DISTRIBUTION OF LANTANA CAMARA AND ITS IMPACTS ON SELECTED ECOSYSTEM SERVICES AND LIVELIHOOD IN EAST USAMBARA MOUNTAINS, TANZANIA

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EXTENDED ABSTRACT

This study aimed at assessing the distribution of *L. camara* and its fractional cover using very high-resolution satellite image (Worldview-3) complemented by field data from plots surveys. Since *L. camara* co-exist with several other invasive alien plants species in East Usambara it is not easy to detect it. Therefore, it was necessary to find ways to differentiate it from the other species using remote sensing. Field Spectrometer was used to obtain spectral data of eighteen invasive plants occurring in East Usambara and thereafter assessing suitable bands that can be used to detect *L. camara* from the available sensors currently but also future satellite missions.

Results suggest that when using Worldview 2 the Near Infra-Red 2 (NIR2) (860-1040 nm) is the best band for *L. camara* detection while for Sentinel 2 the Short Wave Infra-Red (SWIR) Cirrus (1360-1390 nm) is more suitable. Moreover, hyperspectral EnMAP sensor can differentiate the species best in the Visible NIR (423 nm). Evidence from Random Forest model using all bands from 350-2500nm suggest that, the visible ranging 350-383nm is the best region for differentiation of the species with band importance of 0.8-1. With the absence of this range of bands in the visible, the SWIR bands ranging 1778-1790nm can be used to differentiate the species having band importance of 0.5-0.7. Gradient Boosted Machine model showed that SWIR band at 1790 nm was the best in the separation of the invasive species with band importance of 0.4. It is therefore important that while choosing images for detection of similar species it should contain these wavelength ranges. It is also important for future satellite missions to include these wavebands so as similar species as the 18 in this study can be detected and differentiated.

Furthermore, the study used the Worldview 3 Satellite image to map L. camara in East Usambara utilizing the NIR 2 texture properties of the image as was found to be the best band in differentiation of the species. *Lantana camara* in East Usambara was detected along the roads and forest edges with low abundance, while it was more abundant in agriculture lands and abandoned farms and tea plantations. Only 0.5% of the 11% invaded areas had *L. camara* abundance above 50% while 8.8% of invaded areas have *L. camara* abundance of less than 25%.

Presence of *L. camara* affects ecosystem services and livelihood of the inhabitants. East Usambara inhabitants benefit from the ecosystem through provisioning of wild plant species for medicine, fuelwood and construction among others. They benefit also through crop cultivation and honey production which are among major economic activities in the area. Results show that 67% of the respondents perceive *L. camara* to cause a decline in varieties of wild plant species and 50% responded that it caused reduction in honey production which affects their livelihood. The study also found that *L. camara* reduced the growth of maize by 29%, while cassava was not affected. The reduction of maize growth is only caused when growing simultaneously with maize as there was no evidence of allelopathic effects of *L. camara*. Similarly, Inoculation of autoclaved soil with microorganisms from invaded soils increased biomass of cassava and reduced the growth of maize. Therefore, removal of *L. camara* will immediately mitigate its negative impacts on maize.

The livelihood of the inhabitants depending on crop production is not affected directly by *L. camara* abundance rather it is affected significantly by increasing cost of farm maintenance which includes weeding. It is estimation that about 66.62 Million Tanzanian shillings is required to clear about 1277 hectares of *L. camara* invasion in East Usambara. It is recommended that *L. camara* should be managed while it is still not very dense and widely spread. The cost of managing will increase substantially with further invasion and mechanical means may not suffice.

DECLARATION

I, Amina A. Hamad declare to Senate of Sokoine University of Agriculture that, this thesis is my own original work and that it has neither been submitted nor being concurrently submitted in any other Institution.

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LIST OF ABBREVIATIONS AND SYMBOLS

AVIRIS	Airborne Visible Infrared Imaging Spectrometer
BRT	Boosted Regression Trees
EnMAP	Environmental Mapping and Analysis Program
GBM	Gradient Boosted Machine
IAP	Invasive Alien Plants
IAPS	Invasive Alien Plant Species
IAS	Invasive Alien Species
MNRT	Ministry of Natural Resources and Tourism
MODIS	Moderate Resolution Imaging Spectroradiometer
NDVI	Normalized Difference Vegetation Index
NIR	Near Infrared
NRI	Normal Ratio Index
OBIA	Object Based Image Analysis
PCA	Principal Component Analysis
RF	Random Forest
SDM	Species Distribution Model
SRTM	Shuttle Radar Topography Mission
SVM	Support Vector Machine
SWIR	Shortwave Infrared
VHR	Very High Resolution

CHAPTER ONE

1.0 GENERAL INTRODUCTION

1.1 Background Information

There is increasing concern worldwide by biologists, ecologists and conservationists about the negative impacts caused by invasive alien species (IAS) in both natural and agro-ecosystems (Witt, 2010; Obiri, 2011; Shackleton *et al.,* 2014). Specifically, IAS are known to affect among others biodiversity of an area by suppressing growth of many indigenous plant species (Van Wilgen *et al.,* 2008; Pejchar and Mooney, 2009); loss of agricultural productivity by occupying agricultural lands and suppressing growth of crops (Romel *et al.,* 2007a; Thapa *et al.*, 2014); and loss of other ecological services such as river flows (water quality and quantities) and soil nutrients (Simba *et al.*, 2013; Le Maitre *et al.,* 2015). These losses together with efforts to control IAS costs US$33.16billion in USA; US$16.3 billion in Canada and US$ 0.84 billion in South Africa annually (Pimentel *et al.,* 2005*; Colautti et al.,* 2006; de Lange and Van Wilgen, 2010). The cost of IAS in Africa as a whole and specifically in Tanzania is unknown. Since the survival of human beings is closely linked with nature, efforts to combat this problem are important.

Lantana camara is a multi-stem shrub native to South America and a major weed in tropical and sub-tropical countries (Taylor *et al.,* 2011; Sundaram and Hiremath, 2012). It is among the worst ten invaders in the world (Lowe *et al.,* 2000; Goyal and Sharma, 2015). It is believed that *L. camara* was imported to Europe from America in mid-16th and 17th century where it underwent breeding which resulted into hundreds of hybrids. The species were then exported to India, Australia and Africa in mid-19th century. With time they escaped from their originally planted areas into unintended areas and became weeds (Goyal and Sharma, 2015).

The East Usambara is known for the agroforestry practice mainly growing cardamom, cloves and black pepper since they need trees to nurse them. The cultivation of such spices has become unprofitable hence; land was cleared for annual crops such as sugarcane, cassava and maize. This shift played part in the increase of *L. camara* invasion (Hulme *et al.,* 2013; Reyes *et al.,* 2006). Moreover, the soils in the sub-montane forests of East Usambara are acidic and do not favor most crops except tea which can withstand acidic soils. Tea prospers well also since it is permanent crop that keeps soil intact and forms thick canopy that prevents soil loss from runoff and erosion hence retains the organic matter in the soil (Hamilton and Bensted-Smith, 1989a). The abandonment of annual crops due to unfavorable soil conditions has led to the rapid invasion of *L. camara* in East Usambara (Reyes *et al.,* 2006).

According to Mandal and Joshi (2014), *L. camara* is invasive and has negative impacts on biodiversity and crop production, even though it shows potential in forest carbon sequestration. Invasion of understory shrubs like *L. camara* has been associated with reduction in growth of overstorey woody trees. An example is an understory shrub *Lonicera maackii* which showed 16% reduction in basal area growth at invaded areas compared to non-invaded areas in USA (Hartman and McCarthy, 2007). This eventually impacts the overall ability of the forest to store carbon (Mandal and Joshi, 2014).

Managing invasive species requires knowledge of their distribution in space and time. Traditional field inventories and assessment of invasive species have proved to be expensive. They require a lot of man power and takes long time to accomplish (Taylor *et al.,* 2011). On the other hand use of Remote sensing data has proved to be efficient in assessing invasive alien species, a larger area can be assessed in a short time (Taylor *et al.,* 2011). Improved spatial resolution of satellite images has made it easier and more efficient to study species like *L. camara*, which invade in small patches of less than 16m^2. The Worldview 3 (WV3) imagery which has 8 multi-spectral band at 2m spatial resolution and 0.5 panchromatic band has efficiently been used to assess invasive species in forest landscape (Taylor *et al.,* 2011; Cho *et al.,* 2015; Robinson *et al.,* 2016). The high spatial resolution World-View 3 imagery has potential to improve and contribute in different methodologies for ecological monitoring of biodiversity (Dalmayne *et al.,* 2013). With different vegetation indices derived from satellite data such as EVI, NRI, NDVI, etc. detection of invasive species, biomass estimation and carbon can be linked with remote sensing data (Eckert, 2012). The additional bands in the WV3 as compared to other high spatial resolution images like Quickbird and Geoeye-1 gives it the extra ability to model different plant species (Robinson *et al.,* 2016a).

Species Distribution Models (SDMs) have been used to model the distribution of invasive species as well as predicting future invasion. They use concept from ecological and natural science combined with statistical and information technologies (Elith and Leathwick, 2009). Many models are available and have been used such as Maximum Entropy model (Max-Ent), logistic regression, Random Forest and Gradient Boosted Machine (Shiferaw *et al.,* 2019a, 2019b). While it has been proved that individual models have potential in prediction of invasive species distribution, studies show that combining different modelling techniques yield better results. Ensemble species distribution model is one of the example of a model that combines strength of different models while minimizing the weaknesses of any of the model (Stohlgren *et al.,* 2010). Using multi-source Remote Sensing data (DEM, Multispectral images, Climatic data etc.) also improves accuracy for land cover classification using different classification algorithms (Pouteau *et al.,* 2011).

Understanding the costs and associated impacts caused by IAS requires both the intensive assessment (abundance) and extensive assessment (spread) of the invasion. Early detection of potentially IAS is crucial, because the expenses and effort required in halting and managing invasions increases considerably as the species spreads and becomes more abundant, until removal is impossible and control becomes very costly (Van Wilgen and De Lange, 2011). FAO (2005) suggested that one of the important steps in prevention is identification of the possible susceptible sites in which the species would spread into and the pathways in which they can be introduced to those areas. This can be achieved through the use of remote sensing techniques. Remote sensing techniques utilize the information from satellite images in detection and mapping the distribution of invasive alien plants (Cho-ying Huang, 2009).

Appropriate spatial resolution is crucial when using remote sensing techniques to evaluating the distribution and abundance of alien plant invasions (Collingham *et al.*, 2000; Gunawardena *et al.*, 2014). With the currently available very high spatial resolution satellite data there is a potential of getting more accurate information on the spread and impacts of invasive alien plants (Robinson *et al.*, 2016a).

1.2 Problem Statement and Justification of the Study

1.2.1 Problem statement

East Usambara forests, one of the 35 global hotspots is located in Tanga region, north east Tanzania (Mittermeier *et al.*, 2011). The forests are known as the most important forest block in Africa due to high level of biodiversity (plants, invertebrates and birds) and endemic species it contains (Burgess *et al.*, 2007a). The East Usambara forests also support large commercial tea farms, forest plantations as well as small scale cultivation of other crops particularly spices such as cardamom, cinnamon, cloves and black pepper in the enclaves (Hulme *et al.,* 2013). Likewise the East Usambara forests protect and is the source of water supply for more than 200, 000 people in the nearby town of Tanga (Fowler and Nyambo, 1995).

