to sites of both cultural and traditional significance (Mandiporeva et al., 2023). The continued eradication of *L. camara* is important to minimise its extensive dispersal by birds, which are key in the proliferation of the species due to the edible berries (Bitani and Downs, 2022).

The clearing of *L. camara* also opens up space for more productive land for crop cultivation and grazing (Barik et al., 2020; O'Connor and van Wilgen, 2020). This is important, especially in most rural communities in developing countries where the cultivation of land is an important livelihood (Mabhaudhi et al., 2018). This has implications for biodiversity, and efforts have been underway in some areas where *L. camara* has invaded to remove it, based on the declaration of *L. camara* by the International Union for Conservation for Nature (IUCN) as one of the worst invaders. For example, in countries like India, the removal of *L. camara* is an essential part of their protected area management (Rai and Singh, 2021).

In southern Africa, transboundary areas have been established over the past few years collaboratively to manage shared natural and cultural resources, regardless of international borders, to enhance the conservation of biodiversity and socio-economic development (Belaidi, 2019). One of the largest Transfrontier Conservation Areas (TFCAs) in southern Africa is the Kavango Zambezi Transfrontier Conservation Area (KAZA TFCA), which measures about 287 132 km^2 (Van de Meer et al., 2016). It spans five countries and has diverse plants and animals, which include IAPs (van Niekerk and Hildebrandt, 2014). In the KAZA TFCA, *L. camara* is one of the main IAPs found in several communities and protected areas. The existence of *L. camara* in the KAZA TFCA is a threat to biodiversity that requires attention. Restoration efforts, therefore, in developing countries offer an opportunity to reduce these biodiversity losses, and support sustainable development if done with consideration of local priorities (Banin et al., 2023).

Clearing IAPs may either be a success or a failure in contributing to native vegetation recovery depending on the context, extent of invasion, methods of clearing, and follow-up management.

There was limited success of passive restoration in South Africa, where no growth of native species was observed, due to a lack of seeds (Ruwanza et al., 2013). Also, in instances where degradation has been intensive, passive restoration alone may not be the best option as the ecosystems will require some form of management interventions (Holl and Aide, 2011). There is a need for proper implementation of measures to control IAPs to avoid their resurgence, hindering the recovery of native species (Holmes et al., 2020). Moreover, in most instances, there is usually limited post-clearance monitoring to assess ecological changes and whether clearing has been effective (Morris et al., 2008). These dynamics and the different outcomes of restoration efforts have resulted in an increasing focus to enhance knowledge of the impacts of clearing interventions to improve effectiveness. It is vital to understand site-specific responses following the clearing of IAPs (Flory and Clay, 2009). This study, therefore, assessed the impacts of passive restoration of the native woody species and soil properties degraded by *L. camara* invasion in the KAZA TFCA. The specific objectives were to ascertain the effects of clearing *L. camara* on (i) native woody species composition, and (ii) soil properties.

4.2. Methodology

4.2.1. Study site

Refer to the detailed description of the study site in section 1.6.

4.2.2. Data collection

4.2.2.1. Experimental design

In 2010, approximately four hectares were mechanically weeded (cleared) of *L. camara* in the Victoria Falls Rainforest, which is a protected area, and four hectares were cleared in the Ndlovu Communal area. This was facilitated by the Environmental Management Agency in association with Non-Governmental Organisations (NGOs) such as Environment Africa. After clearing, the areas were left to recover naturally. This provided an opportunity to establish whether clearing facilitated native vegetation recovery in both sites. To do this, sampling included the cleared areas, adjacent plots that remained invaded and, uninvaded control sites established approximately 50 m apart. The plot size used was 25 m x 25 m, with seven replicate plots per treatment (Figure 4.1).

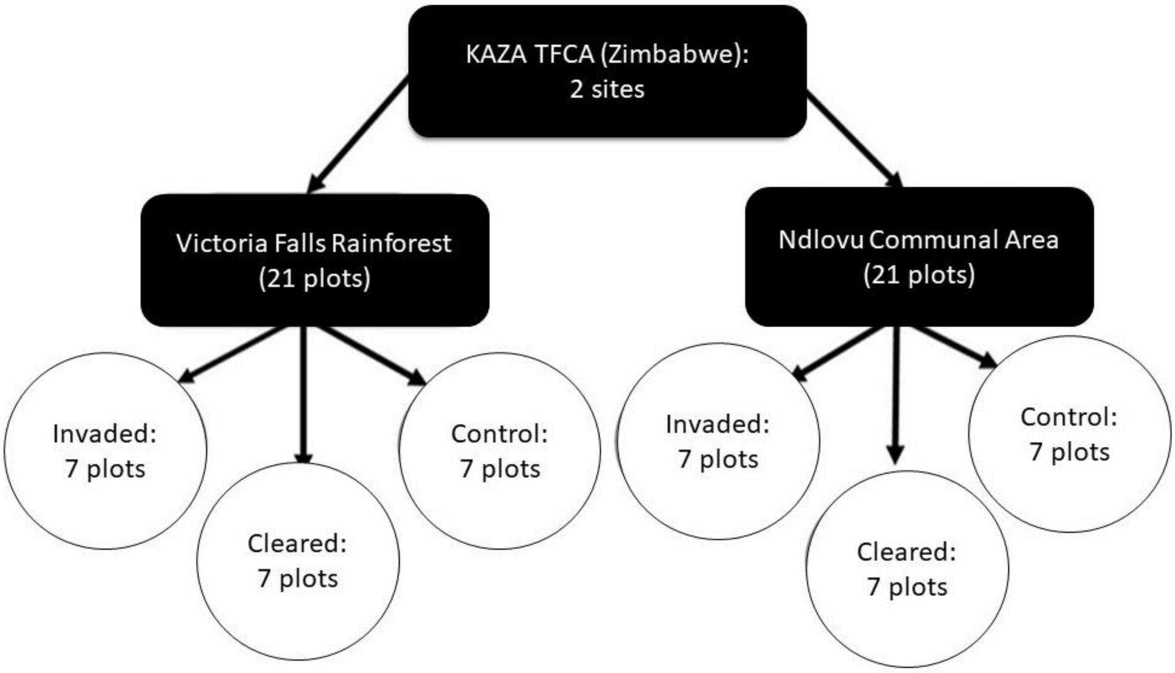

Figure 4.1: Experimental design of the 42 plots.

4.2.2.2. Field sampling

Field data collection was conducted from April to May 2018. Stratified random sampling was used to select the 21 plots for sampling, whereby the treatment in each area was considered as a stratum, and random plots were selected within it. Each plot sample was marked with a Global Positioning System (GPS) and several environmental attributes were recorded. Species identification of all native woody plants as well as individual stem counts were done, based on the definition of native woody species in Chapter 2. Similarly, vegetation attributes, namely height, diameter at breast height (DBH) and mean canopy diameter, were measured, also based on the definitions and descriptions in Chapter 2. Specifically, for mean canopy diameter, the longest diameter of the perpendicular was measured. This was coupled with soil sample collection at a depth of 10 to 15 cm, from four corners of the plot and one in the middle, and these were mixed to make a composite sample (Francis et al., 2018; Ruwanza and Shackleton, 2017). Five variables were analysed for each soil sample: soil pH; nitrate nitrogen (NO_3^--N); phosphorous (P); exchangeable potassium (K) and organic carbon (OC). The soil sample collection, preparation and analysis were consistent with the description in Chapter 2. A total of 36 soil samples were collected, representing 86% of the sites in each of the areas; 18 in the Communal area and 18 in the Rainforest area, all evenly split into six samples per treatment. Due to funding limitations, only 86% of soil samples (out of 42) were tested.

4.2.3. Data analysis

Stem density per hectare, species richness, diversity, mean stem diameter, height, canopy cover and species occupancy frequency were calculated for each treatment in the two study sites. The distribution of the data was reviewed graphically using boxplots and QQ plots to check for normal distribution. One-way analysis of variance (ANOVA) tests with Tukey's Honest Significant Difference test were conducted to assess the effect of clearing *L. camara* on these vegetation attributes, species diversity and soil variables. A nonmetric multidimensional scaling (NMDS), using Bray Curtis dissimilarity for presence-absence data and including highly abundant and rare taxa, was used to assess species composition variation based on the treatments. Statistical significance was determined at $p<0.05$ and all statistical computations were done using R version 4.1.3.

4.3. Results

4.3.1. Effects of clearing on native woody species

Cleared plots had significantly higher stem densities, diversity and species richness compared to the invaded plots in both the Communal and Rainforest areas (Table 4.1). Stem height on the other hand, cleared plots had similar height with the invaded plots in the Communal area, while in the Rainforest area, the invaded plots had a significantly higher stem height. The cleared plots also had a lower mean diameter in both the Communal area and the Rainforest. The canopy cover, however, despite having a higher mean value, was significantly similar in cleared plots and invaded plots (Table 4.1).

Forty-eight native woody species were identified across all the sites. In both the Communal area and the Rainforest, control plots had more species (30 and 33, respectively) compared to the invaded plots (20 and 15, respectively), which had the least number of species in both areas (Table 4.2). Overall, the highest frequency of occupancy was found in Uninvaded (Control) plots of both the Communal area and the Rainforest, which had 30 and 34 different species (Table 4.2).

Table 4.1: Comparison of native woody vegetation attributes and diversity indices in the *L. camara* invaded, cleared and uninvaded plots in the Victoria Falls Rainforest and Ndlovu Communal area. Data presented as mean ± standard deviation and one-way-ANOVA statistic test results. Different superscripts on each mean indicate significant differences at p<0.05.

Communal area					
Variables	**Invaded**	**Cleared**	**Control**	**F- value**	**P-value**
Density/ha	$208{\pm}77^{a}$	$468{\pm}191^{b}$	$644{\pm}201^{c}$	12.26	<0.0001
Shannon-Wiener Index	$0.9{\pm}0.4^{a}$	$2.1{\pm}0.3^{b}$	$2.4{\pm}0.3^{b}$	42.8	<0.0001
Species richness	$4.9{\pm}2.3^{a}$	$11.9{\pm}2.3^{b}$	$18.3{\pm}4.6^{c}$	27.9	<0.0001
Stem height (m)	$5.4{\pm}0.8^{a}$	$5.1{\pm}1.2^{a}$	$8.2{\pm}0.5^{b}$	23.4	<0.0001
Stem diameter (m)	$0.4{\pm}0.1^{b}$	$0.3{\pm}0.1^{a}$	$0.6{\pm}0.1^{c}$	48.1	<0.0001
Canopy cover (%)	$5.4{\pm}1.4^{a}$	$16.1{\pm}3.4^{a}$	$57.9{\pm}15.4^{b}$	64.1	<0.0001
Rainforest area					
Variables	**Invaded**	**Cleared**	**Control**	**F- value**	**P-value**
Density/ha	$180{\pm}85^{a}$	$493{\pm}238^{b}$	$752{\pm}187^{c}$	17.4	<0.0001
Shannon-Wiener Index	$1.3{\pm}0.6^{a}$	$2.0{\pm}0.2^{b}$	$2.5{\pm}0.3^{b}$	16.6	<0.0001
Species richness	$4.3{\pm}1.5^{a}$	$10.3{\pm}2.2^{b}$	$16.1{\pm}1.9^{c}$	29.8	<0.0001
Stem height (m)	$8.5{\pm}1.0^{b}$	$3.5{\pm}1.0^{a}$	$9.9{\pm}1.7^{c}$	48.9	<0.0001
Stem diameter (m)	$0.4{\pm}0.1^{b}$	$0.3{\pm}0.1^{a}$	$0.7{\pm}1.7^{c}$	91.6	<0.0001
Canopy cover (%)	$8.6{\pm}1.7^{a}$	$14{\pm}3.2^{a}$	$64.9{\pm}0.1^{b}$	291.6	<0.0001

Table 4.2: Relative frequency of occupancy (%) species at the various sites

Plant names	**Communal area**			**Rainforest area**		
	Invaded	**Cleared**	**Control**	**Invaded**	**Cleared**	**Control**
Adansonia digitata	29	43	57	14	0	43
Afzelia quanzesis	0	0	57	0	0	57
Antidesma venosum	0	57	57	29	14	29
Azanza garckeana	0	43	71	29	29	29
Baikiaea plurijuga	0	0	57	0	0	0
Bauhinia petersiana	0	57	43	14	0	0
Berchemia discolor	0	29	43	0	29	29
Brachystegia spiciformis	14	29	0	0	0	0
Bridelis mollis	14	0	43	14	0	0

Plant names	Communal area			Rainforest area		
	Invaded	Cleared	Control	Invaded	Cleared	Control
Burkea africana	0	0	0	0	14	14
Colophospermum mopane	0	14	0	0	29	14
Combretum apiculatum	0	0	0	14	14	0
Combretum collinum	0	0	71	0	0	0
Combretum hereroense	43	14	0	29	29	43
Combretum imberbe	71	71	43	29	29	100
Combretum molle	0	0	71	0	29	43
Dichrostachys cinerea	0	57	0	14	100	43
Diospyros mespiliformis	57	43	0	0	29	57
Diospytos quiloensis	57	0	0	0	0	29
Diplorhynchus condylocarpon	0	0	14	0	0	0
Euclea divinorum	29	71	0	0	29	0
Ficus petersii	0	0	0	0	29	0
Fluggea virosa	0	14	14	0	43	14
Grewia flavescens	57	14	0	29	29	29
Gymosporia senegalensis	43	0	0	14	0	14
Hyphaene coriacea	0	0	57	0	0	29
Kigelia africana	0	0	0	0	29	0
Kirkia acuminata	0	57	57	0	0	14
Manilkara mochisia	29	0	100	14	29	57
Peltophorum africanum	14	14	43	0	43	29
Philenoptera violacea	0	71	29	0	29	29
Phoenix reclinata	0	57	0	0	57	0
Piliostigma thonningii	0	43	0	14	0	0
Pseudolachnostylis maprouneifolia	0	43	43	0	14	14
Pterocarpus lucens	0	43	57	0	43	43
Schinziophyton rautamenii	0	0	57	0	0	0
Sclerocarya birrea	0	0	43	0	29	86
Senegalia galpinii	0	43	71	0	0	57
Senegalia nigrescens	0	14	57	14	29	86

Plant names	Communal area			Rainforest area		
	Invaded	Cleared	Control	Invaded	Cleared	Control
Strychnos potatorum	29	14	71	29	43	100
Terminalia sericea	43	29	71	43	29	71
Terminalia stuhlmannii	0	29	86	29	43	100
Trichilia emetica	14	0	57	0	0	43
Vachellia erioloba	0	14	43	0	29	57
Vachellia karroo	0	0	0	14	0	57
Vangueria infausta	0	0	0	14	14	0
Ximenia caffra	0	0	0	0	14	0
Ziziphus mucronata	0	14	57	14	29	29

The NMDS ordination plot had a stress value of 0.09, and was considered a good fit to the original distances and the projected distances of the ordination diagram (Fig. 4.2).

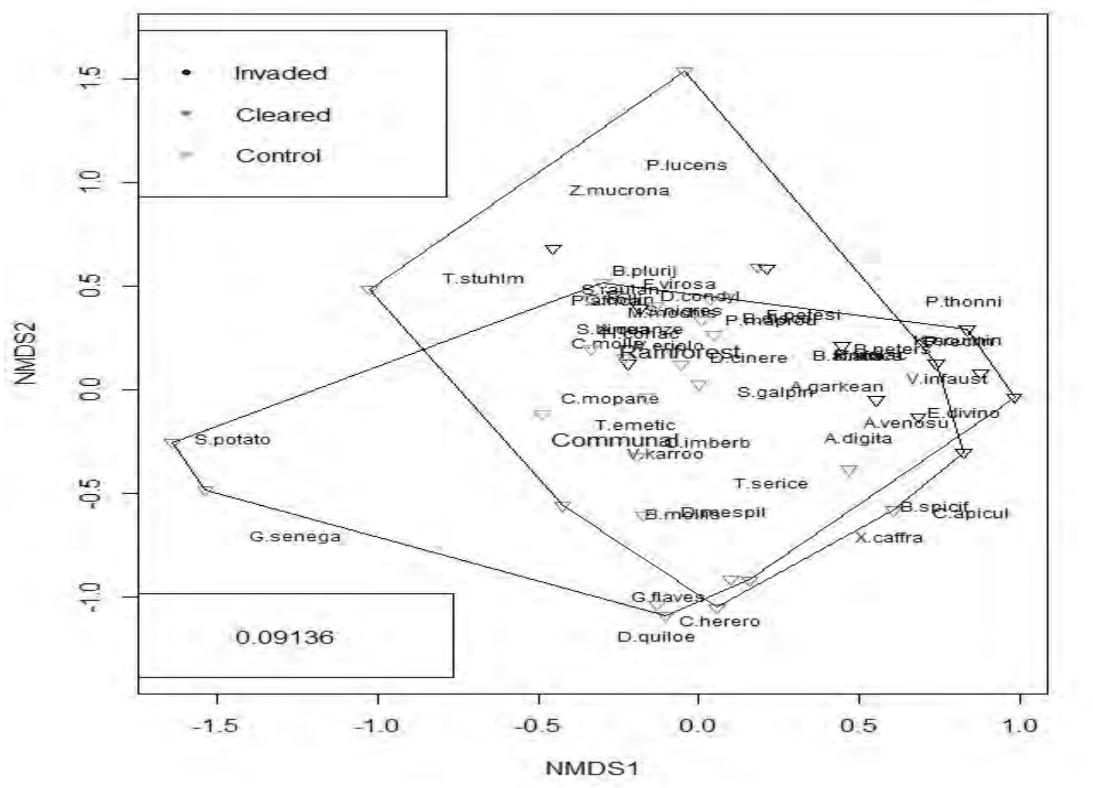

Figure 4.2: NMDS plot showing the aggregation of species in the Park and Communal area based on the three treatments (Invaded, Cleared and Control)

Clear clustering of some species was observed for those found in cleared and uninvaded plots of both the Park and Communal area (Figure 4.2). Species such as *Strychnos potatorum*, *Gymnosporia senegalensis*, *Grewia flavescens, Combretum hereroense, Ximenia caffra, Ziziphus mucronata, Pterocarpus lucens, Terminalia stuhlmanii* and *Diospyros quiloensis* expressed dissimilarity in species composition in the Rainforest and Communal area, compared with other species. These species that expressed dissimilarity with the rest were mainly found in the cleared plots (Figure 4.2). A distinct cluster of species composed of those found in uninvaded areas both in the Rainforest and Communal area is shown in Figure 4.3.

Further analysis using a cluster dendrogram and clustering tree to express the dissimilarity matrix showed four distinct clusters. Two clusters (Cluster A and Cluster D), shown in Figure 4.3, displayed the invaded plots in the Communal area and the Park area to be similar. Cluster A also consisted of most of the cleared plots. The uninvaded clusters are distinctly aggregated in clusters B and C for both the Communal and Park areas, with a few cleared plots in the Park and Communal area also found in these clusters.

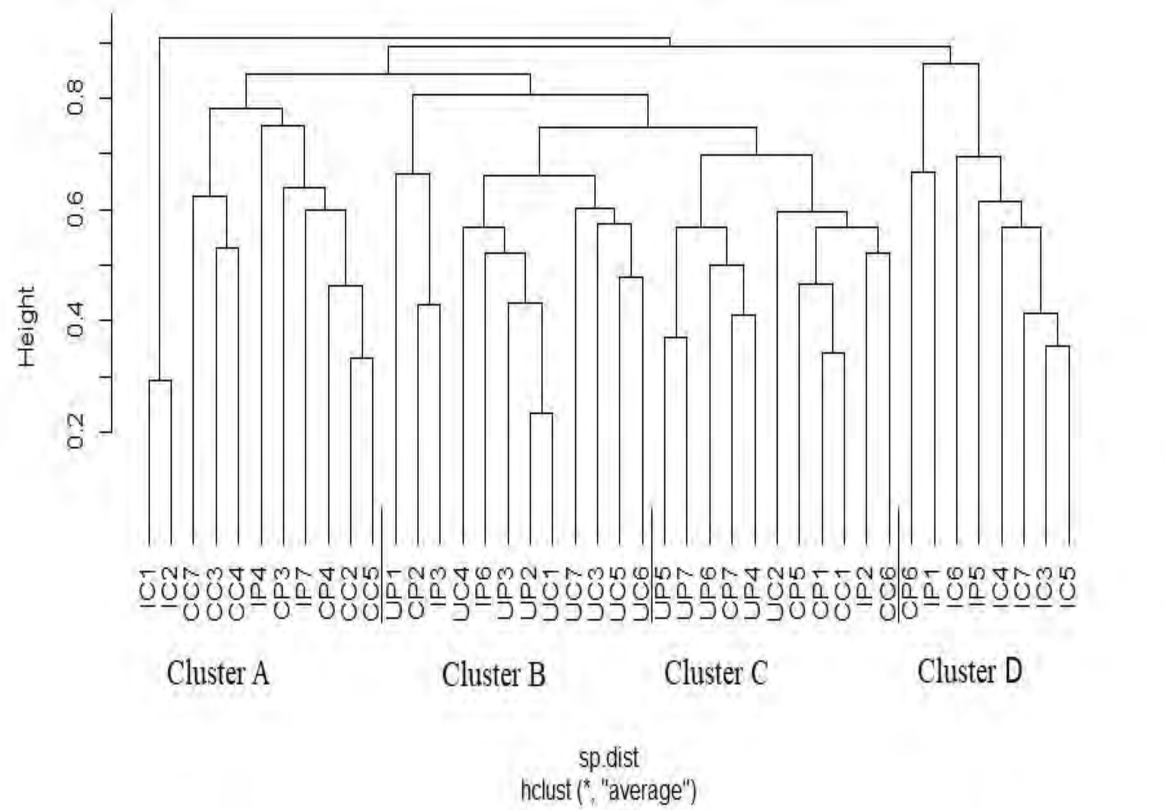

Figure 4.3: Cluster analysis of the woody species in the Communal area and Rainforest Park area in the three treatments (Invaded, Cleared, and Uninvaded). The acronyms represent the following: IC- Invaded Communal ; IP - Invaded; CC – Cleared Communal; CP – Cleared Park; UC – Uninvaded Communal; UP – Uninvaded Park

4.3.2. Effect of clearing *L. camara* on soil properties

The mean soil pH range found in the Communal and Park areas was 6.0 to 7.2, with the pH of the invaded area being 6.0 to 6.5, the cleared area 6.4 to 6.6, and the control 7.0 to 7.2. Significant differences in pH were observed between the invaded and control plots in the Rainforest, whereas no significant differences were found in the pH of invaded and cleared plots in both the Communal area and Rainforest (Table 4.3). The ANOVA results showed no significant differences in other soil variables among the cleared, invaded and control plots in both the Communal and Park areas (Table 4.3). Although the soil measures showed no significant difference for nearly all soil measures, the values for the cleared plots were intermediated between invaded and control.

Table 4.3: Comparison of soil variables in the *L. camara* cleared, invaded and control plots in the Victoria Falls National Park and Ndlovu Communal area. Data presented as mean ± standard deviation and one-way-ANOVA statistic test results. Different superscripts on the means indicate significant differences at p<0.05.