Despite all these benefits, the biodiversity in these forests are threatened by increasing pressure of land from the nearby communities, habitat change due to clearance for subsistence and cash crop cultivation and spread of IAS (Sheil, 1994; Dawson *et al.*, 2008a; Conte, 2010). Invasive alien plants are becoming a big threat not only to the forest reserves but also to the agriculture lands within the vicinity of the forest blocks (Hulme *et al.,* 2013). Early detection of potentially invasive species is crucial in this case, because the expenses and effort required in halting and managing invasions increases considerably as the species spreads and becomes more abundant (Pattison *et al.*, 2011).

Many studies in the East Usambara forests particularly in one large block of Amani Nature Reserve have focused on the ecology, spread and management of some of the invasive alien plant such as *Maesopsis eminii* (Binggeli and Hamilton, 1993); *Cordia alliodora* (Edward *et al.,* 2009) and *Castilla elastica* (Mbwambo, 2009). Dawson *et al.* (2008, 2009), reported 44 species one of which being *Lantana camara L.* which was introduced in Amani Nature Reserve and now have naturalized. These species have the potential to spread to other areas. However, little is known about the extent of spread of these species and other potential invasive species in the entire forests of East Usambara Mountains. Fowler and Nyambo (1996) observed dense infestations of *L. camara* only along the edges of trucks and in other highly disturbed areas and recommended that *L. camara* did not appear to penetrate into intact canopy forest, but there was little evidence that it could colonize natural forest gaps to any great extent. Studies elsewhere indicates that *L. camara* produce biochemical that affect the germination, growth and survival of other plant species (Sharma *et al.,* 2005; Romel *et al.,* 2007a). Moreover *L. camara* was found to affect economic viability of 14 major crops around the world including coffee, tea, rice, cotton, oil palm, coconut and sugarcane (Romel *et al.*, 2007a). The competitive ability of *L. camara* is posing a serious threat to crop productivity hence suggests immediate action has to be taken now.

Given this wide range of invasive plant species, there is a need for getting spatial information on interspecies level accurately for conservation of the native biodiversity and managing the land ecologically. The spatial information is not only useful as an indicator of change but also provide information about the nature and direction of change, which can indicate where and what damage is expected (Foxcroft *et al.,* 2009).

Studies to map distribution and impacts of *L. camara* have been done in other regions. However, the approaches developed can only be used in deciduous forests where *L. camara* can be detected using satellite images acquired during dry season. East Usambara is comprised of an evergreen forest and perennial crops therefore it is very difficult to detect *L. camara*. This study therefore aimed at developing methodology to assess the distribution of *L. camara* and its impact on selected ecosystem services and livelihood in the East Usambara mountains using very high resolution satellite image (Worldview-3) complemented by field data from plots survey (Kimothi and Dasari, 2010; Prasad, 2010a).

1.2.2 Justification of the study

Apart from South Africa most African countries particularly Tanzania are understudied with respects to invasive alien plants. This creates a geographical bias in understanding plant invasion of a specific habitat (Pysek *et al.,* 2008). Therefore, this research tried to reduce this bias by adding knowledge on the extent of spread and impacts of invasive alien plant *L. camara* in the mountain forest of Tanzania. Moreover, the spectral archive of invasive alien plants in East Usambara established in this research will benefit other researchers and scientists working on similar species. The archive will provide the spectral information of each species therefore creating baseline information for detection, differentiation and spatial assessment of these species using remote sensing techniques (Taylor *et al.,* 2011). Knowing the distribution and impacts of *L. camara* on biodiversity and ecosystem services is essential in planning and managing invasive alien plants (Van Wilgen *et al.*, 2008; Foxcroft *et al.,* 2009) . For that reason, findings from this research will assist conservators, forest and agriculture managers both at regional and national level in setting priorities for effective management and monitoring of this noxious weed.

1.3 Research Objectives

1.3.1 Overall objective

The general objective of the study was to assess the spatial distribution of *L. camara* and its impacts on selected ecosystem services and livelihood in East Usambara Mountains, Tanzania.

1.3.2 Specific objectives

The specific objectives of the study were:

i. To evaluate optimum spectral ranges for identification of a range of invasive alien plant species in East Usambara mountain
ii. To assess the current distribution and fractional cover of *L. camara* in East Usambara Mountains using Very High-Resolution satellite image
iii. To assess impacts of *L. camara* on selected ecosystem services and livelihoods

1.4 Conceptual Framework

Lantana camara changes habitat characteristics of the particular ecosystem in which it has invaded. It changes forest and agriculture habitats by replacing native species or hindering growth of other species (El-Keblawy and Al-Rawai, 2007). This in the end affects the ecosystem functioning and how it provides ecosystem services (Agricultural yield, provision of utilizable plants) (Bhatt *et al.,* 1994; Romel *et al.,* 2007a). Therefore, there is a need to monitor and assess the extent of *L. camara* invasion and its impacts in the ecosystem. The assessment needs to be extensive in order to be informative hence a need for spatial assessment. Spatial assessment can be achieved by employing different remote sensing and GIS techniques in interpreting and analysing satellite data (Kimothi and Dasari, 2010; Lei Ma *et al.,* 2015). These techniques utilize information from satellite data in detecting the distribution of the invasive species and their impacts. Currently, it has made possible to detect an individual tree canopy using recent improved satellite sensors that provides very high spatial resolution images (Abd-Latif *et al.,* 2012). Moreover, satellite images provides information about the ecosystem at a larger scale remotely (Cho-ying Huang, 2009). However, to obtain meaningful information about a particular ecosystem there is a need for interpretation of the satellite data with the support of field data. This is achieved by relating the spectral information from satellite data with the presence/absence of *L. camara*, abundance of other species and ecosystem services. Figure 1.1 presents an overview of the linked theories and variables.

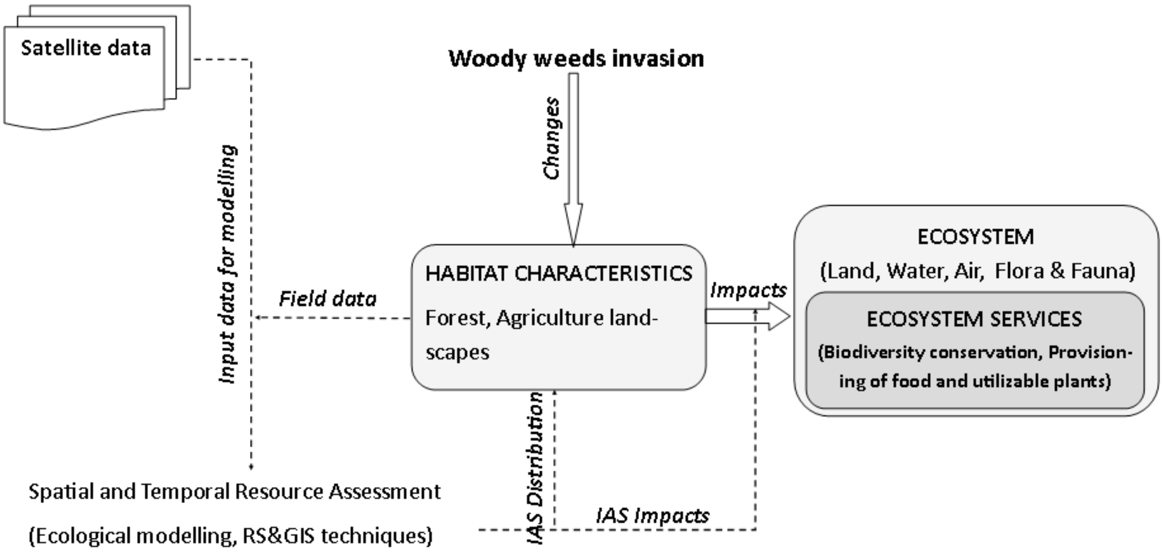

Figure 1.1: Conceptual framework for assessing distribution of *L. camara* and its impacts of ecosystem services and livelihood in East Usambara.

1.5 Thesis structure

This thesis consists of five chapters. Chapter one presents the general introduction covering the background on invasive alien plant species globally and in Tanzania, problem statement and justification, objectives and conceptual model which shows the link between objectives and variables. Chapter two presents the manuscript one. It describes how field spectroradiometry can be used to identify sensor bands and bandwidths suitable for invasive species discrimination. Based on field spectra measurement the study provided spectral library of 18 invasive species and evaluated optimum bands for species discrimination. Chapter three presents the second manuscript on modelling of the distribution and fractional cover of *L. camara* in East Usambara using very high spatial resolution satellite image (Worldview 3) and field data. Fractional cover of *L. camara* was then linked with the digitized farm plots and household surveys to evaluate impacts on livelihood. Chapter four presents the third manuscript on the impacts of *L. camara* on growth of maize and cassava and the mechanisms in which these impacts arise. Chapter six presents the key contributions of the study, the areas for further studies and specific recommendations for central, local government and NGO on the management and control of invasive alien plant species *L. camara* in particular.

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CHAPTER TWO

2.0 OPTIMAL SPECTRAL RANGES FOR THE IDENTIFICATION OF A RANGE OF INVASIVE ALIEN PLANT SPECIES USING FIELD SPECTRORADIOMETRY

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Abstract

Several alien plant species have naturalized, expanded its range and subsequently became invasive in East Usambara. To successfully manage the alien invasion, it is vital to map the distribution of these species. Especially given the wide range of invasive alien plant species present in the area, discrimination on interspecies level is an important key. This study presents a spectral library to facilitate mapping efforts, by providing unique spectral profiles for a large selection of invasive alien plant species in East Usambara. This library is developed as a baseline for remote sensing projects and is made available free of charge. Eighteen different invaders established outside the botanical garden were sampled using a PSR-2500. Measurements were carried out in a controlled laboratory environment and the samples size was set to 16 trees per species, stratified over a minimum of eight different species throughout the study area. The spectra resampled to Worldview 2-8band, Sentinel2 and EnMAP sensors were tested to identify optimum bands for specie discrimination. Findings show that the optimum bands for specie discrimination using Worldview 2 is the NIR2 (860-1 040 nm) while for Sentinel 2 is the SWIR Cirrus (1 360-1 390 nm). Moreover, EnMAP can differentiate the species best in the Visible NIR (423 nm). Furthermore, the spectra were analysed to evaluate how the properties of a sensor should be to better discriminate the species using Random Forest and Gradient Boosted Machine models. Evidence from Random Forest and Gradient Boosted Machine models show that wavelength ranging 350-448 nm in the visible range has the highest importance in the separation of the 18 species. Findings suggest that different sensor have different bands that can differentiate the species best. It is therefore important that while choosing images for detection of the similar species it should contain these wavelength ranges. It is also important for future satellite missions to include these wavebands so as similar species as the 18 in this study can be detected and differentiated.

2.1 Introduction

Alien invasive species(IAS) has become a concern to biologists, conservationist and ecologist worldwide due to its various impacts on ecosystems (Obiri, 2011; Shackleton *et al.*, 2014; Witt, 2010). Apart from affecting biodiversity in forest landscapes (van Wilgen *et al.,* 2008; Pejchar and Mooney, 2009), they are known for affecting negatively the agriculture and livestock productivity (Dalle *et al.*, 2006; Romel *et al.,* 2007b). Moreover, ecological services such as water flows and soil nutrients are also affected by invasion of alien plants (Le Maitre *et al.,* 2015; Simba *et al.*, 2013). Early detection of potentially invasive species is crucial in this case, because the expenses and effort required in halting and managing invasions increases considerably as the species spreads and becomes more abundant (Pattison *et al.*, 2011).

A number of exotic species have naturalized, expanded their range and subsequently became invasive in East Usambara forests in northeast Tanzania. Dawson *et al.,* 2008; 2009, reported a total of 44 plant species among those were introduced in Amani botanical garden have now naturalized. 32 species are recorded to be spreading, that is they have spread widely and far away from their original planting locations (Dawson *et al.*, 2008b). Given this wide range of invasive plant species, there is a need for getting spatial information on interspecies level accurately for conservation of the native biodiversity and managing the land ecologically. The spatial information is not only useful as an indicator of change but also provide information about the nature and direction of change, which can indicate where and what damage is expected (Foxcroft *et al.,* 2009). Spread of the invasive species is influenced by different environmental factors that can be evaluated by knowing their spatial distribution. Therefore, there is a greater need for a technique that can distinctively differentiate plants species and map their distribution.

Current techniques of mapping invasive species using multispectral data may not be sufficient either due to low spatial and spectral resolution of most sensors. A Landsat image of 30m spatial resolution may contain pixels with mixed plant species in areas where invasion is in small patches as well as for species that co-exist in similar environment. Increasing the spatial resolution reduces the errors caused by mixed pixels and works well where species are distributed in patches. The use of very high resolution (VHR) satellite data such as Worldview 2, Geoeye has been proved effective in detecting and monitoring tree species (Abd Latif *et al.,* 2012; Cho *et al.,* 2015). Mapping at specie level in areas with multiple invasions even very high spatial resolution satellite data may not provide sufficient information.

Advances in hyperspectral airborne imaging sensors such as AVIRIS (http://aviris.jpl.nasa.gov) and HYMAP (http://www.hyvista.com/) together with spaceborne sensor such as Hyperion (https://eo1.usgs.gov), provide a way to improve assessment of invasive species using remote sensing. The narrow bandwidth of hyperspectral data provides more information about the finest spectral features of plants species as compared to broad bands of multispectral (S. Taylor *et al.,* 2011b). However, these hyperspectral sensors are currently not available. The planned future hyperspectral mission such as EnMAP (https://www.enmap.org) will provide opportunity for better plant species distribution (Schwieder *et al.,* 2014).

Various vegetation and land use studies have demonstrated the increase use of hyperspectral data (Vaiphasa *et al.,* 2005; S. Taylor *et al.,* 2011b; Ballanti *et al.,* 2016). In particular the use of field hyperspectral measurements in discriminating the spectral signatures of different species in a mixed community (Abbasi *et al.,* 2008; Ouyang *et al.,* 2013; Jiménez and Díaz-Delgado, 2015). Filed spectral measurements has been used to develop spectral libraries such as shrub and mangrove database (Jiménez and Díaz-Delgado, 2015; Kumar *et al.*, 2015), wetlands and land use (Zomer *et al.*, 2009). The high dimensionality of these datasets needed new and specialized approaches that can deal with multi-collinear datasets (Lucas and Carter, 2008; Zomer *et al.*, 2009; Clark and Roberts, 2012; Einzmann *et al.*, 2014). Some studies have used spectral similarity and distance measures to differentiate plant species or vegetation types based on spectral extracted from images (Ahmad *et al.*, 2011; Taylor *et al.*, 2012). However, these techniques can also be applied to field spectral measurements for species monitoring.

Machine learning algorithms such as Random Forest (RF), Support Vector Machine (SVM) and Gradient Boosted Machine (GBM) have also shown to perform adequately in various studies (Foody and Mathur, 2004; Freeman *et al.*, 2015; Berhane *et al.*, 2018; Shiferaw *et al.*, 2019a). They have proved to be robust and have shown reliable regression and classification results (Shiferaw *et al.*, 2019a).

Therefore, the objectives of this study are to a) build a spectral library of trees and shrubs that have become invasive in Amani Nature Reserve, b) determine which spectral bands of currently active sensors in orbit are most suitable to differentiate these species, and c) to determine which wavelength ranges discriminate the species best and advice on how a sensor should be characterized to be able to differentiate the species. The open access spectral library together with the findings of this study will provide a baseline guiding researchers in successfully identifying and differentiating these invasive alien plant species and allow for a more accurate mapping and monitoring of the further spread of these species in this unique biodiversity hotspot.

2.2 Material and Methods

2.2.1 Study area

East Usambara mountain forests, one of the 35 global hotspots is located in Muheza district in Tanga region, north east Tanzania (Mittermeier *et al.*, 2011). The forests are known to be one of the most important forest block in Africa due to high level of biodiversity (plants, invertebrates and birds) and endemic species it contains (Burgess *et al.*, 2007b). The forest is a network of 18 forest blocks covering an area of 263 km^2 of which 8 380ha forms Amani Nature Reserve (Amani N.R.) the biggest block (Burgess et al., 2007b). Amani N.R. lies between 300 to 1 128 m above sea level where the sampling of trees took place. Figure 2.1 shows the study area and sampling locations.

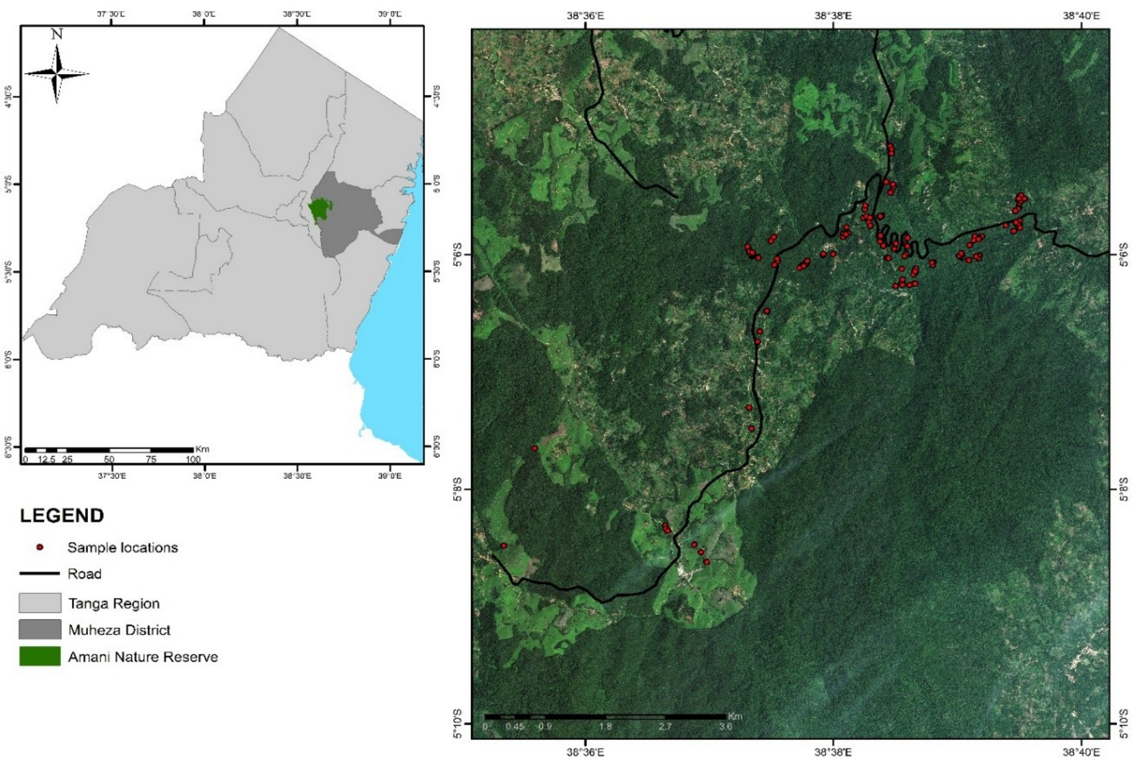

Figure 2.1: Overview map and zoom map of the study area location in Tanga region, Tanzania. In the zoom map, a Worldview 3 true colour image illustrates the study area's land cover with the tree sampling locations highlighted in red.

2.2.2 Data collection

Spectral Evolution PSR-2500 portable spectroradiometer was used to collect spectral data within the 350-2500 nm wavelength range. A number of studies have shown good results using the PSR-2500 for the discrimination of plant species (Vaiphasa *et al.,* 2005; Lucas and Carter, 2008; Clark and Roberts, 2012; Dubula *et al.,* 2016; Maimaitiyiming *et al.,* 2016). A contact probe was used to measure the spectral reflectance in a laboratory environment. Contact probe is described as a reliable method for collecting spectral data as compared to ex-situ measurements (Einzmann *et al.,* 2014). The delay between removing the samples from the trees and the ex-situ measurement has minimal effect of the data, especially when the samples are refrigerated. To overcome sampling issues caused by weather (e.g. precipitation) and growing form of the target species (e.g. >20 m tall trees), similar sampling approach was adopted, as this would ensure independent and continuous data collection. Data was collected in October, 2016.

A field sampling technique used was designed to ensure high quality field sample collection, repeatability and comprehensive documentation. A total of 32 sites were spotted containing mix of the target invasive species and their gps coordinates were recorded. Due to limited time and resources, sampling for the 18 target species (Table 2.1), was done using between 16 and 20 individual plant/tree. The 18 species were chosen ranging from highly invasive to low invasive as well as if it is spreading or naturalized based on Dawson *et al.* (2009). The samples were collected from eight to ten different sites well distributed over the study area (Figure 2.1). Each sampled tree was photographed on site and GPS coordinates taken. Two branches from different sun lit angles were collected from each tree, labelled, refrigerated and brought to the lab within four hours. Five leaves per branch were taken and stacked together upon a non-reflective black surface. Three measurements were made per leaf with contact probe while rotating through the stack making sure that the stack always consisted of five leaves.

Table 2.1: Alien woody invasive species found at the study area and admitted to the spectral library, showing their invasion status and risk

Spp no.	Spp. ID	Spp. Name	Invasion status	Invasion risk
1	AC	*Albizia chainensis* (Osbeck) Merr.	Naturalized	Low
2	AP	*Arenga pinnata* (Wurmb) Merr.	Spreading	High
3	CA	*Cordia alliodora* (Ruiz. & Pav.) Cham.	Spreading	Low
4	CC	*Cinnamomum camphora* (L) J. Presl.	Naturalized	Low
5	CE	*Castilla elastica* Cerv.	Spreading	High
6	CH	*Clidemia hirta* (L.) D. Don.	Spreading	High
7	CJ	*Cryptomeria japonica* (L. f.) D. Don	Naturalized	Low
8	CO	*Cedrela odorata* L.	Spreading	High
9	EG	*Elaeis guineensis* Jacq.	Spreading	High
10	HB	*Hevea brasiliensis* Müll. Arg.	Naturalized	Low
11	HC	*Hura crepitans* L.	Naturalized	Low
12	LC	*Lantana camara* L.	Spreading	High
13	ME	*Maesopsis eminii* Engl.	Spreading	High
14	MP	*Mimosa pudica* L.	Spreading	High
15	PB	*Phyllostachys bambusoides* (Siebold. & Zucc.)	Spreading	High
16	PC	*Psidium cattleianum* Sabine	Spreading	High
17	RR	*Rubus rosaefolius* Sm	Spreading	High
18	SJ	*Syzygium jambos* (L.) Alston	Spreading	High

Source: Adopted from Dawson *et al.* (2009)

The hyperspectral measurements were smoothened using the state-of-art Whittaker smoother (Atzberger and Eilers, 2011). For each sample the fifteen spectra (three measurements * five leaves) were plotted with the DARWin SP Spectral Acquisition software. The plots were evaluated and visually inspected for data quality such as exclusion of outliers from further analysis. Mean, minimum, maximum and standard deviation of the spectral signature for each species were calculated. Spectral library was created using the *hsdar* package in R software (Lehnert *et al.*, 2018).