Communal area					
	Invaded	**Cleared**	**Control**	**F-value**	**P-value**
pH	6.5 ± 0.9^{a}	6.6 ± 0.4^{a}	7.2 ± 0.5^{b}	7.5	0.006**
NO_3^--N (mg/kg)	7.6±11.1	1.1±1.5	14.7±15.8	0.9	0.43
P (mg/kg)	7.4±8.8	2.0±3.5	0.01±0.10	2.6	0.11
Exch. K (mg/kg)	1 426±1109	758±231	582±82	1.1	0.36
OC (%)	2.1±0.9	1.2±1.1	0.7±0.7	1.7	0.20
Rainforest area					
	Invaded	**Cleared**	**Control**	**F-value**	**P-value**
pH	6.0±0.1	6.4±0.4	7.0±0.8	2.1	0.16
NO_3^--N (mg/kg)	3.6±6.4	4.4±3.4	11.3±12.1	2.2	0.14
P (mg/kg)	6.2±9.5	0.01±0.10	0.01±0.10	2.9	0.08
Exch. K (mg/kg)	783±101	710±337	725±176	1.4	0.28
OC (%)	2.6±0.4	3.4±0.8	2.2±0.6	3.4	0.06

4.4. Discussion

There were considerable changes in most of the native woody vegetation attributes after the eight-year recovery period. The density of stems per hectare, species richness and species diversity showed a positive response with a considerable increase in the cleared plots compared to invaded plots. Changes in soil properties were also observed in plots where *L. camara* was removed compared to the invaded plots. Re-emergence of *L. camara* was observed in some previously cleared plots in the Communal area indicating the need for further management or control strategies. These results indicate that the removal of *L. camara* has a positive influence on the native woody species, encouraging growth as well as diversity, and impacting the soil properties. Moreover, the study shows the success of passive restoration in the Hwange District in the KAZA TFCA but highlights the need for further management interventions, especially noting the re-emergence observed in the Communal area plots.

4.4.1. Passive restoration

Recovery of landscapes after restoration efforts is mainly assessed between 5 to 10 years after removal (Meli et al., 2017) and this study site offered an ideal environment for assessment based on the 8-year recovery period. The results show the effectiveness of passive restoration in restoring woody species with the evident recovery in species richness, diversity, density and canopy cover. The choice of this method for this study was based on the already cleared plots in the study area and the need to follow up on the effects. The advantages of passive restoration are that it is less demanding to implement and comes at a lower cost (Prach et al. 2020). However, several limitations exist which are not ideal for an area that has been subjected to intense damage and has limited capacity to recover (Weidlich et al. 2020).

Since part of the study site was the Rainforest area, which is a tourist attraction, there was constant biological monitoring to note the changes, and a need to keep the area as intact as possible by the Zimbabwe Parks and Wildlife Authority, Environmental Management Agency and non-profit organisations such as Environment Africa. The opportunity to assess the same dynamics in a communal setting where there is human disturbance was also compelling, with Holl and Aide (2011) highlighting the complexities of using passive restoration in areas where there are continued disturbances. This study showed there was recovery in native woody vegetation in both the Rainforest and Communal areas despite the fact that the Rainforest is consistently managed as compared to the Communal area.

4.4.2. Response of woody vegetation attributes to clearing

The number of stems per hectare in the cleared plots were significantly higher than those in invaded areas, and the same trend was observed for diversity indices and richness as compared to the invaded plots. This trend was similar in both the Rainforest and the Communal areas. In line with this, Babu et al. (2009) showed how beneficial eradication of *L. camara* was in restoring grasslands in India, where there was evidence of improved native biodiversity and enhanced biological productivity. The number of stems per hectare was also observed to be slightly higher in the cleared Rainforest area as compared with the Communal area, and this may be linked to the different microenvironments in these two sites. The Rainforest, which is considerably wetter than the Communal area, would encourage the growth of more stems as compared to the Communal area. This is in line with Prasad et al. (2018), who showed that wetter forests have a greater recovery of native trees as compared with the drier forests. Morris et al. (2008) also showed that, after the clearing of IAPs, there was an increase in native vegetation densities in areas that have limited human disturbance; this also showed recovery with only passive restoration.

In this study, despite the Communal area having more human disturbance than the Rainforest area, the Communal area also exhibited improved stem density, diversity and richness after passive restoration, as happened in the Rainforest area. Several studies showed that it is not always the case that there will be the recovery of native woody vegetation after passive restoration (Reid et al., 2009). The success of the restoration effort is mainly dependent on the extent of degradation as well as the level of post-clearing disturbance (Prach and Hobbs, 2008). The recovery may also be influenced by factors such as proximity to a natural forest which can act as a seed bank (Shiferaw et al., 2018). This implies that, although the study sites were invaded by *L. camara*, the infestation may have been at a level in which recovery was supported by natural regeneration. In some instances, the damage to the ecosystem depletes the seed banks so that, despite the removal of IAPs, there is limited to no natural regrowth of native plants (Ruwanza et al., 2013). In the case of this study, the increase in species diversity indicates the disturbance had possibly been moderate, so seed banks were still viable enough to allow recovery, based on the close similarities of diversity and richness of cleared plots and the control plots in both the Communal area and the Rainforest area.

Stem height and stem diameter of the native vegetation was lower in cleared plots than in the invaded and uninvaded plots. This may be because the additional new vegetation, that had resprouted, increased the stem density. These woody species that have grown due to the clearing are still younger than eight years and have lower stem height and DBH, and, averaged together with the older and more mature trees, they significantly reduce the mean height and DBH. The canopy cover in the cleared plots had now increased and become significantly higher than the invaded plots. As expected, the control plots had the highest canopy cover and this aligns with assertions by Chen et al. (2022) that forests with a tight canopy may resist the invasion of some species, as the light conditions are reduced below the canopy, limiting the growth of the seedlings of the IAPs. The improvement in the canopy cover after clearing is a good indication of recovery and the forest building resistance to future invasion. This is because *L. camara* proliferates mostly in open areas due to the high production of seed and progress of invasion, as compared with areas with dense canopies (Carrión-Tacuri et al., 2011).

4.4.3. Response of soil properties to clearing of *L. camara*

Passive restoration in the KAZA TFCA increased the pH in the Rainforest area (Table 4.3). No changes were observed in the Communal area. No significant differences were observed in nitrate nitrogen, phosphorus, exchangeable potassium and organic carbon. However, based on the intermediate soil values in all cleared sites in Figure 4.3, this showed there were some changes, but not sufficiently advanced to show statistical significance. This indicates that there was some recovery due to clearing which had taken place eight years earlier, and recovery still had a long way to go for it to be significant. This is in line with other studies that have shown differences in soil variables between invaded and cleared sites. In some instances, such as the study by Moore (2019), removal of IAPs was associated with reduction in nitrates. Moore (2019) also described how in some studies *L. camara* has been associated with high nitrogen content, implying that removal of the *L. camara* should lower nitrate levels. In fact, the mean values of nitrates in this study were lowest in cleared plots, and the lack of significance may be due to the low sample size which was limited by the availability of resources; hence the findings may be taken with a bit of caution. Additionally, the results may also imply that there are other drivers of soil variables besides removal of IAPs. This is also evident in the lack of differences with the control plot, as all sites have different vegetation dynamics.

4.5. Conclusions

This study showed a positive trajectory of recovery based on passive restoration in the KAZA TFCA, indicative that removal of *L. camara* allows growth of native woody vegetation. This study has implications for different parts of the KAZA TFCA and other areas with a similar context, by showing that native woody vegetation can recover naturally after *L. camara* has been removed. This is an important aspect considering the drive to halt and reduce ecosystem degradation, with the United Nations declaring 2021-2030 as a decade of ecosystem restoration (Waltham et al., 2020). Results show the success of passive restoration in both a protected area and an unprotected area, the Rainforest and the Communal area in this case, thus showing that this method can be implemented in different settings. The mechanical removal of *L. camara* in both instances was found to be a viable option for recovery in the KAZA TFCA. The results from the soil properties showed responses that did not yield statistical difference but did show visible changes due to passive restoration, in line with some studies. This may be due to the number of years (8-year recovery period) hence may not necessarily reflect the full response of soil properties to the clearing of *L. camara*, and may reflect the need for further studies after more years and more sampling effort, and for focusing on other dimensions.

References

Aronson, J., Goodwin, N., Orlando, L., Eisenberg, C. and Cross, A.T., 2020. A world of possibilities: Six restoration strategies to support the United Nation's Decade on Ecosystem Restoration. *Restoration Ecology*, 28(4), 730-736.

Babu, S., Love, A. and Babu, C.R., 2009. Ecological restoration of *Lantana*-invaded landscapes in Corbett Tiger Reserve, India. *Ecological Restoration*, 27(4), 467-477.

Bajwa, A.A., Farooq, M., Nawaz, A., Yadav, L., Chauhan, B.S. and Adkins, S., 2019. Impact of invasive plant species on the livelihoods of farming households: Evidence from *Parthenium hysterophorus* invasion in rural Punjab, Pakistan. *Biological Invasions*, 21(11), 3285-3304.

Balintulo, P., 2021. Soil and vegetation recovery following *Acacia dealbata* clearing in the Tsitsa catchment, Eastern Cape Province of South Africa: Implications for ecological restoration. Master's Dissertation: Rhodes University, South Africa.

Banin, L.F., Raine, E.H., Rowland, L.M., Chazdon, R.L., Smith, S.W., Rahman, N.E.B., Butler, A., Philipson, C., Applegate, G.G., Axelsson, E.P. and Budiharta, S., 2023. The road to recovery: A synthesis of outcomes from ecosystem restoration in tropical and sub-tropical Asian forests. *Philosophical Transactions of the Royal Society B*, 378(1867), 20210090.

Barik, S.S., Sahoo, R.P. and Barik, S.S., 2020. *Lantana camara* L.: An emerging threat to native flora and livestock: A review. *Journal of Pharmacognosy and Phytochemistry*, 9(5), 2363-2366.

Belaidi, N., 2019. Socio-ecological coviability confronted with the neoliberal system: The Peace Parks experience (Southern Africa). In O. Barrière, M. Behnassi, G. David, V. Douzal, M. Fargette, T. Libourel, M. Loireau, L. Pascal, C. Prost, V. Ravena-Cañete, F. Seyler and S. Morand (eds), *Coviability of social and ecological systems: Reconnecting mankind to the biosphere in an era of global change: Volume. 1: The foundations of a new paradigm*, 369-385. Springer, Cham.

Bitani, N. and Downs, C.T., 2022. The native avian species visiting fruiting invasive Lantana camara in KwaZulu-Natal, South Africa: Potential dispersal concerns. *South African Journal of Botany*, 148, 573-579.

Blanchard, R. and Holmes, P.M., 2008. Riparian vegetation recovery after invasive alien tree clearance in the Fynbos Biome. *South African Journal of Botany*, 74(3), 421-431.

Brancalion, P.H.S., Campoe, O., Mendes, J.C.T., Noel, C., Moreira, G.G., van Melis, J., Stape, J.L. and Guillemot, J., 2019. Intensive silviculture enhances biomass accumulation and tree diversity recovery in tropical forest restoration. *Ecological Applications,* 29(2), 01847.

Carrión-Tacuri, J., Rubio-Casal, A.E., Cires, A., Figueroa, E. and Castillo, J., 2011. *Lantana camara* L.: A weed with great light-acclimation capacity. *Photosynthetica*, 49(30), 321-329.

Chen, Xi, Chen, Xin, Huang, S. and Fang, D., 2022. Impacts of Moso bamboo (*Phyllostachys pubescens*) invasion on species diversity and aboveground biomass of secondary coniferous and broad-leaved mixed forest. *Frontiers in Plant Science*, 13, 1001785.

D'Antonio, C.M., August-Schmidt, E. and Fernandez-Going, B., 2016. Invasive species and restoration challenges. In M.A. Palmer, J.B. Zedler and D.A. Falk (eds), *Foundations of restoration ecology*, 216-244. Springer.

DiTomaso, J.M., Monaco, T.A., James, J.J. and Firn, J., 2017. Invasive plant species and novel rangeland systems. In D.D. Briske (ed.), *Rangeland systems,* 429-465. Springer International Publishing A.G., Switzerland.

Flory, S.L. and Clay, K., 2009. Invasive plant removal method determines native plant community responses. *Journal of Applied Ecology,* 46(2), 434-442.

Francis, B., Pepler, D. and Ndagurwa, H.G., 2018. Woody species diversity and surface soil nutrients in an *Opuntia fulgida Engelm* invaded semi-arid savanna rangeland of south-west Zimbabwe. In S. Mutanga, T. Simelane, T. Gumbo and M. Mujuru (eds), *Africa at a Crossroads: The continent's future 50 years after the formation of the organization of African unity/African union*, 8-23. Africa Institute of South Africa, Pretoria.

Froeschlin, N., Privett, S.D.J., Richardson, D.M. and Gaertner, M., 2022. Fynbos vegetation recovery twelve years after removal of invasive Eucalyptus trees. *South African Journal of Botany*, 147, 764-773.

Harvey-Samuel, T., Ant, T. and Alphey, L., 2017. Towards the genetic control of invasive species. *Biological Invasions*, 19(6), 1683-1703.

Hoare, D., 2021. Alien and Invasive Plant Species Management Plan. *Cell*, 83(284), 5111.

Holl, K.D. and Aide, T.M., 2011. When and where to actively restore ecosystems? *Forest Ecology Management*, 261(10), 1558-1563.

Holmes, P.M., Esler, K.J., Van, W.B.W. and Richardson, D.M., 2020. Ecological restoration of ecosystems degraded by invasive alien plants in South African Fynbos: Is spontaneous succession a viable strategy? *Transactions of the Royal Society of South Africa,* 75(2), 111-139.

Hussner, A., Stiers, I., Verhofstad, M.J.J.M., Bakker, E.S., Grutters, B.M.C., Haury, J., Van Valkenburg, J.L.C.H., Brundu, G., Newman, J., Clayton, J.S. and Anderson, L.W.J., 2017. Management and control methods of invasive alien freshwater aquatic plants: a review. *Aquatic Botany*, 136, 112-137.

Kerr, T.F. and Ruwanza, S., 2016. Does *Eucalyptus grandis* invasion and removal affect soils and vegetation in the Eastern Cape Province, South Africa? *Austral Ecology,* 41(3), 328-338.

Khaniya, L. and Shrestha, B.B., 2020. Forest regrowth reduces richness and abundance of invasive alien plant species in community managed *Shorea robusta* forests of central Nepal. *Journal of Ecology and Environment*, 44(1), 1-8.

Kumar, M., Kumar, S., Verma, A.K., Joshi, R.K. and Garkoti, S.C., 2021. Invasion of *Lantana camara* and *Ageratina adenophora* alters the soil physico-chemical characteristics and microbial biomass of chir pine forests in the central Himalaya, India. *Catena*, 207, 105624.

Lamsal, P., Kumar, L., Aryal, A. and Atreya, K., 2018. Invasive alien plant species dynamics in the Himalayan region under climate change. *Ambio*, 47(6), 697-710.

Latif, S., Chiapusio, G. and Weston, L.A., 2017. Allelopathy and the role of allelochemicals in plant defence. In G. Becard (ed.), *Advances in botanical research*, 82, 19-54. Academic Press.

Mabhaudhi, T., Chibarabada, T.P., Chimonyo, V.G.P., Murugani, V.G., Pereira, L.M., Sobratee, N., Govender, L., Slotow, R. and Modi, A.T., 2018. Mainstreaming underutilized indigenous and traditional crops into food systems: A South African perspective. *Sustainability*, 11(1), 172.

Mack, R.N., 1981. Invasion of *Bromus tectorum* L. into Western North America: An ecological chronicle. *Agro-ecosystems*, 7(2), 145-165.

Mandiporeva, J., Gamundani, T.P., Gurajena, B., Mpakairi, K. and Magqina, T. 2023. Spatial distribution and effects of *Lantana camara* invasion on soil properties and native vegetation community composition in Tugwi-Mukosi Recreational Park, Masvingo, Zimbabwe. *African Journal of Ecology*, 61(1), 185-203.

Martin, D.M., 2017. Ecological restoration should be redefined for the twenty-first century. *Restoraton Ecology*, 25(5), 668-673.

McIver, J. and Starr, L., 2001. Restoration of degraded lands in the interior Columbia River basin: Passive vs. active approaches. *Forest Ecology and Management*, 153(1-3), 15-28.

Meli, P., Holl, K.D., Benayas, J.M.R., Jones, H.P., Jones, P.C., Montoya, D. and Mateos, D.M., 2017. A global review of past land use, climate, and active vs. passive restoration effects on forest recovery. *PLoS One*, 12(2), 0171368.

Mhlongo, E.S., 2021. *Perceptions of the role of Lantana camara on human well-being and rural livelihoods in Vhembe Biosphere Reserve, South Africa*. Doctoral Thesis: Rhodes University, South Africa.

Mollot, G., Pantel, J.H. and Romanuk, T.N., 2017. The effects of invasive species on the decline in species richness: A global meta-analysis. In D.A. Bohan, A.J. Dumbrell and F. Massol (eds), *Advances in ecological research*, 56, 61-83. Academic Press.

Moore, E.R., Trammell, T.L.E. and D'Amico, V. 2019. Evaluating the success of ecosystem restoration via changes to plant community composition and soil-N cycling in urban forests invaded by *Rosa multiflora*. ESA Annual Meeting, 11-16 August 2019, Louisville, United States of America. Ecological Society of America.

Morris, T., Witkowski, E. and Coetzee, J., 2008. Initial response of riparian plant community structure to clearing of invasive alien plants in Kruger National Park, South Africa. *South African Journal of Botany,* 74(3), 485-494.

Mostert, E., Gaertner, M., Holmes, P.M., O’Farrell, P.J. and Richardson, D.M., 2018. A multi-criterion approach for prioritizing areas in urban ecosystems for active restoration following invasive plant control. *Environmental Management*, 62(6), 1150-1167.

Ndou, E. and Ruwanza, S., 2016. Soil and vegetation recovery following alien tree clearing in the Eastern Cape Province of South Africa. *African Journal of Ecology*, 54(4), 460-470.

O’Connor, T.G. and van Wilgen, B.W., 2020. The impact of invasive alien plants on rangelands in South Africa. *Biological Invasions in South Africa*, 14, 459-487.

Prach, K. and Hobbs, R.J., 2008. Spontaneous Succession versus technical reclamation in the restoration of disturbed sites. *Restoration Ecology*, 16(3), 363-366.

Prach, K., Šebelíková, L., Řehounková, K. and del Moral, R., 2020. Possibilities and limitations of passive restoration of heavily disturbed sites. *Landscape Research*, 45(2), 247-253.

Prasad, A., Ratnam, J. and Sankaran, M., 2018. Rainfall and removal method influence eradication success for *Lantana camara*. *Biological Invasions*, 20(12), 3399-3407.

Prior, K.M., Adams, D.C., Klepzig, K.D. and Hulcr, J., 2018. When does invasive species removal lead to ecological recovery? Implications for management success. *Biological Invasions*, 20(2), 267-283.

Pyšek, P., Hulme, P.E., Simberloff, D., Bacher, S., Blackburn, T.M., Carlton, J.T., Dawson, W., Essl, F., Foxcroft, L.C., Genovesi, P. and Jeschke, J.M., 2020. Scientists' warning on invasive alien species. *Biological Reviews*, 95(6), 1511-1534.

Rai, P.K. and Singh, J.S. 2021. Plant invasion in protected areas, the Indian Himalayan region, and the North East India: Progress and prospects. *Proceedings of the Indian National Science Academy*, 87, 19–35.

Ravhuhali, K.E., Mudau, H.S., Moyo, B., Hawu, O. and Msiza, N.H., 2021. *Prosopis* species—An invasive species and a potential source of browse for livestock in semi-arid areas of South Africa. *Sustainability*, 13(13), 7369.

Reid, A.M., Morin, L., Downey, P.O., French, K. and Virtue, J.G., 2009. Does invasive plant management aid the restoration of natural ecosystems? *Biological Conservation*, 142(10), 2342-2349.

Ruwanza, S. and Dondofema, F., 2020. Effects of exotic guava (*Psidium guajava* L.) invasion on soil properties in Limpopo, South Africa. *African Journal of Ecology*, 58(2), 272-280.

Ruwanza, S., Gaertner, M., Esler, K.J. and Richardson, D.M., 2013. The effectiveness of active and passive restoration on recovery of indigenous vegetation in riparian zones in the Western Cape, South Africa: A preliminary assessment. *South African Journal of Botany*, 88, 132-141.

Ruwanza, S. and Shackleton, C.M., 2016. Effects of the invasive shrub, *Lantana camara*, on soil properties in the Eastern Cape, South Africa. *Weed Biology Management*, 16(2), 67-79.

Ruwanza, S. and Shackleton, C.M., 2017. Ecosystem-scale impacts of non-timber forest product harvesting: Effects on soil nutrients. *Journal of Applied Ecology*, 54(5), 1515-1525.

Ruwanza, S. and Tshillilo, K., 2019. Short term soil and vegetation recovery after *Acacia mearnsii* removal in Vhembe Biosphere Reserve, South Africa. *Applied Ecology and Environmental Research*, 17(2), 1705-1716.

Sher, A.A., El Waer, H., González, E., Anderson, R., Henry, A.L., Biedron, R. and Yue, P., 2018. Native species recovery after reduction of an invasive tree by biological control with and without active removal. *Ecological Engineering*, 111, 167-175.

Shiferaw, W., Demissew, S. and Bekele, T., 2018. Ecology of soil seed banks: Implications for conservation and restoration of natural vegetation: A review. *International Journal of Biodiversity and Conservation,* 10(10), 380-393.

Shono, K., Cadaweng, E.A. and Durst, P.B., 2007. Application of assisted natural regeneration to restore degraded tropical forestlands. *Restoration Ecology*, 15(4), 620-626.

Souza-Alonso, P., Rodríguez, J., González, L. and Lorenzo, P., 2017. Here to stay. Recent advances and perspectives about Acacia invasion in Mediterranean areas. *Annals of Forest Science*, 74(3), 1-20.

Van der Meer, E., Badza, M.N. and Ndhlovu, A., 2016. Large carnivores as tourism flagship species for the Zimbabwe component of the Kavango Zambezi Transfrontier Conservation Area. *African Journal of Wildlife Research*, 46(2), 121-134.

van Niekerk, D. and Hildebrandt, L., 2014. Transboundary protection of biodiversity in the context of human and environmental security and climate change. In L.J. Kotzé and T. Marauhn (eds), *Transboundary governance of biodiversity,* 342-367. Brill Nijhoff.

Waltham, N.J., Elliott, M., Lee, S.Y., Lovelock, C., Duarte, C.M., Buelow, C., Simenstad, C., Nagelkerken, I., Claassens, L., Wen, C.K., Barletta, M., Connolly, R.M., Gillies, C., Mitsch, W.J., Ogburn, M.B., Purandare, J., Possingham, H. and Sheaves, M., 2020. UN Decade on Ecosystem Restoration 2021-2030—What chance for success in restoring coastal ecosystems? *Frontiers in Marine Science*, 7, 71.

Weidlich, E.W., Flórido, F.G., Sorrini, T.B. and Brancalion, P.H., 2020. Controlling invasive plant species in ecological restoration: A global review. *Journal of Applied Ecology*, 57(9), 1806-1817.

Zachariades, C., Paterson, I.D., Strathie, L.W., Van Wilgen, B.W. and Hill, M.P., 2017. Assessing the status of biological control as a management tool for suppression of invasive alien plants in South Africa. *Bothalia-African Biodiversity & Conservation*, 47(2), 1-19.

Zahawi, R.A., Reid, J.L. and Holl, K.D., 2014. Hidden costs of passive restoration. *Restoration Ecology*, 22(3), 284-287.

Zimmerman, C.L., Shirer, R.R. and Corbin, J.D., 2018. Native plant recovery following three years of common reed (*Phragmites australis*) control. *Invasive Plant Science and Management*, 11(4), 175-180.

Zozo, E., 2022. The use of the cochineal insect, *Dactylopius tomentosus* Lamarck, as a biological control agent for the invasive alien thistle cholla, *Cylindropuntia pallida* (Rose) FM Knuth in South Africa. Master's Dissertation: Rhodes University, South Africa.