2.2.3 Data analysis

2.2.3.1 Simulated sensor specific resampling

The collected spectral data was resampled to simulate three selected sensors' response functions. Sensors were chosen such that they are currently in orbit or planned for the near future and have a high spectral resolution and a high spatial resolution to be able to identify single trees or groups of trees and shrubs. Thus, Worldview 2 (8 band), Sentinel-2A, and EnMAP were selected (Table 2.2). Spectral resampling of the pre-processed species mean spectra was performed using the sensor-specific spectral FWHM (i.e. full width at half maximum, FWHM/ λ) functions of wavelength (λ), which was the case for Worldview2 and Sentinel-2A. For the spectral resampling of EnMAP, a Gaussian distribution was assumed for all bands spectral responses. For the resampling the spectral resampling function used is implemented in the *hsdar* package in R software (Lehnert *et al.*, 2018).

Table 2.2: Overview of the selected sensors for spectral resampling

Sensor name	Band description	Spectral range	Bandwidth
WordView2	Coastal	400-450	50
1.8m resolution	Blue	450-510	60
	Green	510-580	70
	Yellow	585-625	40
	Red	630-690	60
	Red edge	705-745	40
	NIR 1	770-895	125
	NIR 2	860-1040	180
Sentinel 2	Coastal aerosol	433-453	20
10-20m resolution	Blue	457.5-522.5	65
	Green	542.5-577.5	35
	Red	650-680	30
	Vegetation red edge	697.5-712.5	15
	Vegetation red edge	732.5-747.5	15
	Vegetation red edge	773-793	20
	NIR	784.5-899.5	115
	Narrow NIR	855-875	20
	Water vapour	935-955	20
	SWIR- Cirrus	1360-1390	30
	SWIR	1565-1655	90
	SWIR	2100-2280	180
EnMAP	VNIR (96 bands)	420-1000	6.5
30m resolution; 96 bands- VNIR and	SWIR 1	900-1390	10

136 bands- SWIR	SWIR 2	1480-1760	10
	SWIR 3	1950-2450	10

2.2.3.2 Simulated sensor band selection

Maximum distance matrices using Euclidean distance for all sensor-specific bands and all species combinations was calculated to determine the optimal bands for species discrimination. Euclidean distance has shown better performance as a distance measure in evaluating magnitude of change and clustering compared to other distance measures such as manhattan and mahalanobis (Carvalho Júnior *et al.,* 2011; Singh *et al.,* 2013). The distance was further used to evaluate normalized ratio indices (NRIs) for all band and species combination. NRI, which is NDVI- like index, is implemented in hsdar package in R software. It gives information on each band combination frequencies of bands and NRIs that had achieved the maximum distance. Bands and ratios with highest frequencies are considered most suitable in discriminating the 18 species. All calculations were performed in R software.

2.2.3.3 Machine learning for optimum wavelength selection

Two machine learning algorithms were used, random forest and gradient boosting machine together with each individual species measurement to identify optimum wavelength regions for the differentiation of the 18 invasive tree and shrub species found in the study area. Field data contained wavelength ranging 350-2 500nm however, wavelengths that are affected by atmospheric influences (1 350-1 440, 1 790-1 990, 2 360-2 500nm) were dropped following Thenkabail *et al.* (2004). Random forest (RF) implemented in the *randomForest* package in R (Breiman, 2001) used 70% of the randomly and independently selected bootstrap sample data as training data. Thirty percent of the samples, also known as out of bag samples (OOB) were used to internally evaluate the model. It is proved that OOB provide a reliable estimate of training error (Cutler *et al.*, 2007).

The second machine learning algorithm used was Gradient boosting machine (GBM) implemented in the *gbm* package in R. It is based on Friedman (2001; 2002) greedy function approximation and stochastic gradient boosting approaches but has implemented a number of variations to choose from. Contrary to RF it fits new models sequentially based on the loss functions calculated in the previous models to provide a more accurate final estimate. There are various loss functions (i.e., distributions) to choose from besides a number of other hyper parameters that need fine tuning. In this study, the model used “gaussian” distribution, shrinkage, which controls the rate of optimization, was set to 0.005, and tree complexity, which controls the maximum number of interactions, was set to 5. The subsampling (i.e., bag fraction) was set to 0.7. The number of terms in the fit was determined by 10-fold cross-validation to prevent overfitting.

Both algorithms allow for the calculation of variable importance measures. In Random Forest *Mean Decrease Accuracy* (MDA) was used for classification, the prediction error rate for each tree was calculated and recorded using permuted OOB data. Then the same was done with permuted predictor variables. The differences between the two prediction error rate calculations were then averaged over all trees and normalized by the standard deviation of their differences. The variable importance function was used which is implemented in the *RandomForest* package in R software to perform this calculation. In GBM *relative influence* (RI) was used to quantify one selected wavelength relative to that of all other wavelengths. Each variable relative influence is estimated in the context of a model that includes all of the other variables and thus each variable effect is adjusted for the effects of the other varia bles (Ridgeway, 2007).

2.3 Results and Discussion

2.3.1 Field spectral library

Spectral library comprised of mean spectral signatures of 18 species derived from the 16 samples for each specie. The spectral curves shows a typical vegetation curve (Peñuelas *et al.,* 1993; Ceccato *et al.,* 2001). Nevertheless, there are significant differences in absolute reflectance and depth of absorption features. These differences are attributed by variations in chlorophyll and water content of these plant species (Smith and Blackshaw, 2003). Reflectance is low in the visible region between 400-700 nm due to absorption by the photosynthetic pigments except for the slight peak in the green wavelengths. For instance, *Cryptomeria japonica* have very low reflectance compared to the others (Figure 2.2). This is a coniferous specie with needle like leaves thus small leaf surface area while other species are broadleaved. This was also observed in a study by Shen and Cao, (2017) where they compared coniferous and broadleaved trees and showed low reflectance for the coniferous trees. On the other hand, *Rubus rosaefolius* showed the highest reflectance in the visible and NIR region compared to others but also deeper absorption features (Figure 2.3) which signifies a healthy plant.

Similarly, during fieldwork it was observed that while species like *Lantana camara* and *Clidemia hirta* were water stressed, yellowish with slightly dry leaves and fruits while *Rubus rosaefolius* had healthy and green leaves with fresh fruits. Furthermore, highest reflectance in the visible region and lower reflectance in the near infrared were observed for *Albizia chinensis* indicating a lower chlorophyll levels compared to *Lantana camara* and *Clidemia hirta* that had lower reflectance in the visible and higher reflectance in the NIR region signifying high chlorophyll levels (Ceccato *et al.,* 2001; Zomer *et al.,* 2009). The differences in absolute reflectances are substantial in the near infrared (NIR) region of the spectrum between 700-1 300 nm (Figure 2.2) due to lack of strong absorbing materials in plants in this region. The NIR regions is associated with biophysical quantity, structure and yield (e.g. biomass and LAI) (Ceccato *et al.*, 2001). Strong absorption windows are found around 1 450 nm and 1 950 nm caused by water content in the plant. Also, small absorption is observed around 950 nm and 1 200 nm in almost all 18 spectras. These observations demonstrate that the differences in chemical composition and physical structure of the plants affects the shape and magnitude of the spectral curve which is also described by Zomer *et al.* (2009).

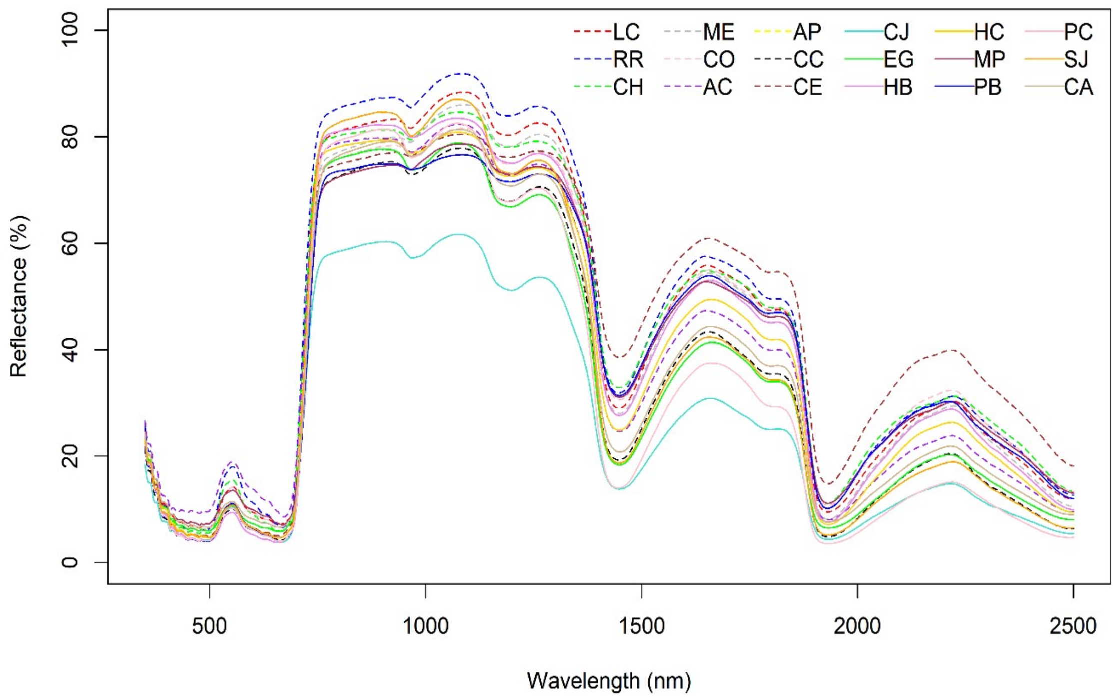

Figure 2.2: Spectral profiles of 18 Invasive species in East Usambara with legend showing species ID’s; refer to Table 2.1 for specie names.

2.3.2 Optimum band for species separation

Optimum bands for specie separation were obtained by simulating the field spectra of invasive species to sensor species bands and calculating Euclidean distance for each band and specie combination.

Frequency of bands with maximum distance was used to determine optimum bands for species discrimination. Results shows that for EnMAP simulated wavelengths, species could highly be differentiated in the regions of Visible near infrared and short-wave infrared. Band 1 (423 nm) had highest frequency with maximum distance for species separation followed by band 103 (1051 nm) and 100 (1 018 nm) (Figure 2.3: top plot). From more than 25 000 band combinations that NRIs were compared for all 232 EnMAP band, the highest frequency with maximum distance is obtained when comparing band 1 and bands with bandwidths ranging from 420 nm-600 nm. Additionally, high frequency was obtained when comparing band 1 and bands with bandwidth 1 018 nm and 1 051 nm. For EnMAP, bandwidth range of 420-600 nm comprises of more than 70 narrow Visible NIR bands, this means the Visible NIR region had more potential in differentiation of the 18 species while for SWIR there were only few narrow bands where species were differentiated. Whereas EnMAP provides more detailed information about spectral reflection of the 18 invasive species through its 70 narrow bands in the Visible NIR, for multispectral data this could be reflected in a single broad band (Adam *et al.,* 2010). Although there were about 134 SWIR bands, only two were optimum in differentiation of the species. SWIR region have fairly high vegetation signal but also contain absorption features that are not available in the VNIR which are important in characterizing vegetation (Herrmann *et al.*, 2010).