Chapter 5: Stakeholders' knowledge of the management and policies pertaining to the *Lantana camara* invasion in the Kavango Zambezi Transfrontier Conservation Area

Abstract

The Kavango Zambezi Transfrontier Conservation Area (KAZA TFCA) is biodiversity-rich. However, this area has been affected by invasive alien species (IAS), such as *Lantana camara*, whose negative ecological impacts have prompted control and management in several areas. Eradication of IAS usually adopts a top-down approach by policymakers, with those directly affected excluded from these decisions, and, at times, unaware of the management goals and policies surrounding the IAS. In other cases, community members may be unaware that the species is invasive and instead consider it naturalised. The study involved thematic analysis of policy documents in Zimbabwe to assess the basis for IAS management. The study also investigated stakeholders' knowledge of the management and governing around the *L. camara* invasion. In this regard, 300 household questionnaires were administered, and 11 key informant interviews were conducted. Findings show that there are no effective policies and legislation in place for KAZA TFCA specifically. In Zimbabwe, general legal instruments about IAS exist but are silent on funding, which results in a lack of implementation and awareness. The policy documents resonate with international treaties and conventions to which Zimbabwe is a signatory. The majority of the respondents (90%) were unaware of the policies that prescribe the management of IAS, including *L. camara*, and 93% believed that authorities should manage it. Some of the key informants, whose direct focus is on protecting natural resources, felt that the management of IAS is everyone's responsibility as stipulated by the legislation, while others believed that management should be optional and called for a review of the legislation, whereby empirical studies on *L. camara* should back the policy. The study recommends a holistic approach to transforming IAS management structures and processes, whereby laws and policies are used to foster compliance, and stakeholders know their duties and responsibilities in IAS management; this approach could pave the way for developing effective management strategies for *L. camara* in the KAZA TFCA.

Keywords: Biodiversity; Environmental management; Environmental policy; Invasive alien species; Legislation; Stakeholders' knowledge

5.1. Introduction

There is a consensus among experts that invasive alien species (IAS) are a global ecological challenge, that poses a major threat to biodiversity, ecosystem function, and socioeconomic well-being (Kannan et al., 2013; Kelsch et al., 2020; McGeoch et al., 2010). This threat is more severe in catchment areas, as these are geographical areas that are important water sources and a disturbance by IAS has an effect on riverine systems (Early et al., 2016; Hussner et al., 2017). In some instances, the loss of biodiversity in a catchment area may cause huge losses in the food production system among vulnerable communities (Early et al., 2016) by affecting nutrient cycling and water availability, and reducing traditional sources of food as is the case in some environments in the United States of America (USA) (Mayfield et al., 2021).

The impact of IAS on the hydrological flow is immense, with studies carried out in South Africa indicating a possible reduction of 1 444 million m^3 yr^{-1} or approximately 3% of the expected overland flow (Rebelo et al., 2017). The economic losses resulting from such ecological disruption is of immense proportions, with societal losses in the USA alone estimated at $120 billion per annum (Mayfield et al., 2021). Some species, such as those listed on the International Union for Conservation of Nature (IUCN) redlist, may be prone to detrimental effects of IAS (Blackburn et al., 2014). In South Africa, *Prosopis juliflora* and *Opuntia ficus-indica* have been known to affect dry rangelands in the Karoo; they affect ground water reserves and, as a result, impact on water availability to communities, agricultural activities and natural game parks (Zengeya et al., 2020).

In response to the growing global problem of IAS, numerous countries and international organisations have developed policies, management approaches and strategies that aim to prevent, control and mitigate the impacts of IAS through actions which ensure that IAS are not seen as favourable or as sources that improve livelihoods (Dehnen-Schmutz and Novoa, 2022; Kourantidou et al., 2022; Meyerson et al., 2022). Davoren and Shackleton (2021) and Shannon et al. (2020) point out that encouraging actions in which people voluntarily conform through appreciating the benefits derived from excluding or limiting the proliferation of IAS, and that integrated management strategies that focus on poverty reduction and awareness, are important. Herrick (2019) and McGeoch et al. (2010) also suggest that policies encompass prevention and early detection, rapid response, management and public awareness.

The European Union (EU) presents a typical example of a major global player with policy uniformity that addresses the threat of IAS throughout member states (Krämer, 2021). The group has learnt from past experiences the disadvantages of lack of policy harmonisation. The development of uniformity in data systems that enable tracking of IAS within the EU is important in developing awareness and involvement of the citizenry in best practices key to the management of IAS (Cardoso et al., 2017). In countries such as the USA and others in the Global North, and in China and India there are tighter policies and legislation governing the control of IAS in comparison to in developing countries. This is attributable to higher volumes of intercontinental manufacturing agreements (Hulme, 2021). This may be also due to more traffic within the region that encourages the proliferation of IAS due to the ease of travel (Turbelin et al., 2017). Such countries have been able to formulate policies that align with the international policy network on IAS through the Convention on Biological Diversity (CBD). South Africa is among the Global South countries that have rapidly aligned to such international policies and ratified CBD statutes and is at the forefront of academic research on IAS, in particular, invasive alien plants (IAPs) (Bax et al., 2003, Bennet and Van Sitter, 2019; Kelsch et al., 2020; McGeoch et al., 2010). Some preferential trade agreements that facilitate increased trade may require beneficiaries to meet certain standards so as to trade with countries such as the USA and may require increased biosecurity measures (Beghin and Schweizer, 2021).

In the United Kingdom some people dedicate their time and resources to ensuring that policies that promote the conservation of the environment are advanced, and they find time to combat IAS (Pages et al., 2019). Local authorities may use such groups to combat IAS, especially during times of limited financial resources (Wallace et al., 2021). In South Africa in the Province of the Western Cape, approximately 52 volunteer teams were involved in clearing close to 5 300 ha per annum and this amounted to an estimated ZAR 5.1 million towards combating IAPs (Jubase et al., 2021).

Likewise, South Africa has the ‘Working for Water Programme’, which has been instrumental in working with communities and non-profit organisations to eradicate IAS along water courses and other areas (van Wilgen et al., 2020). This programme operates a budget of approximately R1.5 billion per annum (van Wilgen and Wannenburgh, 2016). Government and non-profit organisations support the programme with work activities throughout the country (van Wilgen and Wannenburgh, 2016). However, such programmes are absent in countries such as Zimbabwe, where budgetary constraints are evident.

Herrick (2019) and McGeoch et al. (2010) assert that the adoption and implementation of IAS policies remain a challenge due to many factors that include a country's capacity, coordinated approach, and lack of indicators to monitor and trigger necessary activities. McGeoch et al. (2010) further suggest that there is a link between economic growth and a country's capacity to manage the problem of IAS. Countries may not have appropriate scientific structures that monitor change within the ecosystem and advise on what ecological changes are occurring due to IAS invasion (Vicente et al., 2022). This would be typical of countries, such as Zimbabwe, which are currently experiencing economic problems and losing skilled manpower (Chikowore et al., 2023). On the other hand, Kelsch et al. (2020) point out that, for the effective management of IAS, understanding how local people interact with the IAS and comprehensive stakeholder engagement is important because of the relationship that people have with their landscapes. This underlines the importance of manpower with ecological skills and the knowledge to understand IAS policy and apply it to the threats posed by IAS to a specific country's ecological system.

A country may have a policy, and a good one for that matter, but policy implementation and adoption may pose challenges due to problems with technical expertise, fragmented approaches and funding capabilities (Rebelo and Guerreiro, 2017). Despite variability of legislation and policy that govern IAS across the spectrum of both developing and developed countries, there is recognition of the need for policy frameworks and legal instruments to manage the threats posed by IAS. An analysis of these policies shows that some countries such as Australia, Britain, USA and those in the EU, have clear comprehensive legal frameworks, and strict regulations for IAS control and management (Ng, 2021). In countries like Angola, Botswana, Namibia, Zambia and Zimbabwe, IAS management is mentioned within the broader context of other environmental threats, with little or no specific emphasis on IAS. In developed countries contrary to the Global South, the emphasis is more on acts inclined towards specific IAS management and biodiversity. In the USA, there has been a focus to ensure IAS policy integration and relate it to proliferation and control mechanisms (Herrick, 2019).

A common policy weakness in Global South countries is limited monitoring and enforcement, as seen in India (Azhaguraja and Deo, 2021; Singh, 2022) and Zambia (Constantine et al., 2022). This is often attributed to limited financial resources (Early et al., 2016). Countries such as South Africa recognise the need for increasing financial resources for implementation of policy requirements and continued research on emerging threats posed by new IAPs (Zachariades et al., 2017). Such a targeted approach ensures a more effective utilisation of limited resources available. In contrast, challenges in Global North countries have variously included complex administrative procedures in the European Union (Cardoso et al., 2017), lack of policy integration in the USA (Burgos-Rodríguez and Burgiel, 2020) and lack of responsiveness to new IAS in the United Kingdom (Shannon et al., 2020). It is important to note that, in the KAZA TFCA countries, despite limited resources for implementation of policies there is a growing awareness in local knowledge in Angola (Rebelo and Guerreiro, 2017), Namibia (Ruppel, 2022), Zambia (Constantine et el., 2022) and Zimbabwe (Mujaju et al., 2021).

While some of the countries in the Global North have the capacity, the African continent is faced with many challenges regarding the success of policy implementation, with limited studies carried out on IAS (Courchamp et al., 2017; Shackleton et al., 2019). Challenges faced include a lack of public awareness and engagement, which is crucial for the success of the IAS policies (Shackleton et al., 2019). If the public lacks the understanding of the importance of IAS management, then policy compliance may be low or absent, as articulated by Mogomotsi et al. (2022) in the case of Botswana. Public awareness and engagement may involve holding physical meetings with locals, use of public media, and carrying out educational programmes in public places such as schools and clinics.

In Zimbabwe, the exclusion of the private sector in the dissemination of information to communities and stakeholders affects the level of community knowledge on impacts of IAS (Gobster et al., 2022). In most cases, information dissemination is through radio programs or national newspaper media, which are generally not accessible to poor rural communities many of whom might be affected by IAS. Thus, this weakened structure has profound consequences on the wider knowledge and understanding of laws and policies on IAS (Marchante et al., 2021). A study carried out in the various Provinces in Zimbabwe indicated that the use of information technology, such as broadcasts through radio and television, was not adequate to effectively access poor rural communities, as some households did not have access to valuable communication (Mugwisi et al., 2015). Rural and remote communities are more likely to understand and accept policies channelled through meetings, or if external initiatives take the lead in informing citizens how they would benefit from implementing such new policies

(Ngorima and Shackleton, 2019; Shackleton et al., 2007). This situation would require clear explanations of how such policies would improve their livelihoods (Shackleton et al., 2021).

I employed the Sustainable Livelihoods Framework (SLF) as a theoretical underpinning of this chapter, specifically adopting concepts and arguments of the Department for International Development (DFID, 1999). The SLF is an instrument designed to assist the understanding of the livelihoods of the poor. The SLF itself is broad and far-reaching. For the scope of this chapter, the focus was on the transformative structures and processes in the form of levels of government, private sector, laws, policies, culture or institutions (DFID, 1999). The chapter aims to ascertain stakeholders' knowledge of the management and policies pertaining to the *L. camara* invasion at KAZA TFCA. In essence, the structures in the SLF refer to the private and public organisations whose mandate is the implementation of policy and legislation, and delivery of services (DFID, 1999). The Department for International Development (1999) further notes that if people have no contact with government departments, then they will not know how the government operates, and this is common with rural people in remote areas (Rechel et al., 2016). With this background, the objective of the study was to: (i) Analyse the framing and governance of the management of IAS, including *L. camara*, in the national and sub-national legal and policy frameworks in the Zimbabwean side of the KAZA TFCA, (ii) Solicit the views of stakeholders on the policies and legislation pertaining to *L. camara* invasion in KAZA TFCA, (iii) Survey stakeholders' perceptions of *L. camara* management in KAZA TFCA, and iv) Establish ways for improving the management or control of *L. camara* in KAZA TFCA.

5.2. Methods

5.2.1. Study sites

The study sites are described in detail in section 1.6.1.

5.2.2. Data collection

The study involved different data collection methods to solicit perceptions and framing of the policies pertaining to and management of *L. camara* in the KAZA TFCA. The methods were document review, interviews and household surveys. The document review involved four policy and legal frameworks on IAS management in Zimbabwe. The 11 key informants interviewed comprised 50% males and 50% females. The 300 community members who participated in the household surveys comprised 56% males and 44% females.

5.2.2.1. Document review

A document review process involves critically reading written materials that are secondary sources of information for a study (Crabtree and Miller, 2022). In this study, document review involved the collection and reviewing of legal and policy documents related to the management of *L. camara* in Zimbabwe in general and the KAZA TFCA in particular. These documents were accessed through an online repository and personal requests for copies from relevant government departments such as the Ministry of Environment in Zimbabwe.

5.2.2.2. Key informant interviews

A key informant interview is a one-on-one question-and-answer session between a specifically identified participant and the researcher (Faifua, 2014). The study used a semi-structured interview to acquire information from the informants. A semi-structured interview was selected to provide an in-depth understanding of the informants' interpretations and implementation of the IAS policies and management practices. It also allowed the interviewer to ask follow-up questions and further probe the participants' responses (Braun et al., 2021). During the interviews, the informants discussed their thoughts and wishes, and shortfalls in the legislation and policies. Thus, the interviews provided a productive space for dialogue and discussion with all the informants about the management of IAS in Zimbabwe.

Interviews of key informants in the environmental sector (n=11) were conducted between 2018 and 2019. Participant selection was based on non-probability case sampling, based on their ability to provide in-depth insights into IAS management (Chivanga and Monyai, 2021). The informants included representatives from: the African Lion and Research Trust (ALERT), Department of Agricultural, Technical and Extension Services (AGRITEX), Environment Africa (EA), Forestry Commission – Conservation and Extension (FC-CONEX), Hwange Rural District Council (RDC), Department of Research and Specialist Services (DRSS), and Zimbabwe Parks and Wildlife Authority (ZimParks).

The study employed a phenomenological research approach for the interviews, similar to Lauterbach (2018). Creswell (2021) argues that research objectives can be met through interviewing between six to 10 informants. This estimate made the sampling intensity appropriate, as the study targeted the right group of participants and numbers within the recommended range. The phenomenological research approach is also intended to critically discuss an idea (Silverman, 2020) to surface an informant's opinions on *L. camara* management, management activities being conducted by the organisation they represented, perceptions on who should manage *L. camara*, current policies and legislation in existence and their shortfalls, and the way forward in dealing with *L. camara*. The interview time ranged between 45 to 60 minutes.

5.2.2.3. Household survey

A survey usually uses a questionnaire to collect data (Patten, 2016). Questionnaires captured the socio-demographic information of each respondent, policy knowledge, and involvement in the management of *L. camara*. All respondents participated voluntarily in this study, and informed consent was obtained. Questionnaires (300) were administered to the participants (see Chapter 2 for the selection procedure, language used and duration of the interviews). The respondents interviewed were the household heads. Furthermore, ethical clearance was acquired as stated in Chapter 1.7.1. The socio-demographic data included age, education level, gender and number of years residing in the area. For policy knowledge, respondents were asked if they were aware of any policies or by-laws on *L. camara* and if they were involved in policy or by-law formulation, as well as if they thought the policies were effective. Furthermore, respondents' views on whether something should be done to manage *L. camara*, and their knowledge of current activities for the management of *L. camara* and the organisations involved in management and awareness, also captured. Also, the questionnaire captured respondents' involvement in management programs and whom they thought should be involved in the management of *L. camara,* as well as ways for improving the management.

5.2.3. Data analysis

The policy documents and key informant interviews were subjected to an inductive thematic analysis, with themes manually derived, which involved the organisation of data by identifying central ideas such as policy formulation and management practice implementation (Clarke et al., 2015). Creswell and Creswell (2017) suggest a 'data analysis spiral' to avoid problems associated with large amounts of qualitative data. The spiral breaks the task of data analysis into steps. There were no exclusive steps for this study because the analysis started at the beginning of the generation process (Tracy, 2019). The themes were then condensed and presented in the results. For questionnaire data, descriptive statistics such as graphs were used to present participants' knowledge of the management of *L. camara*, knowledge of policies and awareness of control programs.

5.2.4. Data validation

The study was designed to ensure that the data would be able to explain the phenomena (avoiding internal threats to validity), and the results were used to inform the conclusions and implications, and applicability to other contexts, of the study (avoiding external threats to validity) (Baldwin, 2018). Firstly, the study used different methods to understand both the key informant interviews and the household survey interpretations and practices, thereby minimising threats to theoretical internal validity (Flick, 2018). Secondly, through physical and online conferences, workshops and meetings organised by the Centre for International Research and Agricultural Development (CIRAD) under the Dream Project, data checking and verification were done with all the stakeholders involved in the study soon after the initial data analysis. The data checking and verification with the key informants continued until February 2022, mainly to check if any information needed to be updated, especially on policies. Verifying results between the researcher and interviewees was considered effective in ensuring that the data were authentic (Candela, 2019). Furthermore, the feedback on the findings (for both key informants and households) showed respect for the participants' contributions and their rights to feedback.

5.3. Results

5.3.1. Legalisation and policy frameworks of IAS in KAZA TFCA

There is no specific policy or legislation that governs IAS in all the countries (Angola, Zimbabwe, Namibia, Botswana and Zambia) involved in the KAZA TFCA. However, each country acts as an autonomous entity that is bound by its legal frameworks. This facilitates easy implementation and development of home-grown policies and solutions (Stoldt et al., 2020). Therefore, the obtained documents applied to Zimbabwe, where the study was conducted, and there were five documents: Environmental Management Act (EMA) (Chapter 20:27), Hwange Environmental and Natural Resources Conservation By-Laws, National Biodiversity Strategy and Action Plan (NBSAP) of 2014-2020, National Environmental Policy and Strategies of 2009, and the KAZA TFCA Master Integrated Development Plan.

5.3.1.1. IAS threatens biological diversity

According to the *Environmental Management Act* [EMA] (2005:9), IAS are "exotic plants which have become naturalised and threaten the existence of indigenous species by penetrating and replacing indigenous vegetation". The exact definition is stated in the National Environmental Policy and Strategies of 2009 and the Hwange Environmental and Natural Resources Conservation By-Laws. In a related definition, the NBSAP (National Biodiversity Strategy and Action Plan [NBSAP], 2014:69) defines IAS as a "species whose introduction and spread outside their natural past, or present distribution threatens biological diversity". Thus, one of the themes that have emerged from the document review of the policy documents is that IAS threaten biodiversity.

***5.3.1.2.* Legislation and policy regarding *Lantana camara* L.**

All four analysed documents acknowledge that *L. camara* is an IAS. The EMA (2005: Chapter 20:27) indicates that the *L. camara* is locally known as Cherry-pie and is an IAS throughout Zimbabwe. The existence of *L. camara* is also acknowledged by the NBSAP (2014), which designates it as an IAS. It further indicates that "the government spends US$ 50 million annually to fight IAS and livestock diseases emanating from wildlife areas" (NBSAP, 2014:4). Furthermore, "the freshwater ecosystems are threatened by overgrazing, IAS, informal settlements, urban development and industrial and agricultural pollution" (NBSAP, 2014:13). Beyond that, "IAS is exacerbated by lack of proper framework on regulation, enforcement and control or eradication, loss of indigenous biodiversity, resulting in species losses and ecosystem breakdowns" (NBSAP, 2014:17).

5.3.1.3. Control of IAS is a priority for the government

The EMA (2005: Chapter 20:27) is a crucial legal document for nature conversation in Zimbabwe. The EMA mandates relevant government authorities to take tangible action to protect the environment. Part XIII of the EMA is on the Control of IAS. According to Section 126 of the EMA (2005: Chapter 20:27), "A person responsible may petition the Minister to have a plant declared IAS". As one way of managing IAS in Zimbabwe, the NBSAP (2014) set a target that "By 2020, the threats to biodiversity from IAS have been assessed, and measures put in place to control and manage their impact" (p. 25).

The Hwange Environmental and Natural Resources Conservation By-Laws (n.d.) set out the following management protocols for IAS, including *L. camara*:

> Every occupier or owner of land within the council area shall keep their land free from IAS in line with Section 118 of the EMA (Chapter 20:27) as read with the 1st Schedule of the By-Laws; the council may give a written or oral order to the occupier or owner of the land where IAS is growing to clear or cause to be cleared any IAS from his land; concerning IAS "clearing" means to dig up or pull out and burn IAS or to employ other means of destruction authorised by the EMA; and in cases where there is an imminent threat to the environment, livestock or human health from IAS on land belonging to or occupied by a resident in the council area, the council may take measures to control, remove or clear the land of IAS, at its own cost and recover the expenses incurred from the landowner or occupier as the case may be.

5.3.1.4. Lack of absence of control of IAS is illegal

The NBSAP (2014) expresses that:

> A lack of information about biodiversity and ecosystems conservation and management was identified during the development of the NBSAP. The information relates to baselines, the status of species, terrestrial and freshwater ecosystems, use of non-timber forest products, the extent, impact and control of IAS, and ecosystems services and biodiversity valuation (NBSAP, 2014:33).

As such, the EMA (2005: Chapter 20:27) stipulates that:

> No person shall place or cause or permit to be placed any IAS or the seed of any IAS – (a) in any river, stream or irrigation canal or any other watercourse; or (b) on any road or land.

As a penalty, the Hwange Environmental and Natural Resources Conservation By-Laws (n.d.) says that:

> Any occupier or owner of land who fails to keep his land free from IAS without an adequate and reasonable explanation or fails to comply with a Council order issued in terms of subsection 2 of this section shall be guilty of an offence and liable to a fine to be prescribed by Council.

The NBSAP statement is similar to that of the EMA (2005: Chapter 20:27). On the other hand, the National Environment Policy and Strategies (2009) stipulates that:

> The Government of Zimbabwe will develop and implement a comprehensive plan to control the spread of invasive alien plant species generally on private and public lands, including through requiring landowners and occupiers to remove scheduled noxious IAS from land under their control. While recognising that some alien plant species are economically important in forestry and agriculture.

5.3.2. Stakeholder views on the legal instruments governing *L. camara* invasion

5.3.2.1. Lack of awareness of **L. camara** *as an IAS*

The questionnaire responses revealed that none of the community members was aware of any policies that governed the management of *L. camara*, and they had not been involved in the formulation of any village or ward-level by-laws to that effect. Additionally, about 90% of the villagers were not aware that *L. camara* is listed as an alien invader. To them, it is just like any other plant. They know it in the local language (Ndebele) as “ubuhobe”. They have no idea that it is alien and exotic as it has existed in the area for over 70 years. Consequently, respondents could not comment on whether the current policies were satisfactory or needed improvements. While they expressed ignorance of the current policies, they all pointed out that that there was a more detrimental species in the area that the government should list and prioritise for its eradication.

Box 5.1: Not aware that *L. camara* is listed as an IAP

"We have lived with 'Ubuhobe' [Ndebele word for *L. camara*] for over 60 years in this area, and no livestock has ever been killed but we have another tree species that is very noxious in our knowledge and understanding which we feel the government should assist us in eradicating it". **(Household survey participant)** "Here in the Ndlovu area, the locals have experienced the loss of livestock due to the other plant (I am not sure if it is also an invader) such as 'Umkhawuzane' [a Ndebele word for *Dichapetalum cymosum* that is commonly known as *Gifblaar*]. We have a farmer who lost five cows at once and another who lost about three, so as such the farmers are very careful in ensuring that their livestock do not graze near the detrimental plant. I have not seen anyone being concerned about this plant [*L. camara*]. To my knowledge, so far no one has reported it as a plant of concern as you can see that some of the livestock is browsing on it and also, it is used by many villagers as a hedge". (**Key informant 5)**

The utterance of the participants quoted above suggests that the communities have their own designation on IAS. The government designates *L. camara* as an IAS yet the community sees no problem with it; however, one key informant stressed that *L. camara* is a listed species. According to Environmental Management Act, Chapter 20:27, Fifth Schedule, it is clear – all IAS must be eradicated by landowners, users, occupiers, lessees, etc. However, the key informant did not describe how this information is communicated to the landowners to raise awareness of this policy, especially to those that live in rural areas with limited access to the media. This view differed from another key informant who felt that the policies on IAS should be backed up with tailor-made research on the impacts in a particular area. In the opinion of the key informant*,* *L. camara* was under-researched in terms of benefits.

5.3.2.2. Uncoordinated legal and policy responses to L. camara management

Most key informants shared a view that the legal response to *L. camara* management that calls for its eradication is uncoordinated (Box 5.2). This indicates that different stakeholders have differing views on whether *L. camara* is an IAS. It is thus difficult to convince the communities that it is. The key informant quoted below expressed his frustration with the legal and policy structures that suppress the use of natural resources that communities regard as beneficial or unharmful.

Box 5.2: Further research

"In my opinion, *L. camara* needs to be further researched. Yes, we have statutory instruments that prohibit and ban it, but is it the best? We need to research the benefits of *L. camara*. This thing of just adopting policies without empirical evidence is not wise. For example, *Cannabis indica* remained banned in this country for many years until recently, when the legislation was passed in its favour and farmers that are commercialising it is earning forex through exports. So, I am not in favour of this Act as it has no basis that informs it". (**Key informant 1)**

"So far no legal instruments are guiding the KAZA TFCA. Remember that the KAZA TFCA is composed of five countries (i.e., Angola, Botswana, Namibia, Zambia and Zimbabwe). Each country acts as a sovereignty that is guided by its policy and legal frameworks. What is in place are clusters; for example, I am in the forestry cluster which by right should deal with the IAS, but for the last three years, we have not discussed anything in that aspect". (**Key informant 11)**

"Remember in the mid-1980 Forestry Commission embarked on a huge project of promoting woodlots of Gum trees and other species and in certain areas removing the indigenous forestry to create space to grow and that was done without research informing it, and we later realised that it was not a good thing to do. So, all I am trying to say is that it is good and best to research further on *L. camara* than to go ahead and destroy it". **(Key informant 2)**

The problem we have as a country is that we have not turned the disadvantages into advantages such that we tap into the problem and see what we could benefit from. All we have taken is that it is an invasive alien plant – even maize is alien. Yes, we may have a law/legislation that prohibits *Lantana camara* at the moment, but what basis was it informed upon? Was there enough research done? We need research that will look at all aspects of *Lantana camara*, for example, investigating the soils – should we grow it with other plants, how do soils behave, should we take the leaf litter – and how does the soil behaviour? If it is mixed with leaves, what do we get? Can its stems be used? All those things we have not done in Zimbabwe. **(Key informant 8)**

5.3.2.3. No legal consequences for no control of **L. camara**

All key informants and the local participants were not aware of anyone who has ever been fined or prosecuted for having *L. camara* within their premises (see Box 5.3). While the legislation provides for a legal penalty for having IAS, it is yet to be applied.

Box 5.3: Lack of implementation of the legislation

"We have not prosecuted anyone per se (those that are having the invader within their premises) but the law is clear on this aspect. Furthermore, Zimbabwe is a signatory to the Convention on Biological Diversity (CBD) so we have to abide by that" **(Key informant 9).** "I am not aware of anyone who has been prosecuted for having the IAS within their premises but I am aware that it is a listed species, and our by-laws also mention the penalties" (**Key informant 8).**

5.3.3. Stakeholder Perceptions of the Management of *L. camara*

5.3.3.1. L. camara management programmes are unknown to residents

In the household survey, participants were first asked if they thought something should be done to manage *L. camara* in their communities and 52% of the respondents said they didn't know (Figure 5.1). Additionally, most of the respondents indicated that there were no current programs for the management of *L. camara*. While 31% of the respondents were aware of the previous management programme that focused on eradicating *L. camara* in the Ndlovu ward, the rest of the respondents indicated that there was none (29%) or they were not aware (40%) (Figure 5.1). Furthermore, community members felt it was the responsibility of the government departments to eradicate and manage the plant.

Most communities are unaware of any awareness programmes around *L. camara*. When it comes to the management of *L. camara*, it appears that the protected areas seem to have more focus. Resources were directed into a vibrant programme of *L. camara* eradication, despite the ZimParks pointing out that this programme is not its main mandate, and we observed efforts to ensure that the Victoria Falls National Park remains in a healthy state. In contrast, there have been few or no interventions since over a decade ago when the Environmental Management Agency tried to eradicate *L. camara*. There is gatekeeping or lack of collaboration among departments, for example, in the DRSS and the Environmental Management Agency, which hinders successful management. There was a time when Environmental Management Agency officials wanted to know the person who could have released the biological control of the *Opuntia fulgida* even though the IAS found its way to Zimbabwe naturally. On the other hand, the DRSS was aware of the problem and was involved in conversations with their South African counterparts. So far, the DRSS is aware of the biological control of *L. camara* that is present in the country and is more "naturalised". Richardson et al. (2000) define naturalisation as those alien species that adapt to the natural ecosystems or habitat and do not become invasive, but the Environmental Management Agency is unaware of this development.

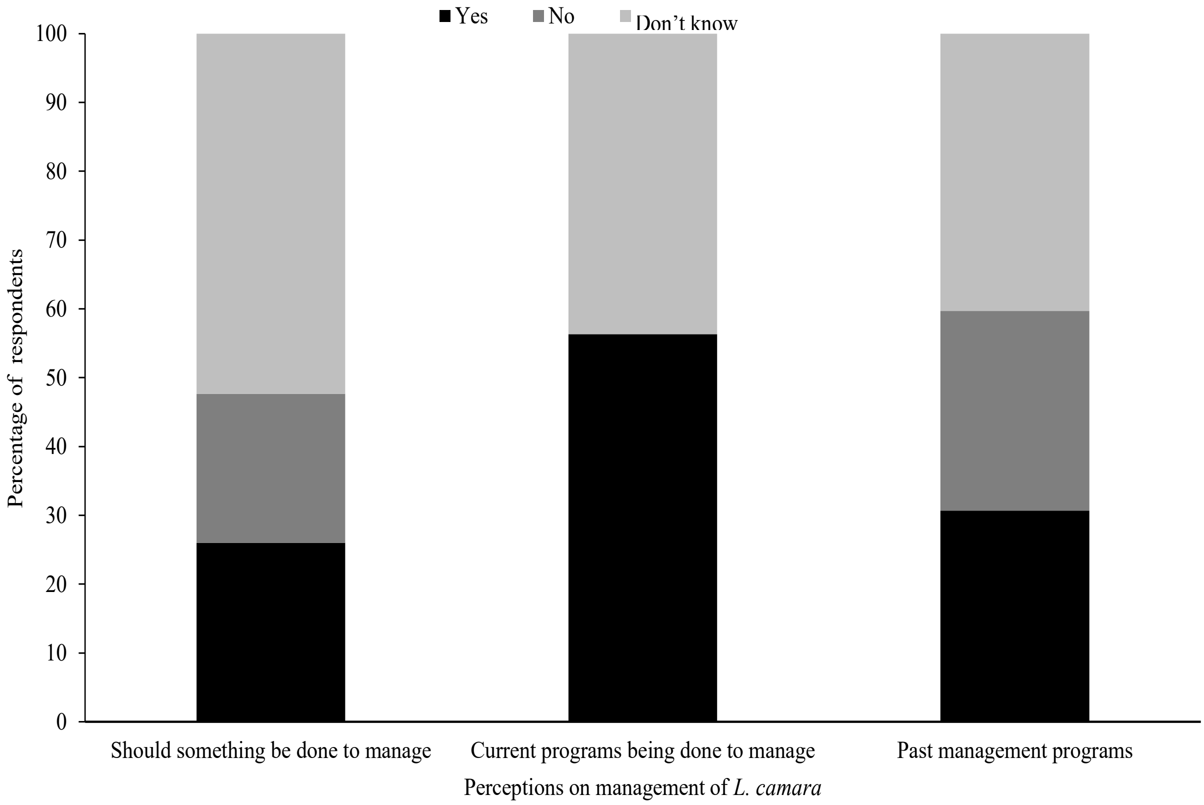

Figure 5.1: Respondents' perceptions of the management of *L. camara* in the Ndlovu ward

5.3.3.2. Roles and responsibility for control and management of IAS

The questionnaire respondents were asked about whose responsibility it was to manage IAS, and 93% indicated that the relevant authorities, such as the Environmental Management Agency and RDC, were responsible. Contrary to this, the key informants stipulated that their roles did not involve the management of IAS. Therefore, the Environmental Management Agency is regarded as the main organisation responsible for IAS management (as mandated by the legislation). This makes other organisations and/or government departments, including local authorities, reluctant to take accountability (Box 5.4).

Box 5.4: Different opinions on roles and responsibilities of IAS management specific to *L. camara*

"Our mandate as ZimParks is to manage the entire wildlife population of Zimbabwe, whether on private or communal lands. By managing, I am referring to protecting and conserving sustainably. We go out of our way and eradicate areas that you have already known such as the Victoria Falls National Park (VFNP) (in collaboration with other partners). At VFNP, other players are participating in eradicating the IAS. However, we do not have the resources that we could use to focus on the eradication of *L. camara* and/or to educate the people in the Communal areas, it is not our primary focus". **(Key informant 7)**

"Our core role is to promote environmental awareness, empowerment of locals to protect and restore the environment, hence our leading role in mobilising other local stakeholders in the eradication of *L. camara* in the rainforest area which has world heritage status and requires to remain pristine" **(Key informant 3)**

"The government should mobilise the resource and run viable programmes for eradicating the IAS, as locals we have other priorities and we do not have the resources" **(Similar sentiment extracted from 93% of the household interviews)**

"EMA is the one that is responsible for the control of invasive alien plants as stated by the legislation. As Forestry Commission we can / could only clear it once it invades our premises be it in the offices or at homes and or once invited by EMA [Environmental Management Agency] in certain projects and so forth, is not our mandate" **(Key informant 2)**

"It is the mandate of everybody affected to remove the weed (*L. camara*), and as an Agency, we are trying hard to educate the communities on this invader more than others but it seems we are not doing enough" **(Key informant 1)**

5.3.3.3. Some management activities involve communities and biological control

The study established that, although many participants are unaware of management programmes for IAS, some members of communities were once involved in some activities that involved manual weeding of *L. camara*, organised by the Environmental Management

Agency in 2010. It was a once-off exercise in the Ndlovu area and the VFNP, specific to the Rainforest area. Similarly, other structures indicated that, in collaboration with the Environmental Management Agency, they were also involved in the mobilisation of local communities to remove *L. camara* manually. Furthermore, the study findings suggest that other structures are willing to support the IAS management, but only if the key government agency takes the lead. Also, it was reported that there is a biological control agent in the country despite the country not promoting its use. Generally, the involvement of communities in IAS management in Zimbabwe has not been clearly defined (Box 5.5).

Box 5.5: Views on who should lead the clearing of *L. camara*

"The project of eradication of *L. camara* was not ours as ours is mainly looking at the conservation of lions and we believed that somehow the presence of the IAS within the protected area was affecting the habitat of lions, hence our involvement in the eradication within the VFNP. So, when Environment Africa invited us, we felt that we should take part in this important project. EMA [Environmental Management Agency] has to be visible on the ground and take a lead on this". **(Key informant 3)**

"We have a strong programme as shown in our case with the handling of the *Opuntia fulgida* where we tried to collaborate with our South African counterparts (Agricultural Research Council) in eradicating it, but due to hindrance of resources we did not complete the project, and the South Africa side went ahead and introduced the biological control whereas with us we could not do anything about it, but lucky enough, the biological control found its way to Zimbabwe. We have a few biological species such as *Calycomyza lantanae and Lantanophaga pusillidactyla* that are known to manage *L. camara* that have self-established in Zimbabwe. They found their way from neighbouring South Africa where they were introduced. Some of the indigenous biological such as *Aristaea onychota* is also present in the country. Unfortunately, despite their presence, they do not seem to be in large quantities hence their minimal effective. **(Key informant 9)**

5.3.3.4. Revised legislation and further research required on L. camara

The findings suggest a need to revise the legislation on IAS and conduct further research on *L. camara*. This need comes against the background that most of the legal stipulations seem unrealistic and that the communities see *L. camara* in a more positive way than as an IAS. The

Environmental Management Agency encourages communities to take responsibility for IAS management, but some stakeholders feel that government needs to take a different approach to this issue (Box 5.2).

Box 5.6: No concern over *L. camara*

"I have not seen anyone concerned about this plant [*L. camara*]. Though I may not have known the scientific name, to my knowledge, so far, no one has reported it as a plant of concern. As you can see, some of the livestock are browsing it, and many villagers use it as a hedge. The locals are more concerned about other problematic plants but not this one. **(Key informant 11)** "I would say those concerned about it should eradicate it. If it is disturbing your peace, remove it. Who has it killed? Nobody! So, why should I would I remove something so beautiful that improves the landscape? I would suggest that those concerned about it should remove it. I call for the review of the legislation" **(Key informant 2)**

The thoughts shared above go back to the earlier observation that what government designates as IAS is not what the communities regard as IAS. As such, management responsibilities could be very tricky. In this situation, some stakeholders feel that the issue should be left to individual discretion.

5.3.4. Improving the management of *L. camara* in KAZA TFCA

5.3.4.1. Four potential management options proposed

Four main management options were suggested in questionnaire responses: (i) mechanical removal, (ii) leaving *L. camara* for community members to use, (iii) using chemicals that destroy *L. camara*, and (iv) use of biological control. Of these options, the preferred one was for *L. camara* to be left for community use (Figure 5.2). Chemical and biological control were the least selected management options (Figure 5.2). Some community members indicated that, although they once took part in physical removal, it was not a management option they would choose as it was labour-intensive. They have seen *L. camara* grow again. Management by using or tapping into the benefits derived from *L. camara* was also a critical factor mentioned. Biological control is part of the options already being explored in the country, albeit with minimal success.

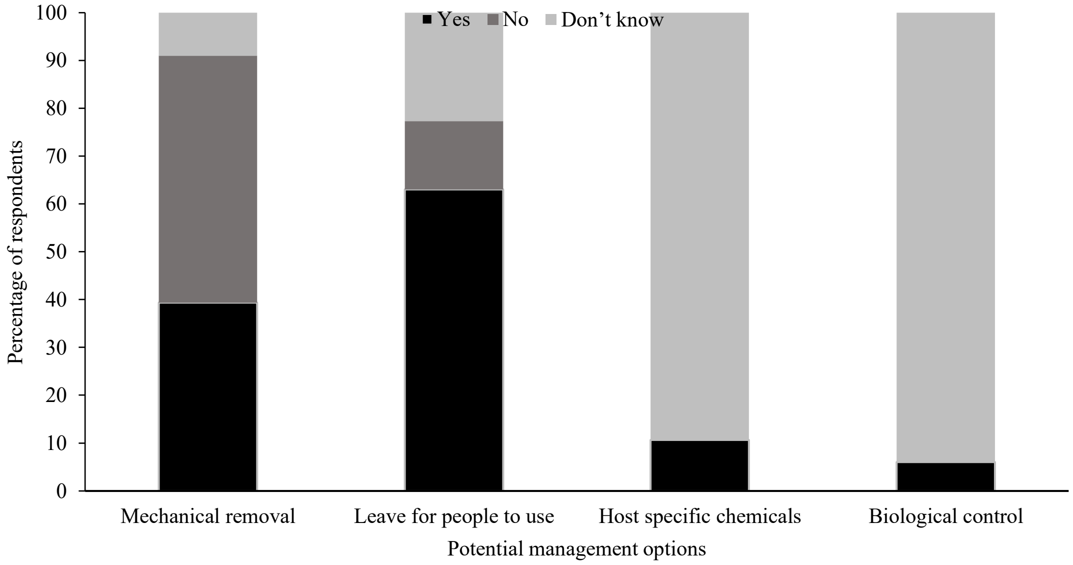

Figure 5.2: Potential management options selected by respondents in the Ndlovu ward

Additionally, factoring in aspects of the ecological role is also vital in shaping the management decisions of *L. camara*. One key informant said:

> When I drive from the office to my house, I see many houses with *L. camara* and at this time around it has flowers and it is so beautiful. Around where I live, some snakes are using it as a habitat and once we destroy it, do we think and consider the ecological consequences of this wildlife that benefit from it? What about soil erosion that it promotes?

5.4. Discussion and Recommendations

5.4.1. Legislation and policy frameworks for IAS management are not effective

The findings of this study have pointed out four crucial points as far as the legislation and policy frameworks IAS in the KAZA TFCA are concerned: (i) IAS threatens biodiversity in the catchment area; (ii) *Lantana camara* is an IAS in the KAZA TFCA; (iii) Different stakeholders have designated the control of IAS as a priority; and (iv) Not controlling IAS is illegal. These results have implications for the livelihoods of the communities around KAZA TFCA. As the SLF helps improve the understanding of the livelihoods of the poor, the framework became necessary in interpreting these research findings. The Department for

International Development (1999) argues that livelihoods entail the mechanism people employ to survive and make a living. In this regard, the worldview or trends in the management of the IAS must be interrogated to transform structures (levels of government and private sector) and processes (laws, policies, culture and institutions).

The countries that form the KAZA TFCA have policies that generally are not specific to IAS, with the responsibility to manage such species incorporated in broader policies and legislation that seek to deal with other environmental or unrelated issues. The section dealing with IAS management is treated as a smaller part of the broader acts tasked with managing a range of other issues which are not within the scope of mitigating IAS threats. In other instances, IAS policies may be spread within various departments which results in infective policy management and implementation. For example, the EMA (2005: Chapter 20:27) of Zimbabwe, has a small section covering the IAS. In certain instances, IAS management is separated from the human well-being component with a greater emphasis towards biodiversity and conservation, yet such strategies should be holistic in addressing various factors including harmonization of the various needs that influence parameters that promote balanced human livelihoods (Moyo et al., 2021).

The KAZA TFCA is an important tourist destination and a significant contributor to the economies of the countries. Local human livelihoods benefit from the continued maintenance of an environment that promotes biodiversity. It is therefore important that policies that are specific to the control of IAS such as *L. camara* are properly assigned and understood by the people who are impacted and those tasked with their implementation. The findings reveal that local people are not aware of the policies (Box 5.1) therefore if the public lacks the understanding of the importance of the IAS management, then compliance with the policy may be low or absent (Mogomotsi et al., 2022).

If *L. camara* is allowed to spread continuously in the KAZA TFCA, particularly the rainforest area, then it has the potential of reducing accessibility to the World heritage site of the Victoria Falls Rainforest, apart from interfering with the diverse vegetation which was observed during the study. *Lantana camara* easily gets spread once it establishes itself in riverine areas and, as shown in the Pauri Garhwal forest in India (Dobhal et al., 2010) and a study carried out in the Zaka District in Zimbabwe (Tarugara et al., 2022), *L. camara* has the potential to outcompete the diverse species recorded within the Victoria falls rainforest areas. The result of this is likely to reduce accessibility and site attraction to tourism, and thus impact on human livelihoods.

The challenges brought by *L. camara* to the management of heritage sites in Zimbabwe are highlighted in a study carried out at Khami World Heritage Site, which saw the complete collapse of the management systems put in place for the maintenance of the site because the negative effects of the invasion resulted in the site being unattractive to visitors (Makuvaza and Makuvaza, 2013).

While the policy is good in paper its implementation has remained a challenge in Zimbabwe. The findings reveal that the EMA (2005: Chapter 20:27) designates different stakeholders the mandate to control the IAS as a priority. However, the various stakeholders seem to have other priorities as seen in Box 5.4. The failure by the mandated organisation to enforce compliance with the IAS instrument makes the EMA weak, more so given that no one has ever been fined as shown in Box 5.3. Herrick (2019) argues that, for a policy to be effective, it requires compliance with rules.

The weight of the scientific literature is that IAS harm species diversity and generally result in substantial economic costs to impacted communities (Diagne et al., 2021). The drive has been to put legislation and policies in place to reduce such negative impacts. During the research, through a review of the policy documents and interviews with key informants, it was found that IAS threaten biological diversity in the area. The EMA (2005) particularly noted that the threat to biodiversity potentially has profound consequences on the livelihoods of the affected communities. However, while it may be true that the IAS harm the ecosystem and can be detrimental, results are context dependent and specific, so there are other factors to consider. In the study site, the findings show that a common request, as seen in Box 5.2, seems to be for evidence-based policies, which also shows that legislation about *L. camara* was adopted from the international protocols rather than country-specific and represents represented needs.

To borrow the wording of the SLF (DFID, 1999), the KAZA TFCA is a natural capital to the communities. When this capital is compromised, other capitals, such as financial, are constrained thus contributing to poverty. Hence there is a need to transform structures and processes. The management of IAS has often been associated with conflicts of interest within communities, especially in cases where communities have found an alternative use for such species and would prefer higher densities in certain areas (Shackleton et al., 2007). This situation is best described in the SLF. Such conflicts can result in difficulties in conveying proposed management initiatives by the authorities (in the various levels of government) tasked with implementing the necessary policy interventions. Due to this complexity, environmental managers and associated practitioners (including the private sector) are faced with situations in which they need to consider the social dimensions or aspects required to manage invasive species effectively (Novoa et al., 2017; Shackleton et al., 2019).

In the SLF, legislation that influences access to or modification of assets impacts the livelihoods of the communities. The assumed purpose of such legislation is to reduce the negative impacts of IAS. This legislation would, as a result, influence the level to which livelihoods are shielded from the stresses and shocks resulting from IAS. Such transforming structures and processes can broadly be referred to as the organisational or institutional mandates designed to mitigate the negative consequences of the presence of IAS (DFID, 1999). The interaction of the legislative body with the communities or entities impacted by the IAS shapes the strategies that communities seek for adaptation.

The key point raised so far is that there is a lack of consensus in the management of *L. camara* in KAZA TFCA. Thus, the situation has created a binary opposition where the community on one side sees more benefits, and authorities on the other side counts the costs and the levels of vulnerability. To negotiate these tensions, interventions tailored to the SLF could help manage such conflicts (Shackleton et al., 2019). Furthermore, the findings show that, according to the law, *L. camara* remains an IAS in the KAZA TFCA. The main mandate of KAZA TFCA is to encourage economic development which is underlined by the conservation of its distinct biodiversity. The unhindered migration of wildlife within the demarcated area is unique to this ecosystem. It is therefore imperative that the management prescriptions within the geographical system are similar. The management of IAS, and in particular the recognition of *L. camara* as an IAS requiring control and eradication, is therefore important. In all the five countries covering the KAZA TFCA, there seem to be some policies governing the control of IAS (Milton et al., 2003; Ntalo et al., 2022).

A look at the specific policies, acts or legislation governing the management of IAS in some countries and comparing such with the countries that are members of the KAZA TFCA shows that there is no unified legislation or policy that seeks to address the threat of IAS with some countries who form part of the unique habitat having done little to address the some of the shortcomings that were identified previously (Jones, 2008). The policies within the member states of the KAZA TFCA lack a transnational strategy in the management of the threat posed by IAS, as there is a general fragmentation of policies. In Southern Africa, such fragmentation in governance policies further complicates enforcement (Lubbe and Kotzé, 2019). When the policy is fragmented, it exists on paper and not in its implementation, as shown by the findings of this study. There is a lack of uniformity in the degree of policy ratification, uniformity and implementation, despite the existence of the Master Development Plan which focuses on wildlife management issues with no emphasis on IAS (Naidoo et al., 2018).