Unfortunately, this wavelength region is not available in all multispectral sensors making the hyperspectral sensor important in species differentiation. Other researches have shown the potential of EnMAP data in vegetation and soil studies however they do not explain importance of individual bands (Schwieder *et al.,* 2014; Malec *et al.,* 2015; Siegmann *et al.,* 2015). The study found that for Sentinel-2, species could be differentiated mainly in the regions of SWIR and coastal bands. The highest frequency with maximum distance was obtained in Band 11 (SWIR Cirrus; 1 360-1 390 nm) followed by band 1 (coastal; 433-453 nm) and band 6 (red edge; 732-747 nm) (Figure 2.3: middle plot). NRIs were compared for each band combination and from a total of 78 comparisons result shows that species could highly be differentiated when comparing bands centred at 443nm and 1375 nm, 443 nm and 1 610 nm, 1 375 nm and 1 610 nm, 560 nm and 1 375 nm, 560 nm and 1 375 nm respectively. In contrary, Sentinel-2 band description in R Hsdar package is slightly different from European Space Agency's (ESA), Figure 2.3 is based on R description (Table 2.3) for the corresponding ESA's sentinel-2 band description. This analysis allowed for identification of most relevant wavelengths that can differentiate the 18 species thus reducing the number of wavelengths to be used for classification of the species. Although Sentinel could only differentiate the species through its broad band in the visible and SWIR bands compared to narrower bands of EnMAP, it has an advantage of spatial and temporal resolution (Ng *et al.,* 2017). Sentinel can provide more details in time through its 5 day revisit time while EnMAP obtains data once a month (Transon *et al.,* 2018). Moreover, Sentinel has higher spatial resolution which enables to map the smallest patches of vegetation without the need for spectral unmixing (Transon *et al.,* 2018). Hyperspectral data are currently not widely available and expensive while multispectral data such as sentinel-2 are not only widely available but also free (Transon *et al.,* 2018). On the other hand, the study showed that Worldview-2 had highest frequency with maximum distance at band 8 (NIR 2; 860-1 040 nm) followed by band 1 (coastal; 400-450 nm) and band 6 (red edge; 705-745 nm) as seen in Figure 2.3 bottom plot. The NRIs comparison for each band combination shows that highest differentiation is obtained by comparing bands centred at 427 nm and 908 nm, 546 nm and 908 nm, 478 nm and 908 nm respectively. Species could mainly be differentiated in the regions of Coastal, red edge and NIR 2 for Worldview-2 sensor. Since Worldview does not have the SWIR wavelength region the differences in the 18species was detected through its NIR2 band. Due to High Signal to noise ratio in the bottom of the atmosphere it is not possible to have both very high resolution sensor and SWIR wavelength region (S. Taylor *et al.*, 2011c). As a result, Worldview is beneficial because of its very high resolution which make it easier to map species at crown level (Cho *et al.*, 2015; Robinson *et al.*, 2016b). Hyperspectral data but also multispectral data with low spatial resolution most of the time contains mixed spectral which needs to be unmixed (Shoshany and Svoray, 2002; Braun and Herold, 2004; Frazier and Wang, 2011). It is therefore important to have prior to analysis the spectral signatures of the species either from pure sample in the image or from field spectroscopy (Schmid *et al.*, 2004). This shows how important the spectral library is in detection and mapping of plant species.

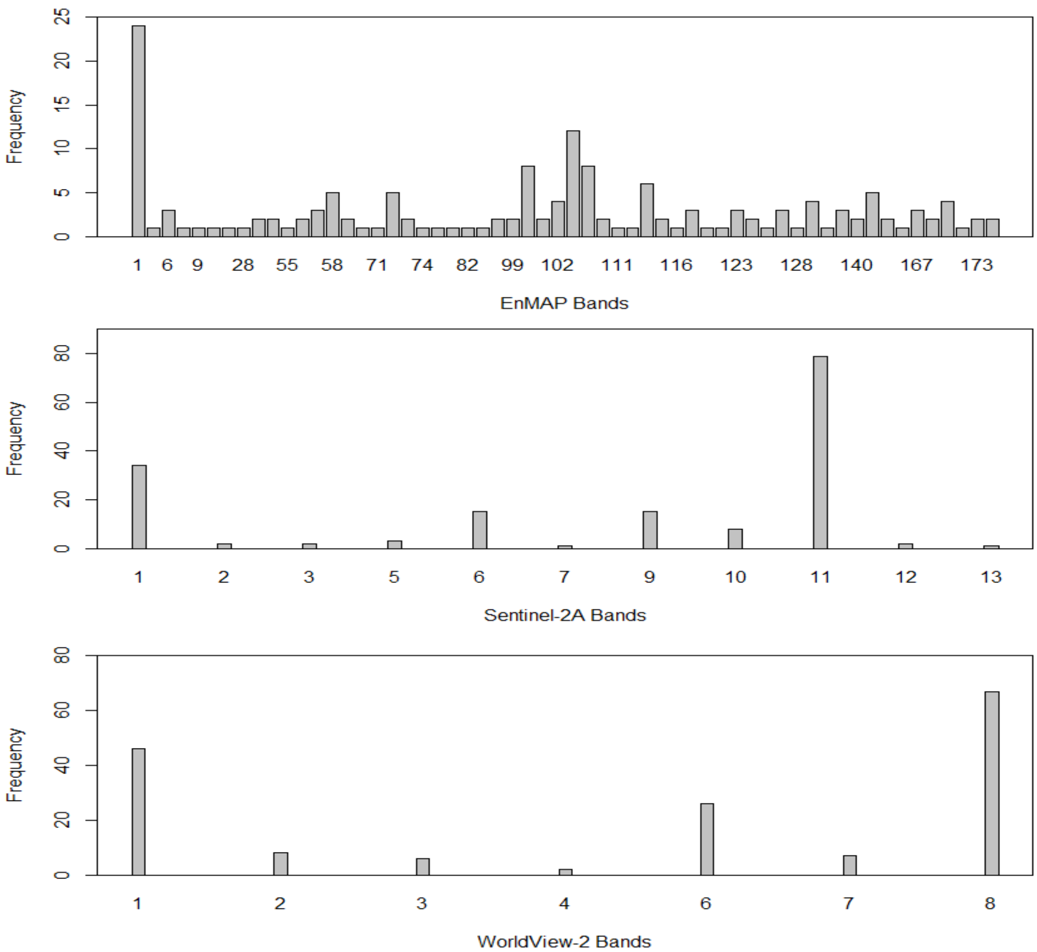

Figure 2.3: Frequency plots of sensor bands with maximum distances for pairwise species separation.

Table 2.3: Band characteristics for Sentinel 2A based on European Space Agency (ESA) and R- HSDAR package

	Band number (ESA)	**S2A Central wavelength (nm)**	**Band number (R)**	**S2A Central wavelength (nm)**	**Bandwidth (nm)**
Coastal aerosol	1	442.7	1	443	20
Blue	2	492.4	2	490	65
Green	3	559.8	3	560	35
Red	4	664.6	4	665	30
Vegetation red edge	5	704.1	5	705	15
Vegetation red edge	6	740.5	6	740	15
Vegetation red edge	7	782.8	7	783	20
NIR	8	832.8	8	842	115
Narrow NIR	8a	864.7	9	865	20
Water vapour	9	945.1	10	945	20
SWIR- Cirrus	10	1 373.5	11	1 375	30
SWIR	11	1 613.7	12	1 610	90
SWIR	12	2 202.4	13	2 190	180

2.3.3 Optimum wavelength ranges for species identification and separation

Figure 2.4 provides the result of random forest variable importance for the 18 species scaled from 0 to 1. Results show that wavelength ranging 350-383 nm in the visible range has the highest importance in the separation of the 18 species with variable importance ranging between 0.8-1. Further observation of Figure 2.4 shows that at SWIR ranging 1 778-1 790 nm and NIR ranging 1300-1350 nm and lies wavelengths important for the species separation with average variable importance ranging 0.5-0.7 and 0.4-0.5 respectively. Prediction error for random forest evaluated using OOB error was 0.31, this shows that the model could predict 70% correctly. GBM had single bandwidths which showed importance, where the visible had highest importance at wavelengths of 350 nm with a variable importance of 1. The SWIR important bands where at wavelength 1 577nm and 1790 nm with variable importance of 0.2 and 0.4 respectively. The NIR wavelength of 995 nm had variable importance of 0.1 (Figure 2.5). GBM model had 0.1 RMSE and 0.95 R^2 which showed high accuracy in the prediction. These results are consistent with previous researches, which showed that the visible region was very important in detection of vegetation using hyperspectral data. Adjorlolo *et al*. (2012) showed that wavelength range of 430-500 nm in visible and 1 500-1 630 nm in SWIR was important in discrimination of two grass species. On the other hand Prospere *et al.* (2014) found that at 400-610 nm in visible and 1 380;1 385;1 390 in the SWIR, plant species in tropical wetland could be highly differentiated using in situ hyperspectral data. The current study has optimum ranges that are not exactly the same as the previous studies. This could be due to sensitivity of such wavelengths to biochemical and biophysical characteristics of particular plant species. Shaokui Ge *et al.* (2008) revealed that spectral reflectance in the visible at 693 nm is highly correlated with chlorophyll content while between 1 038-1 704 nm there is high correlation with carbon content and nitrogen was highly correlated at 1 016-1 522 nm for invasive species giant reed. This shows that the band importance is strongly related to the physical and chemical properties of the plants. In general, this study confirms that the visible and short wavelength infrared regions are important in plant species separation. Particularly the result has provided specific wavelength regions important for the separation of the 18 invasive species found in East Usambara.

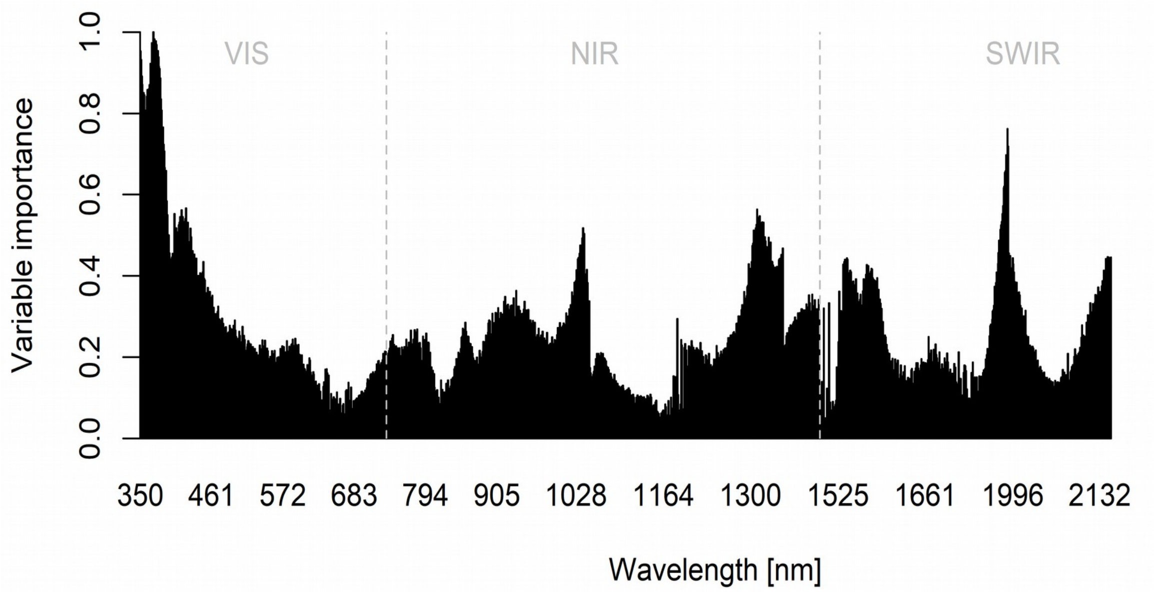

Figure 2.4: Variable importance for each wavelength for the 18 species using RF. Variable importance (MDA) was scaled from zero to one.