According to the laws within KAZA TFCA, *L. camara* is considered an IAS despite the problem that the legislation may not be enforced as per the requirements. Resources and structures in place are not adequate to cover this requirement. The study finds that *L. camara* is not considered a threat by the locals as they seem to benefit from the various uses derived from its presence. This is further complicated because the study shows that no one has been prosecuted as required by the policies and legislation. The failure to enforce legislation by the government, on the control of *L. camara* and other environmental-related issues, is recognised as a major problem in Zimbabwe (Jimu and Nyakudya, 2018). It is further acknowledged that the current economic problems being faced by some of the countries within the KAZA TFCA have further marginalised the implementation of policies that promote effective management of natural resources (Average, 2010).

5.4.2. Key stakeholders lack knowledge of IAS legislation and policies

According to the study's findings, lack of awareness of *L. camara* as an IAS, uncoordinated legal responses to *L. camara*, and lack of legal consequences for failing to control *L. camara* are some of the factors affecting the responses to the IAS invasion in the KAZA TFCA. It is generally accepted that significant research on IAS has dealt with ecological impacts, with much less on issues that guide policy-related aspects and the resultant impact on livelihoods (Shackleton et al., 2019). Due to the high costs associated with the management of IAS, it has become imperative for policies to be formulated to manage the effects of the IAS in a manner that improves the livelihood of communities. The IAS rapidly spread into ecosystems, where they disrupt biodiversity and change how communities rely on the impacted environments. Therefore, policies are needed to provide an integrated management strategy to reduce the negative impacts of the IAS on livelihoods, ecology and the economy (Shackleton et al., 2019). Thus, if structures and processes are not effective, it is imperative to transform them (Box 5.2).

Much as legislation and policies could spearhead the management of *L. camara* in the KAZA TFCA, it would remain rhetorical unless key stakeholders have some knowledge of them.

There are cultural perceptions about the IAS that need to be transformed. For example, Box 5.1 shows that participants are not aware that *L. camara* is an IAS, as they have always lived with it. Therefore, it shows that the legislation is not known, let alone owned, by the local people. Furthermore, it is just toothless legislation (Box 5.3) that echoes international treaties and conventions such as the Convention for Biodiversity, because, since its adoption in Zimbabwe, it has never been implemented, and no one has ever been fined or prosecuted.

Compliance with these policies and effective management of IAS requires a holistic multi-stakeholder approach that engages government agencies, scientific institutions, non-governmental organisations, local communities and the public (Foley et al., 2017). The effectiveness of an IAS policy depends on various factors which include implementation, monitoring, and adaptation to changing circumstances. Policies alone cannot eliminate the challenges posed by IAS, but, when supported by effective management practices, they can be a crucial tool in minimising the impacts and preventing the spread of IAS.

Thus, the general outlook of the research findings is that crucial stakeholders lack knowledge of the legislation and policies on managing IAS, including *L. camara*. In the transforming structures and processes of the SLF (DFID, 1999), the focus is on whether environmental management policies are restrictive or benefit livelihoods. If implemented efficiently, environmental management policies are beneficial in improving community livelihoods. For example, in Madagascar, communities benefitted from ecosystem management policies (Ward et al., 2018). The positive impact of ecosystem management policies was more noticeable in poorer or remote communities. In Zimbabwe, the policy seeks to provide a framework in which the management of the environment is coordinated under one umbrella body, and therefore to avoid uncoordinated responses, which was one of the critical findings of this study from various stakeholders. This situation speaks to a need to transform levels of government as a structure and to enhance institutions as a process.

Management of environmental resources is meant to reduce policy deficiencies and improve resource utilisation (Chitakira and Nyikadzino, 2020). Such efforts have the effect of giving a unified response for tracking and managing the effects of IAS. A unified agency means that affected communities know who is responsible, and is more likely to assist in mitigating the adverse effects of IAS. In Zimbabwe, IAS contribute to high levels of poverty through the threat posed to agricultural yields, and the inherent effect on the achievement of sustainable development goals, especially in poor rural communities (Mujaju et al., 2021). A sound environmental management policy informed by the SLF theoretical framework makes it easier for resources to be channelled into communities where there is a proper needs assessment based on proper research and data to support the need for developing intervention strategies that should resolve the problem of IAS management.

5.4.3. Most stakeholders perceive IAS management as the role of the government

The study found that stakeholders' perceptions of the management of *L. camara* vary widely. However, there is a general perception that the government should be the sole entity responsible for the management of IAS in the KAZA TFCA. This perception affects how people sustain their natural resource base (DFID, 1999). In the same vein, the perception raises questions about whether the stakeholders understand the importance of the formulated environmental management legislation and policies related to IAS, including *L. camara*. Stakeholder consultation is essential in policy formulation to bring together a broad knowledge base and set ideas. Studies carried out in countries such as South Africa, Australia, Canada, India, Spain and the USA have confirmed the importance of stakeholder involvement in the formulation of policies that are more likely to play a mitigatory role on the negative effects of IAS (Shackleton et al., 2019). Such an approach has been found to increase adoption methodologies that result in better management of IAS (Novoa et al., 2017).

In cases where stakeholders are consulted during policy formulation, it makes it easier for the communities to understand why the policy is essential and how it would influence their livelihoods. The consultation aligns policies better and would make it easier to use such policies when addressing the shocks and stresses that communities experience. Consultation for IAS management is important (Shackleton et al., 2019). In the KAZA TFCA, just as in studies in South Africa and other regions, policy understanding seems to be aligned and driven by the desire of the organisations (including the private sector) to fulfil administrative and compliance requirements as opposed to taking into consideration community concerns (7% inclined towards implementation and uptake and feedback) to stakeholders being as low as 11%) (Shackleton et al., 2019). Thus, it is no surprise that the study found that communities perceive IAS management as the sole responsibility of the government. Such low implementation can be attributed to lack of enforcement, resources and knowledge. To mitigate this situation, some experts saw the need for enhancing research and inclusion, to inform decisions, as critical for the effective management of resources within the KAZA TFCA (Box 5.2) (Stoldt et al., 2020).

Zimbabwe has put in place a National Biodiversity Strategy Action plan whose main purpose is to target the effective management of IAS and ensure that legislation is more aligned towards better awareness and control (Mujaju et al., 2021). The NBSAP targets for 2020 have not been met, as gathered from the findings of this study through the key informant interviews. The failure to meet the targets reduces the NBSAP to be just a paper which is one of the challenges that McGeoch et al. (2010) cite. Despite the availability of legislation such as the EMA (2005), there is a greater concern as regards the funding models available for the management of IAS and other environmental challenges (Chikodzi and Chapungu, 2023). Even though the EMA (2005) policy has been in existence for more than a decade it still suffers from past issues relating to a lack of adequate stakeholder consultation and awareness, which is worsened by the lack of funding due to the current economic situation in the country (Marambanyika and Beckedahl, 2016). There is a general collapse of the critical arms of government in Zimbabwe, and much-needed resources both in terms of human and financial capital have not been allocated to areas that are perceived as peripheral to the current economic problems the country is facing. As a result there is less focus put towards public awareness of the long-term negative effects of the continued spread of *L. camara* on local livelihoods.

In other countries such as India, there is an Invasive Species Management Framework which provides a structured system through which guidelines are set for the reporting, and a systematic approach to the reporting and stipulating required measures to control threats from IAS. This framework also involves effective detection and management protocols supported by effective legislation (Priyanka, 2013). The framework also set mechanisms for funding and collaboration with both private organisations and businesses to drive public awareness and control strategies. Unlike the uncoordinated approaches found in KAZA TFCA countries and the lack of both awareness and funding, the Indian approach stems from experience which has resulted in structures being set up which are well-funded to deal with stakeholder involvement and mechanisms for control of IAS using the relevant policies and legislation. A major challenge to policy enforcement is that there may be little or no emphasis on ensuring that policies are understood by the very people whose behaviour they seek to regulate through stakeholder involvement (Shackleton et al., 2019). In other developing countries, such as Nepal, processes are constrained due to the lack of a concise scientific protocol to deal with IAS; this impacts policy and stakeholder involvement initiatives (Shrestha, 2019).

In South Africa, while funding and policy clarity exist, there have been instances where people have differing views about socioeconomic values ascribed to some IAS, and certain instances have been complicated by private land ownership where government may not have much influence on IAS management protocols (Bennett and Van Sittert, 2019). Regions, just as much as countries, experience challenges that are unique to their socio-geographical locations which are influenced by funding, policy, stakeholders, and other issues of governance. There is therefore variability in the management prescriptions taken by the government to manage IAS. Foley et al. (2017) point out that complying with these policies and effective management of IAS requires a holistic multi-stakeholder approach that engages government agencies, scientific institutions, non-governmental organisations, local communities and the public.

Despite the enactment of the environmental management policy in Zimbabwe in 2003, communities still find conflicting situations in how the policy implementation process is driven. Their experience is also the slowness and reluctance associated with properly implementing the policy at the grassroots level, due to the lack of financial and other resources. In other instances, there has been infighting between environmental management bodies and local authorities (Sinthumule and Mkumbuzi, 2019). This level of conflict has resulted in communities getting mixed messages about the role of environmental policies in improving their livelihoods; hence, they see the government as responsible for IAS management. Thus, the levels of government need to be transformed so that communities can be mobilised in the IAS management. Doing so would contribute to the reduced vulnerability of the communities to the effects of IAS.

5.4.4. Transforming the management of *L. camara* requires a holistic approach

Overall, the research findings suggest that transforming the management of *L. camara* in the KAZA TFCA needs a holistic approach which is in line with Foley et al. (2017) and McGeoch et al. (2010). If the structures and processes are transformed in a participatory way, which is a vital underpinning of the SLF, then IAS management could be improved. However, the success would depend on whether enforcing legislation and policies promotes community acceptance of IAS management frameworks, despite the previous approaches that communities saw in IAS management. If KAZA TFCA hopes to succeed in the control and management of *L. camara*, it has to ensure that the strategy involves all affected stakeholders including the rural stakeholders, to integrate the information and views. Shackleton et al. (2019) postulate that the engagement of stakeholders is needed to integrate various information and views in the management of the IAS.

The management of IAS is a matter of concern worldwide, and the issue of deployment of resources presents challenges even in developed countries such as the United Kingdom (Tollington et al., 2017). According to this view, ensuring that policies are implemented presents challenges globally, and more so in developing economies where accessing resources and management are challenging. In Zimbabwe, the implementation of the Environmental Management Act is faced with similar challenges due to scarce resources and the lack of awareness that systems being put in place through laws and policies are meant to improve the livelihoods of the communities. Since the processes are not understood, eventually vulnerability increases and hence there is an increase in poverty.

5.5. Conclusion

This chapter explored stakeholders' knowledge of the management of and policies about the *Lantana camara* invasion in the Kavango Zambezi Transfrontier Conservation Area. Household surveys, key informant interviews and document reviews were employed in the study. The research findings suggest that legislation and policy frameworks for managing IAS are ineffective as no one faces legal consequences for not controlling IAS. The research further revealed that critical stakeholders, such as community members, lack knowledge of IAS legislation and policies. Moreover, most stakeholders perceive IAS management as the role of government. The study recommends a holistic approach to transforming IAS management structures and processes, including *L. camara*, whereby laws and policies provide penalties, and stakeholders know their duties and responsibilities in IAS management.

References

Average, C., 2010. Institutional decay and environmental degradation in Ward 29 Gutu West, Zimbabwe. *Journal of Sustainable Development in Africa*, 12(1), 323-333.

Azhaguraja, N. and Deo, M., 2021. Environmental governance: Compliances and consequences. In H. Kaur (ed.), *Facets of corporate governance and corporate social responsibility in India*, 97-108. Singapore: Springer Singapore.

Baldwin, L., 2018. Internal and external validity and threats to validity. In L. Baldwin, *Research concepts for the practitioner of educational leadership*, 31-36. Brill.

Bax, N., Williamson, A., Aguero, M., Gonzalez, E. and Geeves, W., 2003. Marine invasive alien species: A threat to global biodiversity. *Marine Policy*, 27(4), 313-323.

Beghin, J.C. and Schweizer, H., 2021. Agricultural trade costs. *Applied Economic Perspectives and Policy*, 43(2), 500-530.

Bennett, B.M. and Van Sittert, L., 2019. Historicising perceptions and the national management framework for invasive alien plants in South Africa. *Journal of Environmental Management*, 229, 174-181.

Blackburn, T.M., Essl, F., Evans, T., Hulme, P.E., Jeschke, J.M., Kühn, I., Kumschick, S., Marková, Z., Mrugała, A., Nentwig, W. and Pergl, J., 2014. A unified classification of alien species based on the magnitude of their environmental impacts. *PLoS Biology*, 12(5), 1001850.

Braun, V., Clarke, V., Boulton, E., Davey, L. and McEvoy, C., 2021. The online survey as a qualitative research tool. *International Journal of Social Research Methodology*, 24(6), 641-654.

Burgos-Rodríguez, J. and Burgiel, S.W., 2020. Federal legal authorities for the early detection of and rapid response to invasive species. *Biological Invasions*, 22(1),129-146.

Candela, A.G., 2019. Exploring the function of member checking. *The Qualitative Report*, 24(3), 619-628.

Cardoso, A.C., Tsiamis, K., Gervasini, E., Schade, S., Taucer, F., Adriaens, T., Copas, K., Flevaris, S., Galiay, P., Jennings, E. and Josefsson, M., 2017. Citizen science and open data: A model for invasive alien species in Europe. *Research Ideas and Outcomes,* 3(14), 811.

Chikodzi, D. and Chapungu, L., 2023. Deciphering synergies and trade-offs between COVID-19 measures and the progress towards SDG 15-Life on land in Zimbabwe. In L. Chapungu, D. Chikodzi and K. Dube (eds), *The COVID-19-health systems nexus: Emerging trends, issues and dynamics in Zimbabwe*, 209-220. Springer International Publishing, Cham.

Chikowore, G., Martin, G.D., Chidawanyika, F., Hill, M., Neser, S., Day, M., Grice, T., Chikwenhere, G., Mangosho, E. and Sheppard, A., 2023. Weed biological control in Zimbabwe: Challenges and future prospects. *South African Journal of Botany*, 154, 336-345.

Chitakira, M. and Nyikadzino, B., 2020. Effectiveness of environmental management institutions in sustainable water resources management in the upper Pungwe River basin, Zimbabwe. *Physics and Chemistry of the Earth, Parts A/B/C*, 118-119, 102885.

Chivanga, S.Y. and Monyai, P., 2021. Back to basics: Qualitative research methodology for beginners. *Journal of Critical Reviews*, 8(2), 11-17.

Clarke, V., Braun, V. and Hayfield, N., 2015. Thematic analysis. In J.A. Smith (ed.) *Qualitative psychology: A practical guide to research methods*, 222-248. 3rd edition. Sage.

Constantine, K., Mulila-Mitti, J. and Williams, F., 2022. An invasive species system assessment in Zambia. *CABI Working Paper 27*, 1–21. CABI.

Courchamp, F., Fournier, A., Bellard, C., Bertelsmeier, C., Bonnaud, E., Jeschke, J.M. and Russell, J.C., 2017. Invasion biology: specific problems and possible solutions. *Trends in Ecology & Evolution*, 32(1), 13-22.

Crabtree, B.F. and Miller, W.L., 2022. *Doing qualitative research*. 3rd edition. Sage Publications, United States.

Creswell, J.W. 2021. *Research design: Qualitative, quantitative and mixed methods approaches*. 5th edition. Sage Publications, United State of America.

Creswell, J.W. and Creswell, J.D., 2017. *Research design: Qualitative, quantitative, and mixed methods approaches.* Sage Publications, United States.

Davoren, E. and Shackleton, C.M., 2021. Urban ecosystem disservices in the Global South. In C.M. Shackleton, S.S. Cilliers, E. Davoren and M.J. du Toit (eds), *Urban ecology in the Global South*, 265-292. Springer, Cham.

Dehnen-Schmutz, K. and Novoa, A., 2022. Advances in the management of invasive plants. In D.R. Clements, M.K. Upadhyaya, S. Joshi and A. Shrestha (eds), *Global plant invasions*, 317-330. Springer, Cham.

Department for International Development [DFID], 1999. *Sustainable livelihoods sheets.* DFID, London, United Kingdom.

Diagne, C., Leroy, B., Vaissière, A.C., Gozlan, R.E., Roiz, D., Jarić, I., Salles, J.M., Bradshaw, C.J. and Courchamp, F., 2021. High and rising economic costs of biological invasions worldwide. *Nature*, 592(7855), 571-576.

Dobhal, P.K., Kohli, R.K. and Batish, D.R., 2010. Evaluation of the impact of *Lantana camara* L. invasion, on four major woody shrubs, along Nayar river of Pauri Garhwal in Uttarakhand Himalaya. *International Journal of Biodiversity and Conservation*, 2(7), 155-161.

Early, R., Bradley, B.A., Dukes, J.S., Lawler, J.J., Olden, J.D., Blumenthal, D.M., Gonzalez, P., Grosholz, E.D., Ibañez, I., Miller, L.P. and Sorte, C.J., 2016. Global threats from invasive alien species in the twenty-first century and national response capacities. *Nature Communications*, 7(1), 12485.

Environmental Management Act, *Act No.13 of 2002 (Chapter 20:27), revisions under Act No.5 of 2004 (s.23) and Act No. 6 of 2005 (s.28)* [EMA], 2005. Veritas Trust, Zimbabwe.

Faifua, D., 2014. *The key informant technique in qualitative research*. Sage Publications, United States.

Flick, U., 2018. *Managing quality in qualitative research, Vol 10.* Sage Publications, United States.

Foley, R.W., Wiek, A., Kay, B. and Rushforth, R., 2017. Ideal and reality of multi-stakeholder collaboration on sustainability problems: A case study on a large-scale industrial contamination in Phoenix, Arizona. *Sustainability Science*, 12, 123-136.

Gobster, P.H., Weber, E., Floress, K.M., Schneider, I.E., Haines, A.L. and Arnberger, A., 2022. Place, loss, and landowner response to the restoration of a rapidly changing forest landscape. *Landscape and Urban Planning*, 222, 104382.

Herrick, C.N., 2019. A review of the US invasive species policy mix: Questioning the prospect of an integrated regime. *Environmental Policy and Governance*, 29(4), 262-278.

Hulme, P.E., 2021. Unwelcome exchange: International trade as a direct and indirect driver of biological invasions worldwide. *One Earth*, 4(5), 666-679.

Hussner, A., Stiers, I., Verhofstad, M.J.J.M., Bakker, E.S., Grutters, B.M.C., Haury, J., Van Valkenburg, J.L.C.H., Brundu, G., Newman, J., Clayton, J.S. and Anderson, L.W.J., 2017.

Management and control methods of invasive alien freshwater aquatic plants: A review. *Aquatic Botany*, 136, 112-137.

Hwange Environmental and Natural Resources Conservation By-Laws, n.d. (Unpublished).

Jimu, L. and Nyakudya, I.W., 2018. Fires in exotic forest plantations of Zimbabwe: Causes and management strategies. *World Development Perspectives*, 9, 56-58.

Jones, B.T., 2008. Legislation and policies relating to protected areas, wildlife conservation, and community rights to natural resources in countries being partner in the Kavango Zambezi Transfrontier Conservation Area. *African Wildlife Foundation and Swiss Agency for Development and Cooperation*.

Jubase, N., Shackleton, R.T. and Measey, J., 2021. Motivations and contributions of volunteer groups in the management of invasive alien plants in South Africa's Western Cape Province. *Bothalia-African Biodiversity & Conservation*, 51(2), 1-13.

Kannan, R., Shackleton, C.M. and Shaanker, R.U., 2013. Playing with the forest: Invasive alien plants, policy and protected areas in India. *Current Science*, 1159-1165.

Kelsch, A., Takahashi, Y., Dasgupta, R., Mader, A.D., Johnson, B.A. and Kumar, P., 2020. Invasive alien species and local communities in socio-ecological production landscapes and seascapes: A systematic review and analysis. *Environmental Science and Policy*, 112, 275-281.

Kourantidou, M., Haubrock, P.J., Cuthbert, R.N., Bodey, T.W., Lenzner, B., Gozlan, R.E., Nuñez, M.A., Salles, J.M., Diagne, C. and Courchamp, F., 2022. Invasive alien species as simultaneous benefits and burdens: Trends, stakeholder perceptions and management. *Biological Invasions*, 24, 1905-1926.

Krämer, L., 2021. Managing invasive alien species by the European Union: Lessons learnt. *Journal for European Environmental and Planning Law*, 18(3), 297-312.

Lauterbach, A.A., 2018. Hermeneutic phenomenological interviewing: Going beyond semi-structured formats to help participants revisit experience. *The Qualitative Report*, 23(11), 2883-2898.

Lubbe, W.D. and Kotzé, L.J., 2019. Holistic biodiversity conservation in the Anthropocene: A Southern African perspective. *African Journal of International and Comparative Law*, 27(1), 76-99.

Makuvaza, S. and Makuvaza, V., 2013. The challenges of managing an archaeological heritage site in a declining economy: The case of Khami World Heritage Site in Zimbabwe. *Conservation and Management of Archaeological Sites*, 15(3-4), 281-297.

Marambanyika, T. and Beckedahl, H., 2016. The missing link between awareness and the implementation of wetland policy and legislation in communal areas of Zimbabwe. *Wetlands Ecology and Management*, 24, 545-563.

Marchante, H., Palhas, J., Núñez, F.A.L. and Marchante, E., 2021. Invasive species impacts and management. In W. Leal Filho, A. Azul, L. Brandli, A. Lange Salvia and T. Wall (eds), *Life on land. Encyclopedia of the UN Sustainable Development Goals*, 560-571. Springer, Cham.

Mayfield, A.E., Seybold, S.J., Haag, W.R., Johnson, M.T., Kerns, B.K., Kilgo, J.C., Larkin, D.J., Lucardi, R.D., Moltzan, B.D., Pearson, D.E. and Rothlisberger, J.D., 2021. Impacts of invasive species in terrestrial and aquatic systems in the United States. In T.M. Poland, T. Patel-Weynand, D.M. Finch, C.F. Miniat, D.C. Hayes and V.M. Lopez (eds), *Invasive species in forests and rangelands of the United States: A comprehensive science synthesis for the United States forest sector*, 5-39. Springer Nature, Switzerland.

McGeoch, M.A., Butchart, S.H., Spear, D., Marais, E., Kleynhans, E.J., Symes, A., Chanson, J. and Hoffmann, M., 2010. Global indicators of biological invasion: Species numbers, biodiversity impact and policy responses. *Diversity and Distributions*, 16(1), 95-108.

Meyerson, L.A., Pauchard, A., Brundu, G., Carlton, J.T., Hierro, J.L., Kueffer, C., Pandit, M.K., Pyšek, P., Richardson, D.M. and Packer, J.G., 2022. Moving toward global strategies for managing invasive alien species. In D.R. Clements, M.K. Upadhyaya, S. Joshi and A. Shrestha (eds), *Global plant invasions*, 331-360. Springer, Cham.

Milton, S.J., Dean, W.R.J. and Richardson, D.M., 2003. Economic incentives for restoring natural capital in southern African rangelands. *Frontiers in Ecology and the Environment*, 1(5), 247-254.

Mogomotsi, G.E., Mogomotsi, P.K. and Moeti, O.O., 2022. WTO Law and jurisprudence on invasive alien species in the Global South. *Chinese Journal of Environmental Law*, 6(1), 63-86.

Moyo, H., Slotow, R., Rouget, M., Mugwedi, L., Douwes, E., Tsvuura, Z. and Tshabalala, T., 2021. Adaptive management in restoration initiatives: Lessons learned from some of South Africa's projects. *South African Journal of Botany*, 139, 352-361.