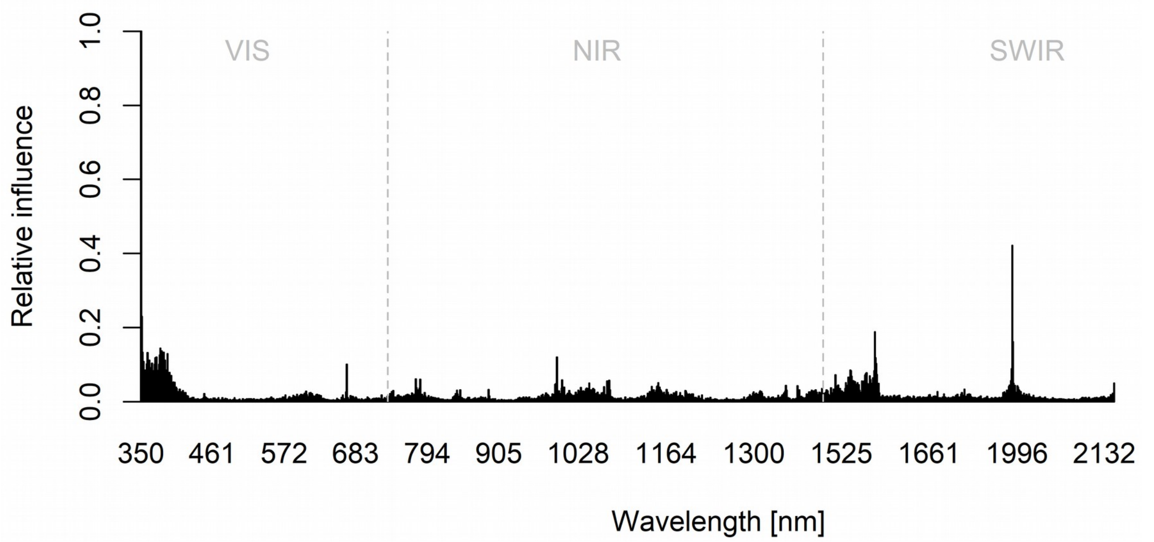

Figure 2.5: Relative influence of each wavelength for the 18 species using GBM/BRT, Relative influence was scaled from zero to one.

2.3.4. Conclusions

Modelling plant species distribution in a mix community using remote sensing requires to have a baseline knowledge on sensor characteristics and whether it can differentiate the species under study. This study tested by resampling the field hyperspectral reflectances to three sensors Worldview2, sentinel and EnMAP and evaluated the important bands for the 18 invasive alien plant species discrimination. Result suggests that the visible and SWIR are important in species discrimination for EnMAP and Sentinel resampled spectras. Since Worldview had no SWIR band the visible and NIR2 were important in separation of the species. Specifically, the wavelengths range of 350-383 nm; 1 300-1 350 nm and 1 778-1 790 nm showed high importance using random forest algorithm and bandwidth 350 nm; 995 nm; 1 011-1 013 nm; 1 577 nm; and 1 790 nm were highly important using GBM. It is evident that one needs a high spectral resolution satellite data to be able to discriminate these invasive species (Tesfamichael *et al.,* 2018) occurring in the same space. However factors like the size of invaded areas, type of plant species and the co-occurring plant species may affect the choice of sensor (S. Taylor *et al.*, 2011c). It is important that while choosing images for detection of the studied and similar species it should contain these wavelength ranges. It is also important for future satellite missions to include these wavebands so as similar species as the 18 in this study can be detected and differentiated. It would be vital for invasive species studies to have sensors such as Worldview with high spatial resolution but also including the SWIR bands.

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CHAPTER THREE

3.0 LINKING *LANTANA CAMARA* REMOTE SENSED DISTRIBUTION PATTERN AND HOUSEHOLD SURVEY DATA AS A TOOL IN ESTABLISHING IMPACTS ON LIVELIHOOD

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Abstract

The ecology and spread of invasive plant species in East Usambara are highly recognized. However, their spatial distribution and abundance has not been quantified. Detecting the distribution and abundance of invasive plant is crucial in conservation and management plans as well as in mitigating the negative effects. Moreover, abundance of *L. camara* serve as baseline data for assessing its effect on landcover, cost of clearing and effect on livelihood. Discriminating *L. camara* from other plants in East Usambara's evergreen forest and agroforest is challenging even with remote sensing since seasonality cannot be utilized and presence of trees obstruct the sensors in detecting understory shrubs. Therefore, this study modelled the distribution of *Lantana camara* using Very High-Resolution Worldview 3 data with 1.2 m resolution using object based random forest classification and estimated its fractional cover. *Lantana camara* was detected along the roads and forest edges with low abundance, while it was more abundant in agricultural lands, abandoned farms and tea plantations. Only 0.5% of the 11% invaded areas had *L. camara* abundance above 50% while 8.8% of invaded areas have *L. camara* abundance of less than 25%. Estimated cost to clear about 1 277 hectares of *L. camara* invasion in East Usambara is TZS 66.62 million, Shebomeza being the highly invaded village. It was also observed that income from crop production was not affected directly by *L. camara* fractional cover rather it was affected significantly by increasing cost of farm maintenance. Thirty percent of the respondents perceive *L. camara* to cause negative impacts while 32% believe it causes both negative and positive impacts. Among the negative impacts identified by respondents includes that it spread fast, hinder light, roots harden the soil, hinder growth of crops, weaken other species and increase crop production cost. Results also shows a decline in varieties of wild plant species and honey production based on community perception. It is recommended that *L. camara* should be managed while it is still not very dense and widely spread. The cost of managing will increase substantially with further invasion and mechanical means may not suffice.

3.1 Introduction

Invasive alien plants species (IAPS) are considered a threat to biodiversity, cultivated lands, ecosystems functioning and livelihood (Paini *et al.*, 2016; Vilà and Hulme, 2017; Linders *et al.*, 2019). These plant species have been moved from their native range into new areas for decades either intentionally or accidentally (Dawson *et al.*, 2011). Specifically, IAPS are known to affect among others biodiversity of an area by suppressing growth of many indigenous plant species (van Wilgen *et al.*, 2008; Pejchar and Mooney, 2009); loss of agricultural productivity by occupying agricultural lands and suppressing growth of crops (Romel *et al.*, 2007b; Thapa *et al.*, 2014). These losses together with efforts to control IAS costs US$33.16 billion in USA; US$16.3 billion in Canada and US$ 0.84 billion in South Africa annually (Pimentel *et al.*, 2005; Colautti *et al.*, 2006; de Lange and Van Wilgen, 2010). The cost of IAPS in Africa as a whole and specifically in Tanzania is unknown. Understanding the costs and associated impacts caused by IAPS requires both intensive assessment (abundance) and extensive assessment (spread) of the invasion. Early detection of potentially IAPS is crucial, because the expenses and effort required in halting and managing invasions increases considerably as the species spreads and becomes more abundant, until removal is impossible and control is very costly (Radosevich, 2006). Eradication of IAPS once they have naturalized and establish themselves in an areas has proven not to be practical (Wilson *et al.*, 2011). Once established, these species affect the ecosystem differently depending on the invasion level as well as extent of spread (Gooden *et al.*, 2009). It is therefore practical to manage the spread of IAPS by considering spatial distribution and invasion level of a particular locality, as well as environmental factors affecting spread and abundance of the species (Shackleton *et al.*, 2017). Different techniques have been applied to map distribution of IAPS over the past years (Joshi *et al.*, 2004; Adam *et al.*, 2017; Meroni *et al.*, 2017; Immitzer *et al.*, 2018). These includes species distribution models (SDM) that depend on the establishment of statistical relationship between environmental factors and presence of the species (e.g. max-ent, GARP, Bioclim) (Pearson and Dawson, 2003; Phillips and Dudik, 2008; VanDerWal *et al.,* 2019). These models give a way to predict current and future distribution of species (Sinclair *et al.,* 2010). Furthermore, prediction based on Remote sensed satellite images have also been applied, utilizing phenological changes occurring in vegetation (Meroni *et al.,* 2017; Taylor *et al.,* 2011a). However, this method performs best when invasion is in wetlands, grassland or desert areas where the absence of tree cover gives the sensor an unobstructed view of the invasion (Reschke and Hüttich, 2014; Adam *et al.,* 2017; Berhane *et al.,* 2018). It is also successful where plants are deciduous where the phenological changes can be utilized to differentiate the plants (Cho *et al.,* 2015; Meroni *et al.,* 2017). Detecting understory plants such as shrubs can be challenging and difficult. However, the presence of very high resolution satellite images (e.g. Cartosat-1, Quickbird, IKONOS, and Worldview) ranging between 3.5 m to less than a meter resolution provide more details and to some extent are able to detect the invasive plants (Robinson *et al.,* 2016b; Taylor *et al.,* 2011b).

One of the major threating invasive plant species globally is *Lantana camara* L. (Verbenaceae), hereafter referred as *L. camara*. It is a shrub native to South America and was mainly introduced by colonialist to East Africa in 19th century as ornamental and living fence (Sharma *et al.,* 2005).

The purpose of this study is to map the current distribution of the invasive alien plant *L. camara* using Worldview 3 satellite image in East Usambara. Thereafter use it to deduce the fractional cover of *L. camara* that can serve as a baseline data to assess the effect of *L. camara* on different land uses, the cost of clearing *L. camara* and effect on livelihood of smallholder farmers who are largely dependent on natural resources.

Particularly looking at the perception of the farmers from areas of different invasion level on impact of *L. camara* on change of variety of wildlife and plant species, as well as honey production.

3.2 Materials and Methods

3.2.1 Study area

The East Usambara mountains comprise a network of 18 forest blocks covering an area of 263 km^2 at an altitude range of 250 – 1 506 m above sea level (Burgess *et al.*, 2007a). Among the 18 forest reserves, Amani Nature Reserve is the largest (Hamilton and Bensted-Smith, 1989b). Amani Nature Reserve covers an area of 8 380 ha and lies between 300 to 1 128 m above sea level. It is a forest ecosystem that has a close interaction with people, since some have settled inside the Nature Reserve and many have settled close but outside the forest (Burgess *et al.*, 2007a; MNRT, 2010). The main economic activity of the inhabitants is cash- and food crop cultivation. The main cash crops include sugar cane and the spices cardamom, cinnamon, cloves and black pepper. The cultivated food crops include cassava, beans, sweet potatoes and maize (Hamilton and Bensted-Smith, 1989b; Teija *et al.*, 2010). Cultivated lands are invaded by invasive plants especially L. *camara* which threaten rural livelihood (Dawson *et al.,* 2008c; Goncalves *et al.,* 2014). In Amani, particularly fields cultivated with annual crops are vulnerable to *L. camara* invasion as they are left fallow for a certain period after harvesting.