Mugwisi, T., Mostert, J. and Ocholla, D.N., 2015. Access to and utilization of information and communication technologies by agricultural researchers and extension workers in Zimbabwe. *Information Technology for Development*, 21(1), 67-84.

Mujaju, C., Mudada, N. and Chikwenhere, G.P., 2021. Invasive alien species in Zimbabwe (Southern Africa). In T. Pullaiah and M.R. Ielmini (eds), *Invasive alien species: Observations and issues from around the world*, 330-361. Wiley Online Library.

Naidoo, R., Kilian, J.W., Du Preez, P., Beytell, P., Aschenborn, O., Taylor, R.D. and Stuart-Hill, G., 2018. Evaluating the effectiveness of local-and regional-scale wildlife corridors using quantitative metrics of functional connectivity. *Biological Conservation*, 217, 96-10.

National Biodiversity Strategy and Action Plan 2014 [NBSAP], 2014. Ministry of Environment, Water and Climate, Republic of Zimbabwe.

Ng, K., 2021. Towards an outcomes-driven approach to environmental law: The ‘Environment Protection and Biodiversity Conservation Act 1999’ (Cth). *Environmental and Planning Law Journal,* 38(3) 223 -236.

Ngorima, A. and Shackleton, C.M., 2019. Livelihood benefits and costs from an invasive alien tree (*Acacia dealbata*) to rural communities in the Eastern Cape, South Africa. *Journal of Environmental Management*, 229, 158-165.

Novoa, A., Dehnen-Schmutz, K., Fried, J. and Vimercati, G., 2017. Does public awareness increase support for invasive species management? Promising evidence across taxa and landscape types. *Biological Invasions*, 19(12), 3691-3705.

Ntalo, M., Ravhuhali, K.E., Moyo, B., Hawu, O. and Msiza, N.H., 2022. *Lantana camara*: poisonous species and a potential browse species for goats in southern Africa—a review. *Sustainability*, 14(2), 751.

Pages, M., Fischer, A., van der Wal, R. and Lambin, X., 2019. Empowered communities or “cheap labour”? Engaging volunteers in the rationalised management of invasive alien species in Great Britain. *Journal of Environmental Management*, 229, 102-111.

Patten, M., 2016. *Questionnaire research: A practical guide*. Routledge.

Priyanka, N., 2013. A framework for management of *Lantana camara* in India. *Proceedings of the International Academy of Ecology and Environmental Sciences*, 3(4), 306.

Rebelo, C. and Guerreiro, J., 2017. Comparative evaluation of the EIA systems in Kenya, Tanzania, Mozambique, South Africa, Angola, and the European Union. *Journal of Environmental Protection*, 8(5), 603-636.

Rechel, B., Džakula, A., Duran, A., Fattore, G., Edwards, N., Grignon, M., Haas, M., Habicht, T., Marchildon, G.P., Moreno, A. and Ricciardi, W., 2016. Hospitals in rural or remote areas: An exploratory review of policies in 8 high-income countries. *Health Policy*, 120(7), 758-769.

Richardson, D.M., Pyšek, P., Rejmanek, M., Barbour, M.G., Panetta, F.D. and West, C.J., 2000. Naturalization and invasion of alien plants: Concepts and definitions. *Diversity and Distributions*, 6(2), 93-107.

Ruppel, O.C., 2022. Environmental law in Namibia–an overview. In O.C. Ruppel and K. Ruppel-Schlichting (eds), *Environmental law and policy in Namibia*, 75-108. 4th edition. Nomos Verlagsgesellschaft.

Shackleton, C.M., Cilliers, S.S., du Toit, M.J. and Davoren, E., 2021. The need for an urban ecology of the Global South. In C.M. Shackleton, S.S. Cilliers, E. Davoren and M.J. du Toit (eds), *Urban ecology in the Global South,* 1-26. Springer, Cham.

Shackleton, C.M., McGarry, D., Fourie, S., Gambiza, J., Shackleton, S.E. and Fabricius, C., 2007. Assessing the effects of invasive alien species on rural livelihoods: Case examples and a framework from South Africa. *Human Ecology*, 35(1), 113-127.

Shackleton, R.T., Adriaens, T., Brundu, G., Dehnen-Schmutz, K., Estévez, R.A., Fried, J., Larson, B.M., Liu, S., Marchante, E., Marchante, H. and Moshobane, M.C., 2019. Stakeholder engagement in the study and management of invasive alien species. *Journal of Environmental Management*, 229, 88-101.

Shackleton, R.T., Richardson, D.M., Shackleton, C.M., Bennett, B., Crowley, S.L., Dehnen-Schmutz, K., Estévez, R.A., Fischer, A., Kueffer, C., Kull, C.A. and Marchante, E., 2019.

Explaining people's perceptions of invasive alien species: A conceptual framework. *Journal of Environmental Management*, 229, 10-26.

Shackleton, R.T., Shackleton, C.M. and Kull, C.A., 2019. The role of invasive alien species in shaping local livelihoods and human well-being: A review. *Journal of Environmental Management*, 229, 145-157.

Shannon, C., Stebbing, P.D., Dunn, A.M. and Quinn, C.H., 2020. Getting on board with biosecurity: Evaluating the effectiveness of marine invasive alien species biosecurity policy for England and Wales. *Marine Policy*, 122, 104275.

Shrestha, B.B., 2019. Management of invasive alien plants in Nepal: Current practices and future prospects. In S. Garkoti, S. Van Bloem, P. Fulé and R. Semwal (eds) *Tropical ecosystems: Structure, functions and challenges in the face of global change*, 45-68. Springer, Singapore.

Silverman, D. (ed.), 2020. *Qualitative research*. Sage Publications, London.

Singh, B., 2022. Biological Diversity Act 2002 and its importance for conservation of biological resources. *Central Asian Journal of Theoretical and Applied Science*, 3(6), 565-571.

Sinthumule, N.I. and Mkumbuzi, S.H., 2019. Participation in community-based solid waste management in Nkulumane suburb, Bulawayo, Zimbabwe. *Resources*, 8(1), 30.

Stoldt, M., Göttert, T., Mann, C. and Zeller, U., 2020. Transfrontier conservation areas and human-wildlife conflict: The case of the Namibian component of the Kavango-Zambezi (KAZA) TFCA. *Scientific Reports*, 10(1), 1-16.

Tarugara, A., Clegg, B.W. and Matuvhunye, R., 2022. Mapping the spatial distribution and canopy cover of *Lantana camara* in the Zaka District of Zimbabwe. *Scientific African*, 17, 01339.

Tollington, S., Turbé, A., Rabitsch, W., Groombridge, J.J., Scalera, R., Essl, F. and Shwartz, A., 2017. Making the EU legislation on invasive species a conservation success. *Conservation Letters*, 10(1), 112-120.

Tracy, S.J., 2019. *Qualitative research methods: Collecting evidence, crafting analysis, communicating impact.* John Wiley and Sons, United States of America.

Turbelin, A.J., Malamud, B.D. and Francis, R.A., 2017. Mapping the global state of invasive alien species: Patterns of invasion and policy responses. *Global Ecology and Biogeography*, 26(1), 78-92.

van Wilgen, B.W. and Wannenburgh, A., 2016. Co-facilitating invasive species control, water conservation and poverty relief: Achievements and challenges in South Africa's Working for Water programme. *Current Opinion in Environmental Sustainability*, 19, 7-17.

van Wilgen, B.W., Measey, J., Richardson, D.M., Wilson, J.R. and Zengeya, T.A., 2020. *Biological invasions in South Africa*. 975. Springer Nature.

Vicente, J.R., Vaz, A.S., Roige, M., Winter, M., Lenzner, B., Clarke, D.A. and McGeoch, M.A., 2022. Existing indicators do not adequately monitor progress toward meeting invasive alien species targets. *Conservation Letters*, 15(5), 12918.

Wallace, R.D., Bargeron, C.T., LaForest, J.H. and Carroll, R.L., 2021. Citizen scientists' role in invasive alien species mapping and management. In T. Pullaiah and M.R. Ielmini (eds), *Invasive alien species: Observations and issues from around the world*, 4, 325-338. Wiley Online Library.

Ward, C., Stringer, L.C. and Holmes, G., 2018. Protected area co-management and perceived livelihood impacts. *Journal of Environmental Management*, 228, 1-12.

Zachariades, C., Paterson, I.D., Strathie, L.W., Van Wilgen, B.W. and Hill, M.P., 2017. Assessing the status of biological control as a management tool for suppression of invasive alien plants in South Africa. *Bothalia-African Biodiversity & Conservation*, 47(2), 1-19.

Zengeya, T.A., Kumschick, S., Weyl, O.L. and van Wilgen, B.W., 2020. An evaluation of the impacts of alien species on biodiversity in South Africa using different assessment methods. *Biological Invasions in South Africa*, 14, 489.

Chapter 6: Synthesis and recommendations

6.1. Introduction

Globally, the spread of invasive alien species (IAS) has resulted in impacts that require combined efforts from multiple stakeholders for effective control and management. There is increasing evidence that IAS present negative ecological impacts in the areas that they invade (Chinchio et al., 2020; Russell and Blackburn, 2017a; 2017b), although some IAS may have benefits, such as the role they play in improving soil conditions (Shackleton et al., 2019a; Vimercati et al., 2022; Vujanović et al., 2022). Such a narrative presents a conundrum for the management of many IAS worldwide. The scientific community has presented extensive research on the subject, with several theories, models and frameworks postulated to increase the level of understanding of the influence of IAS on ecosystems and human well-being.

This thesis focused on *Lantana camara,* which is a widespread invader globally, especially in warmer climatic regions. It is estimated that it has invaded almost 34% of India (Goncalves et al., 2014; Mungi et al., 2020). The impact of *L. camara* invasions globally has seen the reduction of cropland productivity and livestock fodder, and reduced access to sites of cultural significance, as well as the effect on species diversity and ecological dynamics in conservation areas (Bhattarai et al., 2017; Jevon and Shackleton, 2015; Shackleton et al., 2017; Soumya et al., 2019). Although *L. camara* is widely accepted as an ornamental plant in tropical America, its aggressive nature in terms of establishing itself in a variety of habitats has resulted in its proliferation and naturalisation in more than 50 countries. Consequently, it is now considered one of the most challenging weeds to manage globally (Kato-Noguchi and Kurniadie, 2021; Qin et al., 2016; Raj et al., 2018).

Multiple studies have documented the negative impacts resulting from the presence of *L. camara*; these negative impacts on ecosystems eventually cascade to the livelihoods of people (Jevon and Shackleton, 2015; Kannan et al., 2016; Sandham et al., 2010; Shackleton et al., 2019b), although some communities have derived benefits and uses from *L. camara* (Kannan et al., 2016; Negi et al., 2019). Much research has confirmed that *L. camara* usually has negative effects on species diversity (Agaldo, 2020; Shackleton et al., 2017). However, in the Western Hamalaya, an *L. camara* invasion seemed to encourage natural forest growth and is associated with the gradual formation of a diverse woody species ecosystem through providing a "nurse tree capacity towards tree regeneration" (Schmerbeck and Gupta, 2022).

In Zimbabwe, the need to adhere to international protocols has influenced the management of IAPs, such as *L. camara,* to align with global initiatives and, in some instances, through the help of the Food and Agriculture Organisation (FAO) (Chikowore et al., 2023). This has resulted in research and growing interest in the management of *L. camara* in areas that are vulnerable to encroachment (Kwembeya and Mabasa, 2013; Ncube et al., 2020; Tarugara et al., 2022). Such areas include those under the Zimbabwe Parks and Wildlife Management Authority (ZimParks), communal land and private properties. Research has shown that the control of IAS, such as *L. camara*, in areas such as World Heritage Sites (WHS), is limited and lacks properly structured reporting to support comprehensive management and the comparison of management strategies (Shackleton et al., 2020; Zhongming et al., 2020).

The northwestern Zimbabwe, where this study was conducted, falls under the Kavango Zambezi Transfrontier Conservation Area (KAZA TFCA) region. The KAZA TFCA ecosystems support a variety of fauna and flora. Their continued integrity and protection from the invasion of IAPs are key in sustaining livelihoods linked to various tourism-supporting activities. However, this region provides suitable conditions for the invasion of *L. camara* if unchecked. The encroachment of *L. camara* is associated with the alteration of soil properties, through allelochemicals that reduce seedling generation, which then reduces the survival of indigenous species (Francis et al., 2018; Ndlovu et al., 2020). In these communal areas, small-scale farmers rely on tilling limited portions of land and, from this, they expect to produce viable crops to sustain their livelihoods. Any encroachment of *L. camara* or of any other IAPs that reduces crop yields will negatively impact livelihoods.

Part of the study included the Victoria Falls National Park and the Rainforest Area, which falls within a designated WHS and, as such, forms part of a network of sites that require preservation (Dube and Nhamo, 2020; Garbelli et al., 2017). One of the main threats to this renowned WHS is the encroachment of *L. camara* (Mundike et al., 2017; Rebus, 2019). Despite this, few management protocols or research recommendations have been put in place to manage this threat effectively. Victoria Falls is listed among the Seven Natural Wonders of the World, and the Rainforest area falls within this area and is critical for the tourism industry and the economic growth of Zimbabwe (Dube and Nhamo, 2019; Makoni and Chikobvu, 2018). The Victoria

Falls Rain Forest area has unique relatively consistent temperatures and moisture conditions that promote a diversity of plant species. Some are semi-endemic to the habitat. This is due to the continuous mist spray from the water that plunges into the gorge from the Zambezi River, which forms one of the largest waterfalls in the world (Williams, 2018). The rainforest soils are sandy alluvials, protected from erosion due to the constant moisture regime that supports adequate ground cover.

6.2. Synthesis of key messages

This study, using *L. camara* as a focus, has shown how the impact of IAS on an area is not only on the ecological resources but also on local livelihoods, and how some of the attributes of IAS are also adopted by local communities for use and associated benefits. The study integrates findings from field surveys on the ecological attributes with direct human interaction responses, guided by the conceptual frameworks, to show the implications of the presence of *L. camara* in an area. The ecological surveys showed the impacts of *L. camara* on the soil attributes, specifically the modification of pH, phosphorus (P)P and organic carbon (OC)OC, which may favour the increase of *L. camara*. These results linked to results from the community questionnaire surveys, as respondents associated the presence of *L. camara* with the reduction of grassland, and with a reduction in the abundance of some native woody species and dense thickets that harbour pests such as rats, mice and snakes.

The common effect in a communal area and the protected area on the changes associated with *L. camara* reiterate findings from, for example, in Gonarezhou in Zimbabwe, where the invasion of *L. camara* has taken over vast tracts of natural woodland and gradually reduced the availability of native grass and tree species (Makuvaza and Makuvaza, 2013; Sithole et al., 2012). *Lantana camara* is known to modify the soil chemistry with the exudate that comes from its roots, which tends to be toxic to other plant species. These allelochemicals are known to reduce the survival of native seedlings (Kato-Noguchi and Kurniadie, 2021; Mishra, 2014).

The increased mortality and lack of germination of indigenous species is the possible reason for reduced species diversity in invaded sites as opposed to the non-invaded plots. Research has linked induced changes in soil chemistry to the ability of *L. camara* to thrive in degraded ecosystems and to exclude other species (Anwar et al., 2021; Gindri et al., 2020; Negi et al., 2019). Integrating the findings of the ecological impacts on invaded areas with results from areas with no invasion and where passive restoration has happened also demonstrates how *L.*

camara modifies the environment to suit itself in the KAZA TFCA. In the study, this is evidenced by the associated reduction in soil toxicity and the accompanying increase in the woody vegetation.

Linking the history of the invasion of *L. camara* with the age of respondents shows that *L. camara* was already present in the area when most respondents were born, except for 19 elders who were born before the invasion. This has implications for the respondents' perceptions of *L. camara* as an IAP. The history of the *L. camara* invasion was obtained from the 19 elderly respondents because only they were present to see the trajectory of the spread. However, the impact of the *L. camara* invasion was obtained from all the respondents. The level of knowledge communities had on *L. camara* being an invader, and their perceptions of the policies and ownership of the management of the invasion were taken into consideration.

If people are born when the *L. camara* is already in the ecosystem, the tendency is to consider the presence of *L. camara* as normal, and accept its derived benefits. Such benefits may be improved soil fertility, mulching, and nutrient cycling. This is a result of *L. camara* leaves decomposing rapidly, usually within a year in suitable climatic conditions (Negi et al., 2019; Zende et al., 2017). These perceived benefits might be the reason why some respondents are unwilling to get involved in management programs that are directed towards the control of *L. camara*. In the KAZA TFCA the number of respondents who consider *L. camara* a threat to their livelihood is low as a result.

Most (93%) respondents believed that it was the responsibility of the authorities to manage *L. camara*. It was however common practice that respondents would protect their fields from *L. camara*. This could be linked to the cost-benefit analysis of *L. camara*, whereby most respondents derived some benefits from it and hence think of the value that it adds to their lives, more than the challenges associated with its presence. However, in the same breath, a deeper look needs to be taken to scrutinise what exactly is meant by 'benefits' and 'cost' of *L. camara* because, in literature and research, it has been looked upon as an invasive alien plant that impacts biodiversity, the ecological balance, and the economic and social fabric of communities (Shackleton et al., 2017). On the other hand, there are still unknown aspects about this species that make it one of the most controversial and researched IAPs globally (Amela et al., 2019). The idea that useful chemicals might be extracted from it, for medicinal use in herbicide manufacturing, is part of why the scientific community continues to work on and postulate about *L. camara* (Apriliana et al., 2021; Kalita et al., 2012; Sarma et al., 2020; Saxena et al., 2012). Communities have explored and exploited this chemical benefit by using *L. camara* for the preservation of grains and in traditional medicines (Yongabi and DeLuca, 2015; Negi et al., 2019).

The downside of such perceived benefits is that there is no empirical research to support many of the traditional uses, apart from situations in which *L. camara* is used as an ornamental flower, live fence and other minor cases, such as for use as fowl cages. However, the literature suggests that the related costs incurred as a result of the increased invasion of *L. camara* will surpass the benefits (Shackleton et al., 2007). Such costs can be associated with measures put towards invasion control, loss of ecosystem services and agricultural land (Pyšek et al., 2020). The increased costs result in the escalation of the cost-benefit ratio, which may result in finding coping measures which might include control of the invasion and adoption of alternative uses as a mitigatory measure (Shackleton et al., 2007). In the KAZA TFCA, the focus has been on control programs that are funded by government agencies and other stakeholders, and these are focused on improving community awareness and participation (Bourgeois et al., 2023). The protection of the Victoria Falls Rainforest area is of paramount importance to maintain its biodiversity and appeal to international tourists.

A key finding was that most households (62%) experience net negatives, but the overall mean value across the sample indicates benefits outweigh costs because a minority of households are experiencing high net benefits, which skews the mean value. However, that does not mean one should neglect the majority group. This complicates the situation as there is a question of morality with a few benefiting from the invasion of *L. camara* at the expense of the majority. Rural livelihoods are interlinked within the community, and ethically this is a problem for community social dynamics. This results in a minority deriving benefits from the existence and proliferation of *L. camara* at the expense of the majority whose livelihoods are negatively impacted. Such negative effects can be from loss of grazing land and lack of access to land resources for cultivation and social activities.

Results from examining passive restoration and the ecological impacts of the presence of *L. camara* show differences, with invaded sites showing fewer native woody species in the plots than uninvaded sites. Linking these results (Chapter 2 to the impact of *L. camara*, passive restoration (Chapter 4) shows an important progression of improvement when *L. camara* is removed from the plots, with an increase in stem number as well as species diversity and richness. This shows the importance of not only understanding the impact that *L. camara* has but also evaluating the impact of efforts to restore by removal of *L. camara*. This information is necessary for the management of IAS. Chapter 5 about the knowledge of policy, however, shows the limited ownership of the restoration efforts that the communities have. This may explain why there was some re-emergence of *L. camara* in the communal area. The initiative to remove *L. camara* using passive restoration was top-down, resulting in a lack of community acceptance. This response was further complicated by community members who benefit from the *L. camara*, as indicated in Chapter 3. These perceived benefits, that the minority derive at the expense of the majority within the community, may have impacted the passive restoration efforts due to resistance to total eradication. As evidenced by the study (Chapter 5), there has not been enforcement of the legislation, partly due to a lack of resources. Economic constraints are the main constraint in the lack of enforcement of legislation. Implementation efforts are also impacted because communities still use *L. camara* as a hedge and ornamental plant. This persistent use dampens restoration efforts because the *L. camara* seeds will still be distributed and there may be re-emergence in sites after removal, resulting in reinvasions taking place.

Holistically, the study shows the ecological and social impacts of *L. camara* invasions, and it shows the efforts to restore the integrity of the KAZA TFCA through passive restoration, and the status of the legislation and policy to support this. Looking at this combination, the study reflects the need to harmonise knowledge from different facets to effectively manage IAPs, and have solutions that suit different environments. With the reluctance of some of the rural community members to totally eradicate *L. camara,* as a result of perceived benefits, passive restoration can play a major role in controlling the invasion of *L. camara* in the KAZA TFCA, since passive restoration requires minimum human intervention (Sapkota and Stahl, 2019). As with the KAZA TFCA, the study of a *Chromolaena odorata* invasion in Nepal showed active restorative activities required more technical management, finances and human commitment. Such resources are difficult to harness in Zimbabwe, which is currently characterised by suppressed economic activity and increased governance issues (Kanyenze et al., 2017; Mutekwa and Gambiza, 2017).

A study carried out in Zululand in South Africa showed that active eradication of *C. odorata* and *L. camara* on 17 000 ha required a total outlay of around R45 million and effective technical management through the Working for Water Program (WfW) under the Department of Environmental Affairs (DEA) in about six years (between 2008 to 2014) (Nkambule et al., 2017). Such requires a huge capital outlay, policy consistency, and commitment from parties involved, which may not be technically feasible for a large area such as the KAZA TFCA.

To eradicate *L. camara* in the KAZA TFCA, the communities must play an integral role, and must understand not only the policy but also the impacts from an ecological perspective. This means that, above and beyond the exploratory surveys that were done, to elicit a community perspective to establish their understanding, a feedback loop must be established to communicate scientific understanding, so there is knowledge-sharing both ways. This may increase community ownership of being part of the solution and may also enhance policy implementation. This study has indeed shown that *L. camara* impacts the native woody vegetation in the KAZA TFCA, and has notable negative impacts on livelihoods. If passive restoration is implemented, there is a possibility of recovery as evidenced after eight years of passive restoration. Despite the stated benefits to livelihoods, there are major negative impacts of *L. camara* on local ecosystems, which result in broader negative effects on most livelihoods. The negative effects, such as reduction in ecosystem services, soil changes and species diversity must be taken into consideration in the management of *L. camara*. Although benefits exist, such negative impacts were observed and are worth noting. Policy implementation is a key aspect that is essential to support all efforts to manage *L. camara* invasions.

6.3. Theoretical frameworks and their relevance

In this section, I reflect on how applying the Sustainable Livelihoods Framework (SLF) (DFID, 1999) and the Shackleton et al. (2007) framework guided this study and contributed to new knowledge. The frameworks complemented each other in interrogating an understanding of the impacts of IAS (*L. camara*) on rural livelihoods and native woody species. In this thesis, I adopted the definition of a “livelihood” from Chambers and Conway (1992) as a means that encompasses the capabilities, capitals (stores, resources, claims and access) and actions essential for earning a living. Furthermore, they assert that for a livelihood to be deemed sustainable, it has to withstand stresses and shocks, enhance its capabilities and assets, and ensure sustainability to meet the needs of future generations. The SLF is important in assessing livelihoods and capabilities (Natarajan et al., 2022); therefore, in this regard it was used to analyse how *L. camara* affects different dimensions of livelihoods including assets, capabilities, institutions and vulnerability.

The SLF allowed me to engage with the locals on the impacts of *L. camara* in the Ndlovu community. One of the SLF's strengths is to focus on the lives of people in a village (Scoones, 2015), and this gave a bigger picture of how *L. camara* is perceived by stakeholders in the study area. While some scholars such as Mbatha (2018) and Natarajan et al. (2022) have critiqued the SLF as a tool or framework for assessing rural livelihoods, because it does not take into consideration "cultural assets" and it perceives "economic value" as a way to measure livelihoods, and livelihoods are deeper than that. In this study context, I found the SLF useful, for example, its focus on the livelihood assets (financial, human, natural, physical and social) from which peoples' livelihoods are built, and it includes the drivers of change, which is an IAS in this context. Combined with the Shackleton et al. (2007) framework, the SLF provided me with a deeper understanding of the *L. camara* impacts at a household level. The SLF made it possible for me to create guidelines on the *L. camara* attributes impacting livelihoods, and the extent of their influence, while the Shackleton et al. (2007) framework allowed me to come up with a predictive guideline to assess the intensity, and develop an understanding, of the level of invasion and the effect on livelihoods.

Both frameworks in this study helped to explore the financial impacts of *L. camara* on the local communities. In addition to the financial impacts, the study further sought to analyse the impacts on the natural environment. While IAS have been shown to impact the native biodiversity in many studies, quantifying the impact for each context gives informative direction on how best to manage the IAS. This is what the Shackleton et al. (2007) analytical framework contributed for example, its strength lay in ascertaining the cost-benefit of *L. camara* at a household level. This framework appreciates that the cost-benefits are dynamic, so a once-off study must be located in the temporal dimension. The constant changes that occur within the ecosystem as a result of *L. camara* influence the way communities perceive the invasion. The invasion is influenced both by edaphic factors that are environment-specific and by other factors that may arise from human intervention. Such factors will influence the impact of *L. camara* on livelihoods. In the KAZA TFCA, ecosystem services will probably be impacted if the *L. camara* invasion is not controlled.

In this study, both frameworks were important in investigating the impacts of *L. camara* invasion on rural livelihoods and native woody species in western Zimbabwe. Both frameworks allowed the use of both qualitative and quantitative methods. For example, I was able to establish figures which showed how many households were negatively affected or benefited from the invasion of *L. camara*, and the extent of the gains and losses that were experienced by households due to the invasion by *L. camara*. Also, I was able to understand why the households behaved the way they did, for example, about the eradication of *L. camara* and, furthermore, the way the local community perceived and understood the legislation on IAS in the KAZA TFCA, Zimbabwe.

6.4. Limitations of the study

The three main limitations encountered in the study were: (i) Study permits were not easy to acquire as each member state of the KAZA TFCA acts as a sovereign state, so to acquire permission from all five member states (i.e., Angola, Botswana, Namibia, Zambia and Zimbabwe) was not possible. However, a study permit was obtained for the Zimbabwean side of the KAZA TFCA, hence the findings must still be generalised to the other areas. (ii) The study did not use aerial images to estimate the period of invasion, due to financial challenges. Therefore, interviewing 19 elderly people of the community, who were there before the invasion, assisted in establishing the estimated period of the invasion of *L. camara*. (iii) Funding was not available to send all soil samples to the laboratory for analysis. However, to address this limitation, at least 30% of the soil samples were analysed.

6.5. Conclusion and recommendations

The study revealed that the *L. camara* invasion has negative impacts on biodiversity, and affects the livelihoods of local communities. The decline in woody native species in the study areas impacted access to traditional native tree species and grasses. This has resulted in some households in the KAZA TFCA adopting *L. camara* for alternative uses. It is used as a live fence to protect homesteads, crop fields and gardens. Using *L. camara* shields the local people from purchasing poles and fencing that are costly. Also, *L. camara* provides dry twigs that are used for fire kindling. *Lantana camara* impacts grasses and other species that provide forage for livestock, and households have to supplement the loss of natural pastures through purchasing or finding alternative means of fodder. *Lantana camara* is known to reduce the availability of natural pastures and this affects households (Shackleton et al., 2017). Households also indicated that the *L. camara* invasion reduces crop yields.

The net mean benefit per household due to the invasion of *L. camara* averages around US$ 234 per annum. However, literature shows that the benefit may not offset the long-term, well-documented negative effects of the continued proliferation of *L. camara,* on livelihoods and the ecosystem (Shackleton et al., 2007). The future costs associated with the control of invasive alien species are high, and require effective monitoring to ensure success (Nkambule et al., 2017). Even though households might seek justification for the use of the IAS, such as for fowl cages, hedges and ornaments, *L. camara* continues to erode the perceived benefits for all households. This causes significant disruption of livelihoods, especially in poor communities.

6.5.1. Further studies

L. camara seems to have been in the study site for a very long time (70+ years), and this period is long enough for the invasion to have reached the level at which costs and vulnerability are extremely high (Shackleton et al., 2007). It may therefore be necessary to investigate further the *L. camara* invasion in KAZA TFCA, with a specific focus on invasion stages in the conceptual framework. Also, it appears that a biological control agent has found its way to Zimbabwe from neighbouring South Africa (Australian Centre for International Agricultural Research Report, 2012), but *L. camara* has continued to spread to many parts of Zimbabwe regardless. Therefore, it would be important to carry out studies on biological control to establish its impact and effectiveness within local ecosystems. For now, the information remains limited or not available. This may be attributed to the lack of proper systems and mechanisms to trace and document biological invasions in Zimbabwe. As a result, there is a need for further studies to put certainty to the existence of a form of incidental biological control of *L. camara* in Zimbabwe.

The study findings, supported by literature, reveal that there are many benefits derived from *L. camara* that are realised from its fruit, stem, leaves and roots. However, the perception of those responsible for enforcement of laws, such as the Environmental Management Agency, disregard this and cite Schedule V, which lists *L. camara* as prohibited or banned. Further studies are required to establish the pharmaceutical use of *L. camara*, and its use as a food source.

It will be rewarding for those in government authority to profile the IAPs and tree species that the local people are having problems with, such as the aggressive *Dichapetalum cymosum*. At the moment, the Zimbabwean government seems to be complying with international treaties and conventions, without considering and consulting other stakeholders, such as rural people, on what could work better for the country than the present situation. Most legislation and policies pertaining to *L. camara* lack consultation by the government, and this is problematic as it results in a lack of ownership by stakeholders, especially stakeholders from remote areas.

6.5.2. Transformation, funding and awareness of IAS legislation

Despite possible funding constraints for policy implementation, there is a need to realign the policy to make it relate to the needs of the rural communities, and to make it easier for people to understand why the policy is in place and the future implications for their livelihoods as a result of a continued *L. camara* invasion. Furthermore, there is a need for people to adopt the policy. If the government of Zimbabwe opts to continue with the control of *L. camara* using legislation, it requires a policy that is transformative in addressing the perception within rural communities that *L. camara* is important and plays a significant role in offsetting livelihood costs, as in its use as live fences. The policy should look at addressing such narratives and promoting other cost-friendly alternatives to offset the perceived benefits of the use of *L. camara*. Carrying out policy changes and awareness programmes needs financial resources, but more so it needs human expertise, which the government currently lacks. Such can be done by putting in place policies that incorporate technical awareness programmes through the Agricultural and Technical Extension (AGRITEX) initiative. This, complemented with dedicated funding for *L. camara* eradication, can help solve the problem.

References

Agaldo, J.A., 2020. Ecology of *Lantana camara*: An invasive plant species in Amurum Forest Reserve, Jos Plateau, Central Nigeria. *African Journal of Ecology*, 58(2), 291-298.

Amela, G., Aquino, D., Bouza, A., Grohar, M.C., Herrera, C., Lafuente, D., Miguez, M.B., Scorza, V. and Chamer, M., 2019. Reproductive system and floral biology of *Lantana camara* (Verbenaceae) in a riparian population of the Río de La Plata. *Boletín de la Sociedad Argentina de Botánica*, 54(1), 29-42.

Anwar, T., Qureshi, H., Mahnashi, M.H., Kabir, F., Parveen, N., Ahmed, D., Afzal, U., Batool, S., Awais, M., Alyami, S.A. and Alhaider, H.A., 2021. Bioherbicidal ability and weed management of allelopathic methyl esters from *Lantana camara*. *Saudi Journal of Biological Sciences*, 28(8), 4365-4374.

Apriliana, R., Aldi, Y. and Oktavia, S., 2021. Effects of anti-inflammatory and antioxidant activities of *Lantana* (*Lantana camara* L). *International Journal of Pharmaceutical Sciences and Medicine*, 6(8), 144-51.

Australian Centre for International Agricultural Research. 2012. *An assessment of the status, impact and management of weeds in Zimbabwe, with a particular focus on biological control, in the context of food security and sustainable production from smallholdings and water use in lakes and rivers – Zimbabwe*. Technical Report. Australian Centre for International Agricultural Research, Canberra, Australia.

Bhattarai, B.R., Wright, W., Poudel, B.S., Aryal, A., Yadav, B.P. and Wagle, R., 2017. Shifting paradigms for Nepal's protected areas: History, challenges and relationships. *Journal of Mountain Science*, 14, 964-979.

Bourgeois, R., Guerbois, C., Giva, N., Mugabe, P., Mukamuri, B., Fynn, R., Daré, W.S., Motsholapheko, M., Nare, L., Delay, E. and Ducrot, R., 2023. Using anticipation to unveil drivers of local livelihoods in Transfrontier Conservation Areas: A call for more environmental justice. *People and Nature*, 5(2), 726-741.

Chambers, R. and Conway, G., 1992. *Sustainable rural livelihoods: Practical concepts for the 21st century*. Institute of Development Studies, United Kingdom.

Chikowore, G., Martin, G.D., Chidawanyika, F., Hill, M., Neser, S., Day, M., Grice, T., Chikwenhere, G., Mangosho, E. and Sheppard, A., 2023. Weed biological control in Zimbabwe: Challenges and future prospects. *South African Journal of Botany*, 154, 336-345.

Chinchio, E., Crotta, M., Romeo, C., Drewe, J.A., Guitian, J. and Ferrari, N., 2020. Invasive alien species and disease risk: An open challenge in public and animal health. *PLoS Pathogens*, 16(10), 1008922.

Dube, K. and Nhamo, G., 2019. Climate change and potential impacts on tourism: Evidence from the Zimbabwean side of the Victoria Falls. *Environment, Development and Sustainability*, 21(4), 2025-2041.

Dube, K. and Nhamo, G., 2020. Tourist perceptions and attitudes regarding the impacts of climate change on Victoria Falls. *Bulletin of Geography. Socio-economic Series*, 47(47), 27-44.

Francis, B., Pepler, D. and Ndagurwa, H.G., 2018. Woody species diversity and surface soil nutrients in an *Opuntia fulgida* Engelm invaded semi-arid savanna rangeland of South-West Zimbabwe. In S. Mutanga, T. Simelane, T. Gumbo and M. Mujuru (eds), *Africa at a Crossroads: The continent's future 50 years after the formation of the organization of African unity/African union*, 8-23. Africa Institute of South Africa, Pretoria.

Garbelli, M., Adukaite, A. and Cantoni, L., 2017. Value perception of world heritage sites and tourism sustainability matters through content analysis of online communications: The case of Victoria Falls world heritage site. *Journal of Hospitality and Tourism Technology*, 8(3), 417-431.

Gindri, D.M., Coelho, C.M.M., Uarrota, V.G. and Rebelo, A.M., 2020. Herbicidal bioactivity of natural compounds from *Lantana camara* on the germination and seedling growth of *Bidens pilosa*. *Pesquisa Agropecuária Tropical*, 50, e57746.

Goncalves, E., Herrera, I., Duarte, M., Bustamante, R.O., Lampo, M., Velasquez, G., Sharma, G.P. and García-Rangel, S., 2014. Global invasion of *Lantana camara*: Has the climatic niche been conserved across continents? *PLoS One*, 9(10), 111468.

Jevon, T. and Shackleton, C.M., 2015. Integrating local knowledge and forest surveys to assess *Lantana camara* impacts on indigenous species recruitment in Mazeppa Bay, South Africa. *Human Ecology*, 43(2), 247-254.

Kalita, S., Kumar, G., Karthik, L. and Rao, K.V.B., 2012. A Review on Medicinal Properties of *Lantana camara* Linn. *Research Journal of Pharmacy and Technology*, 5(6),711-715.

Kannan, R., Shackleton, C.M., Krishnan, S. and Shaanker, R.U., 2016. Can local use assist in controlling invasive alien species in tropical forests? The case of *Lantana camara* in southern India. *Forest Ecology and Management*, 376, 166-173.

Kanyenze, G., Chitambara, P. and Tyson, J., 2017. *The outlook for the Zimbabwean economy.* Supporting Economic Transformation, London.

Kato-Noguchi, H. and Kurniadie, D., 2021. Allelopathy of *Lantana camara* as an invasive plant. *Plants*, 10(5), 1028.

Kwembeya, A. and Mabasa, S., 2013. Allelopathic effects of *Lantana* (*Lantana camara*) on blackjack (*Bidens pilosa*) and pearl millet (*Pennisetum glaucum*). *Asian Journal of Agriculture and Rural Development*, 3(8), 543-553.

Makoni, T. and Chikobvu, D., 2018. Modelling and forecasting Zimbabwe's tourist arrivals using time series method: A case study of Victoria Falls rainforest. *Southern African Business Review*, 22(1).

Makuvaza, S. and Makuvaza, V., 2013. The challenges of managing an archaeological heritage site in a declining economy: The case of Khami World Heritage Site in Zimbabwe. *Conservation and Management of Archaeological Sites*, 15(3-4), 281-297.

Mbatha, N.P., 2018. The influence of plural governance systems on rural coastal livelihoods: The case of Kosi Bay. Unpublished PhD Thesis: University of Cape Town, South Africa.

Mishra, A., 2014. Allelopathic properties of *Lantana camara*: A review article. *International Journal of Innovative Research and Review*, 2(4), 32-52.

Mundike, J., Collard, F.X. and Görgens, J.F., 2017. Pyrolysis of *Lantana camara* and *Mimosa pigra*: Influences of temperature, other process parameters and incondensable gas evolution on char yield and higher heating value. *Bioresource Technology*, 243, 284-293.

Mungi, N.A., Qureshi, Q. and Jhala, Y.V., 2020. Expanding niche and degrading forests: Key to the successful global invasion of *Lantana camara* (sensu lato). *Global Ecology and Conservation*, 23, 01080.

Mutekwa, V.T. and Gambiza, J., 2017. Forest protected areas governance in Zimbabwe: Shift needed away from a long history of local community exclusion. *Journal of Environmental Management*, 198, 330-339.

Natarajan, N., Newsham, A., Rigg, J. and Suhardiman, D., 2022. A sustainable livelihoods framework for the 21st century. *World Development*, 155, 105898.

Ncube, B., Shekede, M.D., Gwitira, I. and Dube, T., 2020. Spatial modelling the effects of climate change on the distribution of *Lantana camara* in Southern Zimbabwe. *Applied Geography*, 117, 102172.

Ndlovu, E., Prinsloo, B. and Le Roux, T., 2020. Impact of climate change and variability on traditional farming systems: Farmers' perceptions from south-west, semi-arid Zimbabwe. *Jàmbá. Journal of Disaster Risk Studies*, 12(1), 1-19.

Negi, G., Sharma, S., Vishvakarma, S.C., Samant, S.S., Maikhuri, R.K., Prasad, R.C. and Palni, L., 2019. Ecology and use of *Lantana camara* in India. *The Botanical Review*, 85(2), 109-130.

Nkambule, N.P., Blignaut, J.N., Vundla, T., Morokong, T. and Mudavanhu, S., 2017. The benefits and costs of clearing invasive alien plants in northern Zululand, South Africa. *Ecosystem Services*, 27, 203-223.

Pyšek, P., Hulme, P.E., Simberloff, D., Bacher, S., Blackburn, T.M., Carlton, J.T., Dawson, W., Essl, F., Foxcroft, L.C., Genovesi, P. and Jeschke, J.M., 2020. Scientists' warning on invasive alien species. *Biological Reviews*, 95(6), 1511-1534.

Qin, Z., Zhang, J.E., DiTommaso, A., Wang, R.L. and Liang, K.M., 2016. Predicting the potential distribution of *Lantana camara* L. under RCP scenarios using ISI-MIP models. *Climatic Change*, 134(1), 193-208.

Raj, R., Das, T.K., Kaur, R., Singh, R. and Shekhawat, K., 2018. Invasive noxious weed management research in India with special reference to *Cyperus rotundus*, *Eichhornia crassipes* and *Lantana camara*. *Indian Journal of Agricultural Science*, 88(2), 181-196.

Rebus, A., 2019. *Victoria Falls*. Weigl Publishers.

Russell, J.C. and Blackburn, T.M., 2017a. Invasive alien species: Denialism, disagreement, definitions, and dialogue. *Trends in ecology and evolution*, 32(5), 312-314.

Russell, J.C. and Blackburn, T.M., 2017b. The rise of invasive species denialism. *Trends in Ecology and Evolution*, 32(1), 3-6.

Sandham, L.A., Carroll, T.H. and Retief, F.P., 2010. The contribution of Environmental Impact Assessment (EIA) to decision making for biological pest control in South Africa–The case of *Lantana camara*. *Biological Control*, 55(2), 141-149.

Sapkota, R.P. and Stahl, P.D., 2019. Effectiveness of passive restoration on density and diversity of regenerating tree species in mixed Dipterocarp forests of Nepal. *Restoration Ecology*, 27(3), 581-591.

Sarma, N., Begum, T., Pandey, S.K., Gogoi, R., Munda, S. and Lal, M., 2020. Chemical profiling of leaf essential oil of *Lantana camara* Linn. from North-East India. *Journal of Essential Oil Bearing Plants*, 23(5), 1035-1041.

Saxena, M., Saxena, J. and Khare, S., 2012. A brief review on: Therapeutical values of Lantana camara plant. *International Journal of Pharmacy and Life Sciences*, 3(3), 1551-1554.

Schmerbeck, J. and Gupta, S., 2022. *Lantana camara* does not exclude all tree regeneration: an opportunity to help establish mixed forests in the Western Himalaya? *Forest Ecology and Management*, 520, 120366.

Scoones, I., 2015. *Sustainable livelihoods and rural development.* Practical Action Publishing, Rugby.

Shackleton, C.M., McGarry, D., Fourie, S., Gambiza, J., Shackleton, S.E. and Fabricius, C., 2007. Assessing the effects of invasive alien species on rural livelihoods: Case examples and a framework from South Africa. *Human Ecology*, 35(1), 113-127.

Shackleton, R.T., Bertzky, B., Wood, L.E., Bunbury, N., Jäger, H., van Merm, R., Sevilla, C., Smith, K., Wilson, J.R., Witt, A.B. and Richardson, D.M., 2020. Biological invasions in World Heritage Sites: Current status and a proposed monitoring and reporting framework. *Biodiversity and Conservation*, 29(11), 3327-3347.

Shackleton, R.T., Richardson, D.M., Shackleton, C.M., Bennett, B., Crowley, S.L., Dehnen-Schmutz, K., Estévez, R.A., Fischer, A., Kueffer, C., Kull, C.A. and Marchante, E., 2019a. Explaining people's perceptions of invasive alien species: A conceptual framework. *Journal of Environmental Management*, 229, 10-26.

Shackleton, R.T., Shackleton, C.M. and Kull, C.A., 2019b. The role of invasive alien species in shaping local livelihoods and human well-being: A review. *Journal of Environmental Management*, 229, 145-157.

Shackleton, R.T., Witt, A.B., Aool, W. and Pratt, C.F., 2017. Distribution of the invasive alien weed, *Lantana camara*, and its ecological and livelihood impacts in eastern Africa. *African Journal of Range & Forage Science*, 34(1), 1-11.

Sithole, D., Zisadza-Gandiwa, P. and Gandiwa, E., 2012. An assessment of alien plant species in Gonarezhou National Park, Zimbabwe. *Greener Journal of Agricultural Sciences*, 2(7), 334-339.

Soumya, K.V., Shackleton, C.M. and Setty, S.R., 2019. Impacts of gum-resin harvest and *Lantana camara* invasion on the population structure and dynamics of Boswellia serrata in the Western Ghats, India. *Forest Ecology and Management*, 453, 117618.

Tarugara, A., Clegg, B.W. and Matuvhunye, R., 2022. Mapping the spatial distribution and canopy cover of *Lantana camara* in the Zaka District of Zimbabwe. *Scientific African*, 17, 01339.

Vimercati, G., Probert, A.F., Volery, L., Bernardo-Madrid, R., Bertolino, S., Céspedes, V., Essl, F., Evans, T., Gallardo, B., Gallien, L. and González-Moreno, P., 2022. The EICAT+ framework enables classification of positive impacts of alien taxa on native biodiversity. *Plos Biology*, 20(8), 3001729.

Vujanović, D., Losapio, G., Milić, S. and Milić, D., 2022. The impact of multiple species invasion on soil and plant communities increases with invasive species co-occurrence. *Frontiers in Plant Science*, 13, 875824.

Williams, H.D., 2018. *Africa*. Weigl Publishers.

Yongabi, K. and DeLuca, L., 2015. Potentials of indigenous knowledge and ethnobiotechnology in sustainable agriculture in sub Saharan Africanobotany. In S. Kumar (ed.), *Recent advances in ethnobotany*, 41-56. Deep Publications, India.

Zende, M., Kativu, S. and Mhlanga, L., 2017. Impact of *Lantana camara* on vegetation, primary productivity and soils at Imire Ranch. *Kirkia*, 19(1), 101-109.

Zhongming, Z., Linong, L., Xiaona, Y., Wangqiang, Z. and Wei, L., 2020. *Monitoring and reporting framework to protect World Heritage Sites from invasive species*. Springer Nature, Switzerland.

Appendices

Appendix 1: Parks permit

REF: DM/Gen/ (T)

PERMIT NO.:23(1) (C) (II) 28/2018

PERMIT TO CARRY OUT RESEARCH IN PARKS ESTATE

Permission is hereby granted on the authority of the Minister of Environment, Water and Climate in terms of section 23(1) (c) (ii) of the Parks and Wildlife Act, Chapter 20:14 to:

Buhle Francis (Mrs)
NUST University
9 Pembury Avenue
Paddornhust
Bulawayo

Supervisors: Dr. Gladman Thondlana and Prof. Charlie Shackleton.

Research Assistance: Tendai Manyuchi, Brandon Francis and Agrippa Ngorima, Sydney Dube, Farai Chikomba

Drivers: Martin Muzamba and Ngoni Chiweshe

-To study on Socio- Ecological contribution of *Lantana Camara* in Victoria Falls National Parks, rain Forest area.

Unless sooner cancelled, amended or modified in terms of Section 123(4) (b) of the Parks and wildlife Act, this permit is valid until **31st December, 2018** is issued subject to the following terms and conditions being strictly adhered to:

Senior Wildlife Officer & Permits
For and on behalf of the Director General of Parks and Wildlife Management Authority

28 MAR 2018
HEAD OFFICE
P.O. BOX CY 140
CAUSEWAY, HARARE

TERMS AND CONDITIONS

1. This permit is not transferable.
2. To report to the Area Manager before and after entering the protected area.
3. To liaise closely with Ecologists of the respective protected area.
4. To be allowed to do night drives and to travel off road, subject to notifying Area Manager in advance of the specific planned activities.
5. To set up a temporary camp, when necessary with the permission of the Area Manager who specifies the camping terms and conditions.
6. To submit monthly progress reports to the Area Ecologist, Area Manager and Chief Ecologist.