3.2.2 Target specie

Lantana camara has several features that makes it a good invader, including being an evergreen plant with all year flowering and fruit production in many areas ranging from sea level to 1800 m; preferred by birds and some mammals hence long range dispersal; its ability to coppice; allelopathic effect; the ability to hybridise with over 600 cultivars and hybrids; and vegetative propagation (Howard, 1969; Thomas and Ellison, 2000; Sharma *et al.*, 2005; Priyanka and Joshi, 2013; Shackleton *et al.*, 2017). It is a multi-stemmed, medium-sized woody shrub which is native to Central and South America (Sundaram and Hiremath, 2012b). *Lantana camara* was introduced to Eastern Africa in the 19th century as an ornamental plant (Day and Zalucki, 2009; Sharma *et al.*, 2005). *Lantana camara* is reported to invade 5 million ha in Australia, 13 million ha in India and 2 million ha in South Africa and it continues to spread in these regions (Bagwat *et al.*, 2012). The spread of *L. camara* is still increasing globally, even in areas where it has existed for many years (Day *et al.*, 2003; Goncalves *et al.*, 2014). It invades cropland and grassland, thereby reducing yield and accessibility (van Wilgen *et al.*, 2008; Tadele, 2014; Gantayet *et al.*, 2014) and fodder production (Kohli *et al.*, 2006). It is reported to have negative impacts on biodiversity and native tree regeneration (Gooden *et al.*, 2009). It increases the risk of fire and is poisonous to livestock if eaten in high quantities (Sharma *et al.*, 1988).

3.2.3 Data collection

Field data

Training and validation data for landcover classification were collected between September and November 2017 and in February 2018. A total of about 1700 points were collected for different land uses across the study area by stratified selective sampling. Since *L. camara* invades in small patches and most of the time is mixed with other invaders, points were taken at the center of a minimum of 5x5 plot to make sure it falls inside a 1.2 m pixel in the satellite image. The points collected had coverage of above 50 % *L. camara* and exposure to sunlight of more than 50 % which were estimated visually. This was done so that the plot is dominated by *L. camara* and no other invaders, and also to avoid obstruction of trees that could make it difficult to detect *L. camara* in a satellite image.

Household survey

Household socio - economic data were collected in October 2016 through a structured questionnaire survey. A total of 146 households were surveyed across the 10 villages in the study area. The head of the family or the oldest family member available was interviewed in native Swahili language with the help of local field assistants and translated to English. Questionnaire had six parts (1) Perception of the household about *L. camara* (2) Impacts of IAS on rural household livelihood (income, expenditure and consumption) (3) Impacts of woody IAS on biodiversity, ecosystem services and land use decisions (4) Measures taken and management options (5) Social services (6) Sample household background information.

Household farms were digitized and field estimation of *L. camara* fractional cover done in February 2018. Out of 146 households interviewed, 101 were available at the time for digitization. Farms were digitized in the field using a personal digital assistant (PDA) with inbuilt GPS.

Spatial data

Worldview 3 data (WV3) was used to map *L. camara* distribution. This sensor was launched in August 2014 and collects panchromatic data at 31 cm spatial resolution and is owned by DigitalGlobe. Furthermore, the sensor has eight multispectral bands (red, red edge, coastal, blue, green, yellow, near-IR1 and near-IR2) from 400 nm – 1 040 nm at 124 cm spatial resolution (Worldview3.digitalglobe.com). Unfortunately, the short wave infra-red (SWIR) bands at 1 195 nm – 2 365 nm was not acquired. The image was mainly free of cloud cover and was acquired on 3rd February 2016. Thematic data included forest, tea, eucalyptus and water bodies boundaries which were manually digitized from WV3 image.

3.2.4 Data analysis

3.2.4.1 Land cover mapping and *Lantana camara* fractional cover estimation

To accurately compensate for atmospheric effects, WV3 was atmospherically corrected by applying the MODTRAN-based Fast line-of-sight Atmospheric Analysis of Spectral Hypercubes (FLAASH) (Fernández-Manso *et al.,* 2014). Rural aerosol model was applied with initial visibility of 60km and ground elevation of 0.7 km. The image was orthorectified using RPC and RSM resampling in ENVI and converted to UTM zone 37S projection from its original geographic coordinate system.

Texture information were extracted in ENVI using Co-occurrence measures filter with 5x5 window and gray scale of 64. Texture information extracted included second moment, contrast, dissimilarity, entropy and homogeneity. Normalized difference vegetation index (NDVI) was also calculated for use in post classification.

Classification of landcover was done using object-based image analysis (OBIA) approach. In the OBIA approach the Multiresolution Segmentation (MS) is used to delineate the WV3 image using pixel similarity group through three radiometer image weights: scale factor (SF), shape (sh), compactness (cp) (Schultz *et al.,* 2015). In this study, several combinations value for SF, sh and cp were tested, and best results was obtained with SF (20), sh (0.10) and cp (0.50). The parameters were applied in the image segmentation, to classify *Lantana camara*, forest, cultivated, and others.

Using 1 344 field collected training points, the image was classified into nine classes by applying object based random forest (OB-RF) model in Ecognition. The RF model is a decision tree supervised classifier which use out-of-bag (OOB) technique to create a training data set (Pal, 2005). Several machine learning classifiers have been used to classify remote sensed images, however the Random Forest (RF) algorithm is one of the latest which has shown better accuracy results in classifying different land covers and plant species (Rodriguez-Galiano and Chica-Rivas, 2014; Berhane *et al.,* 2018; Shiferaw *et al.,* 2019a). Main classes identified include *L. camara*, tea plantation, bare, settlements, cultivated land/agriculture land, water bodies, forest and eucalyptus and teak plantation. Forest edge areas that had shadows were classified as water bodies and therefore were reclassified by assigning waterbody with NDVI above 0.3 to forest. Bare soil which was misclassified as settlement were reclassified using length/width ratio threshold. Settlements that had length/width ratio above 4 were assigned bare soil class, this was mainly to capture roads since they are linear features.

A total of 368 field points that were not used for training were used for validation. Validation was done for eight classes eliminating teak plantation class. Teak plantation was not included in the validation since it was only in one place in the study area and covered a small area. Evaluation of the result used the overall accuracy Index (OA) and Kappa Index (κ) (Foody, 2002). The confusion matrix was created by comparing the reference point map with the classification results.

The classified image was reclassified giving *L. camara* a value of 1 and remaining classes 0. The sum of pixels with *L. camara* were computed by the block statistics through a 12 x 12 window using neighborhood tools under spatial analyst tools. The sum was then dividing by 144 (total pixel in 12x12 window) using math tool in ArcMap. The resulting image was the fraction of pixel with *L. camara* among total pixels in 12x12 window with values ranging from 0 to 0.99.

3.2.4.2 Assessment of cost of clearing *Lantana camara*

Data on the cost of clearing *L. camara* at different cover levels (100%, 75%, 50% and 25%) was obtained through questionnaire survey from 100 households. A linear regression in R was used to relate cost of clearing and fractional cover that resulted in a model to calculate cost of clearing *L. camara* as shown in equation five (see EQ5). The model was then applied in a fractional cover map in ArcGIS using map algebra resulting in total cost of clearing *L. camara* in East Usambara.

EQ5 $Cost\ of\ clearing\ lantana\ per\ acre = 126248 * Lantana\ cover - 5648$

3.2.4.3 Assessment of effects of *Lantana camara* fractional cover on livelihood

Lantana camara fractional cover for each household was evaluated using the digitized farms and fractional cover map. A stepwise regression using generalized linear model was used to assess the effect of household *L. camara* fractional cover, cost of clearing *L. camara* at 100%, 75%, 50% and 25% cover, cost of farm maintenance per acre and annual working hours on income from crop production.

To assess the effect of *L. camara* on change of wild plant species and honey production, responses from household's respondents were plotted using bar charts. The charts were used to assess the respondents' perception on whether honey production and wild plant species are increasing, decreasing or they see no change at all with respect to *L. camara* invasion. Moreover, the perception of respondents on general effects of *L. camara* invasion was analyzed to assess if they see *L. camara* as having positive or negative effects; both positive and negative effects or no change at all.

3.3 Results and Discussion

3.3.1 *Lantana camara* distribution and fractional cover in East Usambara

The distribution modeling was done with an overall classification accuracy of 86% and kappa coefficient of 0.83 (Table 3.1). Results show that 88.7% of the study area is currently not invaded by *L. camara*, which is about 13 837 ha. About 11.3 % of study area is invaded with *L. camara* corresponding to an area of 1 761 ha. *L. camara* invades mostly cultivated land, forest edges and tea plantations edges as well as in eucalyptus plantation (Figure 3.1). The study area comprised 75% intact forest and tea plantation, which are known to be invaded by *L. camara* only in the edges and not inside. This makes only 25% of the study area prone to *L. camara* invasion of which 17 % is cropland. Cropland have constant application of chemical or organic fertilizer, herbicide and use of agricultural machinery, which are significantly related to increased richness and dominance of alien plant species (Chen *et al.*, 2013). The results supports the previous findings which showed that *L. camara* does not invade intact forest and that it mostly invades open canopy such as agriculture and rangelands (Prasad, 2010b; Abebe, 2018). It is proven by Sharma and Raghubanshi, (2010) that *L. camara* has a negative relationship with tree density, which means it invades areas with low tree density. Low tree density entails more canopy openness and hence more light availability to understory plants (Sharma and Raghubanshi, 2010). It also means less competition for resources such as soil nutrients and moisture (Sharma *et al.*, 2005; Sundaram and Hiremath, 2012b) and thus increase invasion (Prasad, 2011).