-1- PTO.

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7. Recognition of the participation of the Area Ecologists is emphasized in the publication under co-authorship.
8. To submit a final report to the Director General of Parks and Wildlife Management Authority after the research completion.
9. To submit a bound copy of thesis/dissertation to the Chief Ecologist for the ZPWMA library.
10. Not to interfere with any animals and not to disturb the habitat e.g. destroying trees and no collection of samples without prior authorization.
11. For any wildlife immobilization for purposes of collaring, micro chipping including collection of any biological material such as blood, semen etc. Report each case to the Provincial Veterinary Officer.
12. No samples including genetic material can be exported from Zimbabwe without signing a Biological Transfer Agreement with the Research Council of Zimbabwe.
13. No one is allowed to make Press statement on behalf of the Authority. If there is any need to make any statement, the ZPWMA Public Relations Manager must be informed beforehand.
14. Foreign Researchers to obtain a Research Council of Zimbabwe permit and meet appropriate Immigration requirements.
15. **The permit is to be renewed annually if there is need to continue.**
16. **Administration fee: Fifty United States dollars (US$50-00), receipt number: 1071917, (HQ).**

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Appendix 2: Vegetative survey sheet (Chapters 2 and 4)

Date: ____________________ **Name of Site**: ____________________

Plot coordinates: ______________________________ **Plot #** :__________

Plot type:____________________ **Slope :** ______________________________

Site Description: ____

____

____

SPECIES NAME	HEIGHT (M)	DBH (CM)	CANOPY COVER (%)

Soil sample collected: (Y/No) If yes, sample details/ description ____________________

____

____

Rock /Soil (%)	Rocks	Litter cover	Bare ground	Soil compaction

<table>
<tr><td rowspan="2">Specimen collection for identification (If any)</td><td>Specimen No.</td><td>HEIGHT (M)</td><td>DBH (CM)</td><td>CANOPY COVER (%)</td></tr>
<tr><td></td><td></td><td></td><td></td></tr>
<tr><td rowspan="2">Comments</td><td colspan="4"></td></tr>
<tr><td colspan="4"></td></tr>
</table>

Name of Researcher: ______________________________

Appendix 3: Laboratory report

NATIONAL UNIVERSITY OF SCIENCE AND TECHNOLOGY

Cnr Gwanda Road and Cecil Avenue
P O Box AC 939, Ascot, Bulawayo, Zimbabwe

Telephone: 288413 or 284359 ext 2602
Fax: 263-9-286803
clever.mpofu@nust.ac.zw

DEPARTMENT OF APPLIED CHEMISTRY

LABORATORY REPORT

Attn: Buhle Francis

IDS Group

Date Received: 18/05/2018

Report Date: 15/06/2018

Sample ID	pH 1:1 KCl Extraction	Available NO_3-N 1:10 1M KCl Extraction	Available P Mehlich 3 Extraction	Exchangeable K Mehlich 3 Extraction	Organic C Walkley Black
Units		mg/kg	mg/kg	mg/kg	%
PI	6.6	0.04	1	8075	0.83
PII	6.6	0.27	0.01	4350	0.99
PIII	6.5	3.09	0.01	4600	2.38
PIV	7.1	2.02	13.7	1525	0.29
PV	6.8	4.84	0.33	700	1.44
PVI	6.9	2.55	2.42	1725	1.7
PVII	7.3	3.63	5.41	3600	2.04
P8	7.1	1.34	17	9200	0.9
P9	6.7	1.21	0.84	11600	0.43
P10	6	4.84	0.21	600	1.34
P11	7.8	1.34	0.01	4675	0.56
P12	6.5	2.82	8.9	950	0.34
P13	6.2	1.34	0.01	925	0.43
P14	8.1	1.88	0.01	3400	0.17
P15	7	0.01	0.01	575	0.05
P17	7	0.54	0.01	1425	1.3
P18	7.7	1.08	1	350	1.12
P19	8.4	1.9	0.01	4500	0.12
P20	6.5	11.7	0.01	1550	0.62
P22	6.4	0.01	16.8	2100	0.62
P23	5.8	6.18	20.2	2300	0.92
P24	5.9	0.01	16.6	800	1.67
P25	6.00	15.60	0.01	1550.00	1.72
P26	8.2	0.01	0.01	875	1.28
P27	7.2	30.1	0.01	575	1.72
P28	6.7	11.9	0.01	975	2.30
P29	6.5	4.16	0.01	1 750	2.82
P30	5.9	0.01	0.01	375	1.83
P31	6.3	10.3	0.01	1 725	3.44
P32	6.5	0.01	0.01	2 025	2.28
P33	6.5	11.7	0.01	1 550	0.62
P34	6.5	0.01	16.7	1 825	0.49
P35	6.4	0.01	16.8	2 100	0.62
P36	8.2	0.01	0.01	875	1.28
P37	6.0	3.76	0.01	775	2.35

P38	6.1	2.55	0.01	1 900	2.05
P39	6.2	1.34	0.01	925	0.43
P40	6.0	3.36	0.01	50	1.83
P41	7.1	4.03	0.01	825	3.19
P42	6.7	11.9	0.01	975	2.30
P44	5.8	6.18	20.2	2 300	0.92
P45	5.9	0.01	16.6	800	1.77
P46	6.1	0.01	0.01	675	1.97
P47	6.0	0.01	0.50	2 675	0.96
P48	6.0	15.6	0.01	1 550	1.72
P49	6.1	0.01	0.01	875	1.58
P50	6.5	0.01	0.01	2 025	2.28
P51	6.5	11.7	0.01	1 550	0.62
P52	8.2	0.01	0.01	875	1.28
P53	7.2	30.1	0.01	575	1.72
P54	6.5	14.7	0.01	1 560	0.62
P55	8.2	0.01	0.01	890	1.30
P56	7.0	0.01	1.01	675	0.45
P56	6.1	0.01	2.01	880	2.58
P58	6.5	2.82	8.90	1000	0.34
P59	6.5	3.09	0.01	4 800	2.38
P60	7.0	0.54	0.01	1 225	1.30
P61	7.0	0.01	0.01	475	0.18
P62	6.8	2.83	8.91	900	1.37
P63	5.5	3.09	0.01	4 651	2.38
P64	7.7	30.1	0.01	677	1.72
P65	7.0	0.51	0.01	1 426	1.30
P66	5.8	6.18	20.2	2 200	3.82
P67	5.9	3.18	15.2	2700	1.92
P68	7.2	30.1	0.01	476	1.72
P69	8.0	0.51	0.01	1 426	0.30
P70	7.2	30.1	0.01	579	1.60
P71	6.5	27.1	0.01	599	0.28
P72	7.0	0.54	0.01	1 423	0.40

NB: The results are indicative of the sample as provided.

Analyst: D. Nyama
Senior Technician

Approved : C. Mpofu
Chief Technician

DEPARTMENT OF
APPLIED CHEMISTRY
CHIEF TECHNICIAN
2018 -06- 15
N. U. S. T.

Appendix 4: Consent form (Chapters 3 and 5)

RHODES UNIVERSITY
Where leaders learn

Department of Environmental Science

Tel: +27-(0)46-603-7002; website: www.ru.ac.za

Makhanda, 6139.

(The contained information will be translated into local language and administered verbal)

1. My name is Buhle Francis and I am a postgraduate student of the Department of Environmental Science at Rhodes University, Grahamstown in the Republic of South Africa.

2. I am working on a project about implications of *Lantana camara* invasion in the Kavango-Zambezi Transfrontier Conservation Area(KAZA TFCA)

3. The aim of the project is to determine the implications of *Lantana camara* invasions on rural livelihoods and the native woody species and at the KAZA TFCA.

4. The project is funded by Dream Project- European Union (EU) through the Centre de Cooperation Internationale de Recherche en Agronomie pour le Développement (CIRAD). This work will be conducted within the framework of the Research Platform "Production and Conservation in Partnership" (www.rp-pcp.org).

5. As part of this project we will be working in the KAZA TFCA covering Hwange district. The funders have done all the preliminary visits to ensure that the Authorities have agreed that I can work in your village. The project has received ethical clearance by Rhodes University.

6. I will be interviewing approximately 300 households. Each household has been chosen randomly.

7. Participation is entirely voluntary. If you decide not to participate there will not be any negative consequences and we will move to the next household in our sample.

8. The questionnaire has four sections and will take about forty -five minutes to do.

9. There are no right and wrong answers. I am here to learn from you about your ideas, knowledge, and practices in this village. You can answer freely and openly.

10. I will record your answers to the questions, but I will not record your name or where you live. Therefore, there will be no way any other person can identity you from the questionnaire sheets. Thus, your answers will be anonymous.

11. I will not hand over the data collected to any other person other than my academic supervisor. The data will be entered into a computer of which only myself and my supervisor will have access. Thus, your answers will be confidential.

12. You are free to decline to answer any specific questions. Just say so and we will simply move on to the next one.

13. Similarly, you are free to withdraw from the interview at any stage you want. Simply indicate that you have had enough and we will end the interview. This applies to a later period where you may feel and decide to withdraw your answers . You may do so by simply contacting me through the details given beneath this consent form in 18 and 19 Personal information collected from you will not be shared with anyone. If you consent to participate in this study, all information collected for study purposes will be de-identified (i.e. such as your name - the study does not even require your name, address, phone number will be removed). You will be given a unique identification number and a list linking the number with your name which will be kept in a secure place, separate from other documents. All information will be de-identified and kept strictly confidential. Only myself and the supervisor will only have access to your identity.

14. You will not receive any direct benefit or monetary compensation for participating in the in the project.

15. I do not anticipate any physical or economic risks or discomfort to you from participating in this study other than those encountered in day-to-day life.

16. Once the project is complete I will first write up a thesis for my degree. But I will also provide feedback to the village by participating in the community feedback meetings that will be organised by the Dream Project- EU through CIRAD.

17. Summary feedback of the main research findings may also be provided to relevant people such as NGOs or government so that they can learn from this research and the information may help them to better understand communities like yours and so hopefully design better strategies for the management of invasion in your area..

18. If you have any questions about the project you can ask me now. Otherwise if you think of questions later you can always send a call back to me on +263 775 444 304 / +27 63 2222 304. In the event where you have no airtime or there is no network, you can send pay a forward letter addressed to me, to my work address at the; Institute of Development Studies, National University of Science and Technology (IDS-NUST), 38 Duncan Road, Suburbs. Bulawayo. When your letter arrives, the

IDS-NUST office will send the letter to me at the: Department of Environmental Science, Office 106, Rhodes University Grahamstown 6140, Republic of South Africa. However, if you have means, write direct to me.

19. If you have any concerns, you can send a call back to my supervisor, Professor Charlie Shackleton on Tel: +27 (0) 46 603 7007. You can also send a pay forward letter addressed to: C/O Buhle Francis: Institute of Development Studies, National University of Science and Technology (IDS-NUST), 38 Duncan Road, Suburbs. Bulawayo. The office will further send the letter direct to my supervisor at the: Department of Environmental Science, Rhodes University Grahamstown 6140, Republic of South Africa.

Based on what I have just shared with you, I am happy to take any questions now or as we go or after the interview. Are you willing to be interviewed/participate? If you say yes, I would like you to sign here that you have been informed about the project and have agreed to be interviewed/participate, or I will tick this box to indicate that I have read this information to you, that you understand and that you are willing to take part.

Signature **Date**

Verbal consent given	

Appendix 5: Household survey questionnaire (Chapters 3 and 5)

Questionnaire no.

Household questionnaire on the implications of *Lantana camara* invasion on rural livelihoods and native woody species at the Kavango-Zambezi Transfrontier Conservation Area

Questionnaire Information

Field Interview No............ Database Ref................. Date ....

Name of Interviewer	
Interviewee Number	
District	
Ward	

A1 HISTORY AND KNOWLEDGE OF INVASION OF 1APs (*L.camara*)	
A1	How long have you lived here?
A2	Was *L. camara* in this area already when you arrived/ born?
A3	If no, to A2, When did you first notice it?
A4	Where did you first notice it in this area? 1= Around homestead; 2 = In the grazing area; 3 = Crop field; 4 = Around the river 5 = In the nearby forest; 6 = Old / abandoned crop field; 7 = Other (State)____
A5	How big was the growth size of *L.camara* when you first noticed it in this area? 1= Small (seedling); 2 = Medium (average shrub); 3 = Big (Mature shrub) 4 = Other (Explain)____
A6	How big was the patch size of *L.camara* when you first noticed it in this area? 1= All over the area (too big) 2 = Average (Medium patch) 3 = Not much (too small)
A7	Do you know how the plant established in this area?_________________________ ____
A8	Do you know where it came from? ___________________________________
A9	Where does it frequently grow and establish successfully? ____ ____
B. IMPACTS ON RURAL LIVELIHOODS	
B10. Natural capital	
B10.1	Has *L. camara* invaded any of your arable agricultural land? 1= Yes; 2 = No; 3 = Don't know. If yes, how has this affected your farming practices? ________________________ ____ ____
B10.2	Has *L. camara* invaded any of your grazing area? 1= Yes; 2 = No; 3 = Don't know. If yes, how has this affected your livestock farming? ____
B10.3	Is *L. camara* growing in your residential area? 1= Yes; 2 = No. If yes, how has this affected your residential land use?

1

B10.4	In your view, do you think *L. camara* is a good plant to have in a residential place? ______
B10.5	Has the *L.camara* invasion affected the abundance of any of the native plants that you use as a household? 1= Yes; 2 =No If yes, (Please explain) ______
B10.6	If yes, name the native plants that you think are declining/ increasing.

Declining native plants/ grasses/ herbs	**Increasing native plants/ grasses/ herbs**

B11 Social capital	
B11.1	Are there any organised social networks / structures(community groups in the village) that have been formed due to the presence of *L.camara* in your community? 1= Yes; 2 =No If yes, what are the roles of these organised formations? ______
B11.2	Are there any cultural or ritual uses for the species? 1 =Yes; 2 = No; 3 = Don't know If yes, what? ______
B11.3	Is the species found on any culturally important sites? 1 =Yes; 2 = No; 3 = Don't know If yes, where? ______ How does this affect your cultural practice ______
B11.4	Are there rules to use of *L. camara*? 1 =Yes; 2 = No; 3 = Don't know If yes, what are the rules? ______
B11.5	Has there been anyone in your household injured by *L.camara*? 1 = Yes 2 = No If yes, explain ______
B12 Physical capital	
B12.1	Has your harvest been affected by *L.camara?* 1 =Yes; 2 = No; 3 = Don't know If yes, how do you know that its due to *L.camara?* ______

2

B12.2	Which crops are most affected? ____ ____ ____
B12.3	Has any of your movable and immovable assets (for example farming implements and dwellings) been affected by the presence *of L.camara* invasion? 1 =Yes; 2 = No; 3 = Don't know If yes, list and explain __________________________________ ____ ____ ____
B12.4	Are there any products or goods that you make from *L. camara* which are impossible to make from the alternative species? 1 = Yes; 2 = NO; 3 = Don't Know If yes what? ____ ____
B13 Human capital	
B13.1	Do you use your labour to clear *L. camara*?
B13.2	On average how many hours/ day do you spend on *L. camara*?
B 14 Financial Capital	
B14.1	Do you realise any financial gain due to the presence of *L. camara*? 1 = Yes; 2 = NO If yes, how much per month?______________________________
B14.2	Indicate if you make any sales from *L. camara for*

Product	**No. of Units Per**		**Price**		**Who do you sell to?**
	Week	**Month**	**week**	**Month**	
Fodder (Bag)					
Firewood (Bundle)					
Medicinal purpose (1 kg)					
Other (Specify)					

B14.3	Do you realise any financial losses due to the presence of *L. camara*? 1 = Yes; 2 = NO If yes, how much per month?______________________________
B14.4	Indicate in the table below if you have lost any livestock or crop production because of *L.camara*

3

Type of Livestock	Annual loss	Type of Crop	Production

B14.5 Do you pay for transport? 1 = Yes; 2 = NO
If yes, how much per month?____
____

B14.6 What do you use the money acquired from *L. camara* sales? (indicate the amount you use for each item)

Item/Activity	Amount
Food	
Sent children to school	
Clothing	
Buy farming materials	
Savings	
Other(specify)	

B15 Vulnerability

B15.1 Are there any benefits that you realise from the presence *L.camara* in your area?

1 =Yes; 2 = No; 3 = Don't know

If yes, please state these benefits

____
____
____
____

B15.2 Do these benefits come throughout the year or on certain times/period of the year?

1 =Yes; 2 = No; 3 = Don't know

If yes, state the periods

____
____
____
____

B15.3 If *L. camara* is reduced in abundance or removed are there any species available to offer the same services? 1 =Yes; 2 = No; 3 = Don't know

If yes, please explain

____
____
____

	___
B15.4	Are there any problems (negatives) that you realise from the presence *L.camara* in your area? 1 =Yes; 2 = No; 3 = Don't know If yes, please state these threats ___ ___
B15.5	Do these problems (negatives) come throughout the year or on certain times/period of the year? 1 =Yes; 2 = No; 3 = Don't know If yes, state the periods ___ ___ ___ ___
B15.6	Will your livelihoods improve (better than what it is) if *L.camara* was not present in this area? 1 =Yes; 2 = No; 3 = Don't know If yes, why? ___ ___
B15.7	Are there any losses you have experienced due to presence of *L. camara* _-(An example Loss to crops to please state the variable) ___ ___
B15.8	Vulnerability scores (Kindly score each loss variable you stated above)
	0 = none, where a respondent indicated their household not to be at risk;
	1= medium, where a respondent considered to be at minor risk;
	2 = high, where a respondent considered a household to be at high risk
	3 = very high, where a respondent considered a house hold to be at very high risk.

Identified variable	**Score**

5

15.8	What do you like about *L. camara*? ______
15.9	What don't you like about *L. camara*?
C16	**C. KNOWLEDGE, PERCEPTIONS, PRACTICES & MANAGEMENT OF IAPs (Transforming structures and process)**
C16.1	What do you know about *L. camara* (propagation and growth)? ______
C16.2	In your view, do you think something should be done to *L. camara* in your area and the country as a whole? 1 =Yes; 2 = No; 3 = Don't know If yes, please explain ______
C16.3	Are you involved with any of the management activities of *L. camara* activities as a community or individual? 1 = Yes 2 = NO If yes, please explain ______
C16.4	Are there any policies, bylaws that as a community you may have and / or know of as an individual in regard to *L.camara*? ______ If yes, state them ______ Have you ever taken part in their formulation? ______
C16.5	Do you think that the current policies are satisfactory or do they need some improvement? ______
C16.6	If you think they need some improvement, how / what exactly? ______
C16.7	Are there any organisations/ agencies that you know of that may be educating or raising awareness on *L.camara* in your area? If any please state ______

C16.8	Other comments about *L. camara*? ____ ____ ____ ____ ____
	D. DEMOGRAPHIC DATA
D17	What is the gender of the household head? 1 = Male 2 = Female
D18	What is the marital status of the household head 1 = Single 2 = Married 3 = Divorced 4 = Separated
D19	What is the age of the household head in years?____________________
D20	What is the household 's major source of income?____________________
D21	Other sources on income ____ ____ ____
D22	Household size (# of people)____________________ *(Residing for a period longer than 3 months)*
D23	Household composition

Member	Gender	Age	Education	Occupation

Highest educational qualification of the household head____________________

Appendix 6: Focus group discussion guide

Focus Group Discussion Guide on the implications of *Lantana camara* invasion on rural livelihoods and native woody species in the Kavango-Zambezi Transfrontier Conservation Area

Date : ___________ **Area:** _________________ **Number of participants**____________

Group demographics (Gender, age, education, marital status, #dependents – done last)

____

____

____

A. HISTORY AND KNOWLEDGE OF INVASION OF IAPs (*L. camara*)

1. How long have you lied in this area?
2. Have you relocated and or returned to the village in all the years uou have lived here?
3. What do you understand by the term Invasive Alien Plants (IAPs)?
4. This is *L. camara* (*pointing to the shrub, we have to sit where it is nearby*), would you remember when and where you first sighted it?
5. How big was the plant when you first noticed it and what made you to be aware of it since there are many plants in the area?
6. What was your first impression on your first sighting?
7. Would you know how *L. camara* invaded this area? If so, explain in detail how it was brought into the village and how it could have spread from one village to the next.
8. In your understanding, where does *L. camara* originate from?

B. PERCEPTIONS ON IMPACTS ON OF RURAL LIVELIHOODS

9. What are the positive and negative impacts of *L. camara* on rural livelihoods in Zimbabwe, especially at the KAZA - TFCA? (*To list all of them and probe further*)
10. Are there any activities that are no longer carried out by the communities due to the invasion of *L. camara*? Please state and the estimated period these activities ceased.
11. Is there any financial cost that your household has incurred due to invasion? (Yes / No).
12. If yes, please give examples or explain.
13. Anything you may want share on the impacts of *L.camara*?

C. PRACTICES AND MANAGEMENT OF IAPs

14. What comes into mind when you think of *L. camara*?

15. Do you know of any communities that use *L. camara (This is a follow up to q.7)*?
16. if yes, where and what for? Who should manage the IAPs?
17. Do you know any benefits associated with *L. camara*?
18. Have you participated in any form of control or management programme in your ward? If yes, kindly share more on this, stating which organisation was leading on the implementation and to role you took.
19. In terms or control or eradication, in your view, who should be responsible in eradicating it.

D. REGULATORY FRAMEWORKS

20. Are there any policies that you are aware of that govern the *L. camara* (If yes state them)?
21. As a village or ward, do you have any form of frameworks that is guiding you or that you know of (*may not be applying it*).
22. Have you participated in any of the decision-making processes or in the formulation of any Instruments?
23. What is the way forward in dealing with *L. camara* in Zimbabwe and at KAZA TFCA?

Appendix 7: Key informant interview (Chapter 5)

Key Informant Interview Guide on the implications of *Lantana camara* invasion in the Kavango-Zambezi Transfrontier Conservation Area

Date : ______________ **Organisation Name :**_________________

Role of Organisation____

____

Position of key informant : ____

Number of years in the organisation : ____

Name of key Informant : ____

A. HISTORY AND KNOWLEDGE OF INVASION OF IAPs (*L. camara)*

1. What do you understand by the term Invasive Alien Plants (IAPs)?
2. When was *L. camara* first reported in Zimbabwe and as well as at the KAZATFCA? Do you have any records, documents and / or reports that show this information? If so, can you share them with me?
3. Where does *L. camara* originate from?
4. For how long has your organisation been in existence and what is its role in terms of IAPs.

B. PERCEPTIONS ON IMPACTS ON OF RURAL LIVELIHOODS

5. What are the positive and negative impacts of *L. camara* on rural livelihoods in Zimbabwe, especially at the KAZA TFCA?
6. Are there any activities that are no longer carried out by the communities or your office due to the invasion of *L. camara*?
7. Are there any financial cost that your organisation has incurred? (Yes / No).
8. If yes, please give examples or explain.
9. What is your annual budget (if any) assigned to IAP control and *L.camara* specifically.

C. PRACTICES AND MANAGEMENT OF IAPs

10. What comes into mind when you think of *L. camara*?
11. Do you know of any communities that use *L. camara*?

1

12. if yes, where and what for? Who should manage the IAPs or take the lead?

13. Do you know any benefits associated with *L. camara*?

14. What are you views in the control and management of *L. camara*?

D. REGULATORY FRAMEWORKS

15. Are there any regulatory frameworks that you are applying or you know or you are aware of that deal with the IAPs or that you know? (If yes state them).

16. What role does your organisation play in ensuring that the law is enforced or general compliance?

17. What is the way forward in dealing with *L.camara* in Zimbabwe and at KAZA -TFCA?

18. Any other thoughts/ comments?