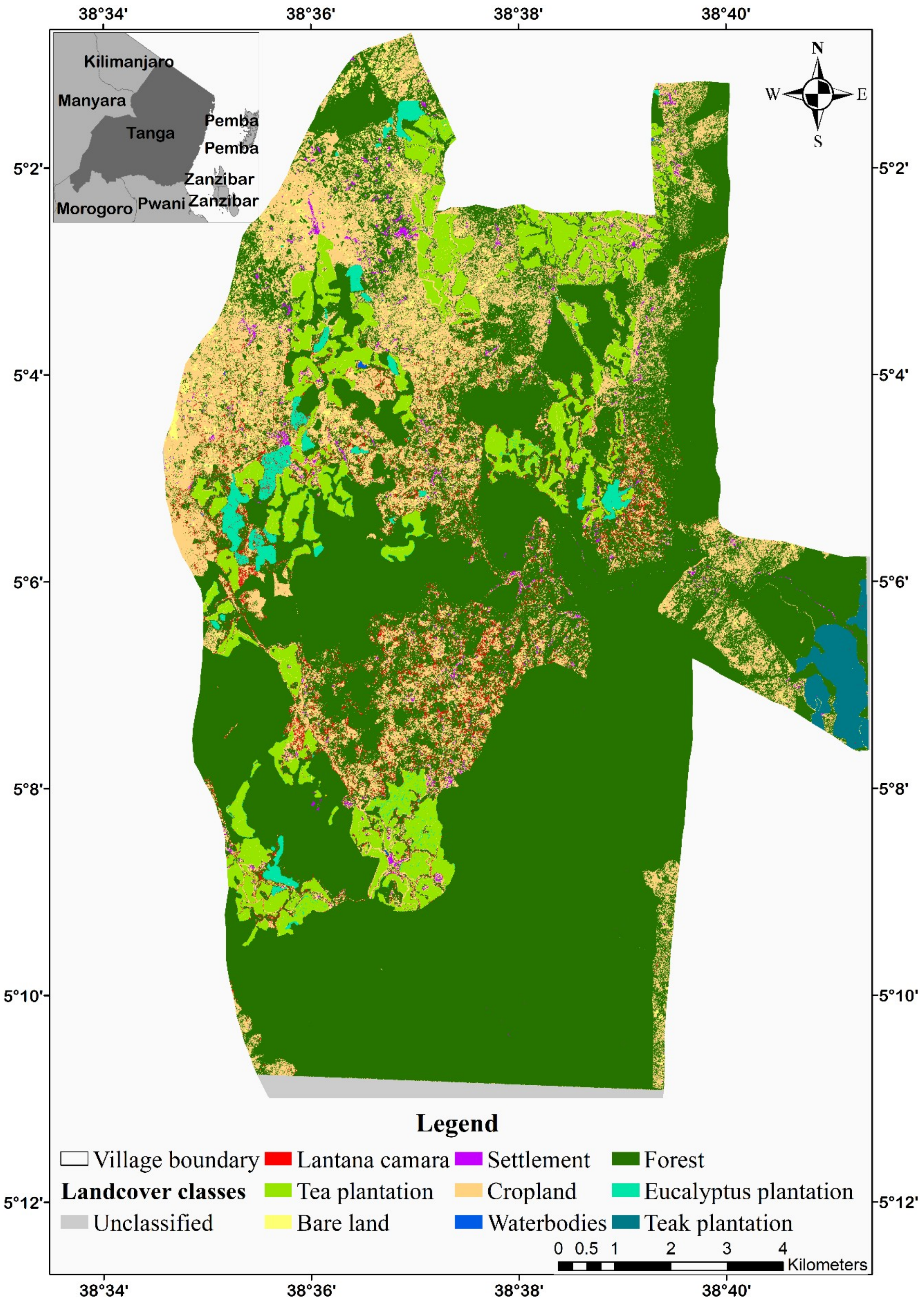

Figure 3.1: Landcover map showing *L. camara* distribution in East Usambara

Furthermore, within the 11.3% of the study area that is invaded by *L. camara*, 8.8% (1379.5 ha) has abundance of less than 25%, while 2% (305 ha) of the area is between 25% - 50% of *L. camara* abundance. The remaining 0.5% (76.5 ha) of the areas has *L. camara* abundance of more than 50% (Figure 3.2). Agroforestry with cultivation of perennial crops such cardamon, black pepper and cloves is common in East Usambara which is one of the reason *L. camara* is not abundant in East Usambara. These tree crops are continuously covering the land and are well maintained leaving less chance for high invasion by *L. camara* (Gallandt and Weiner, 2015). Even though *L. camara* does not seem to invade large areas of the East Usambara and is not present abundantly, continuous disturbances and human activities may intensify the spread of invasion and increase its abundance (Duggin and Gentle, 1998; Edward *et al.,* 2009; Chen *et al.,* 2013).

Table 3.1: Accuracy assessment for landcover classification

		Reference									
		Lantana camara	**Tea**	**Bare soil**	**Settlement**	**Cultivated**	**water bodies**	**Forest**	**Eucalyptus**	**Total**	**User accuracy**
Classification	**Lantana**	36	0	0	0	6	0	0	0	42	86
	Tea	0	45	0	0	0	0	0	0	45	100
	Bare soil	0	0	40	9	0	0	0	0	49	82
	Settlement	0	0	11	42	0	0	0	0	53	79
	Cultivated	6	6	1	0	43	0	0	0	56	77
	water bodie	0	0	0	0	0	10	0	0	10	100
	Forest	6	0	0	0	2	0	49	8	65	75
	Eucalyptus	0	0	0	0	0	0	0	47	47	100
	Total	48	51	52	51	51	10	49	55	367	
	Producer accuracy	75	88	77	82	84	100	100	85		

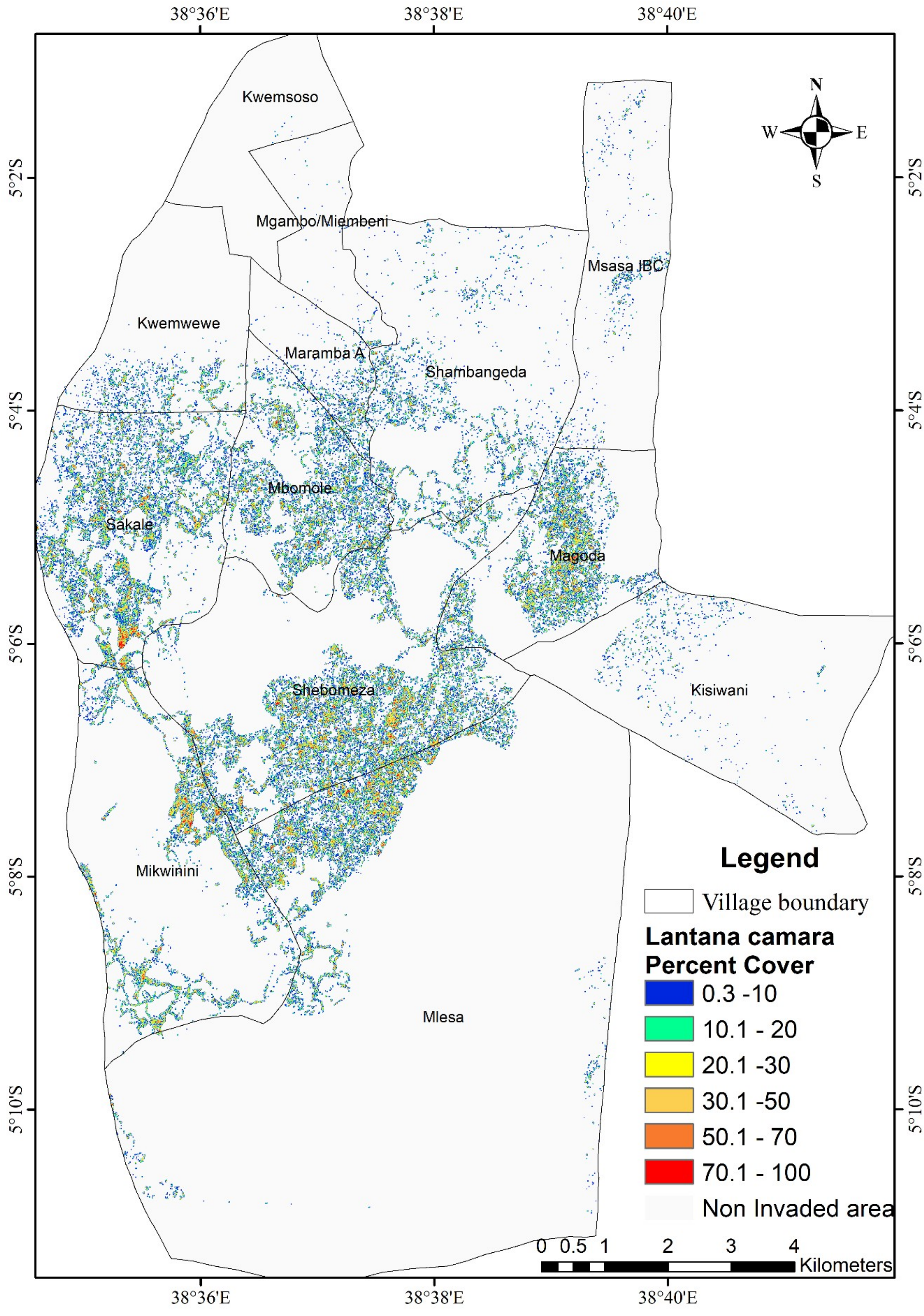

Figure 3.2: *Lantana camara* Fractional cover map for East Usambara

Fractional cover map was validated by comparing *L. camara* cover estimated through satellite image obtained in February 2016, field estimation done in February 2018 and *L. camara* cover estimated through questionnaire survey in October 2017. Results show a significant correlation between field and mapped data with R= 0.54 and p value <0.001, even though they were many zero values for field estimates (Figure 4.3). This is because many areas had been cleared at the time of field data collection. *L. camara* cover estimated through questionnaire survey did not correlate to mapped data (Figure 3.3) which may be a result of different perspective in visual estimation of *L. camara* between the farmers and the experts. Therefore, the mapped fractional cover map was adopted in the analysis of impacts of *L. camara.*

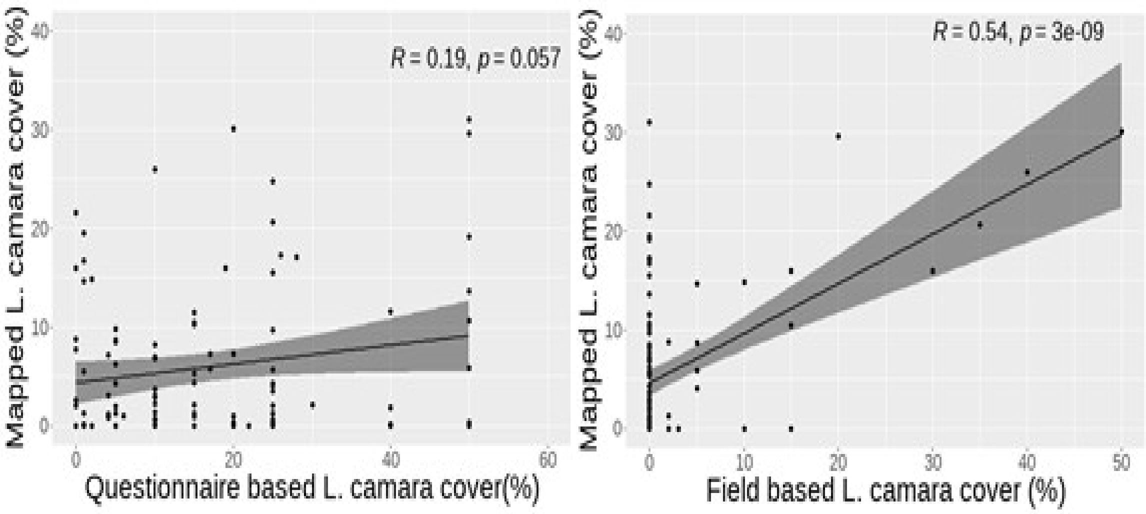

Figure 3.3: Correlation between *L. camara* fractional cover estimation from map, field and questionnaire data

3.3.2 Cost of clearing *Lantana camara* invasion in East Usambara

Shebomeza village had the highest invaded area, hence highest estimated cost of clearing amounting to TZS 16.61 million. Total estimated cost of clearing 1 277 hectares of *L. camara* invaded area in the study area is TZS 66.02 million. Results suggest that management of *L. camara* in East Usambara should focus on clearing the weed in the six villages with high invasion (Figure 3.4) to avoid further invasion in East Usambara and spread to other areas. The cost of managing *L. camara* would increase if no action is taken currently and is left to spread widely and become more abundant (Van Wilgen *et al.,* 2001). Mechanical removal of *L. camara* is labour demanding and therefore expensive to use in widespread and dense infestations, or in secluded or rugged areas (Van Wilgen *et al.*, 2001).

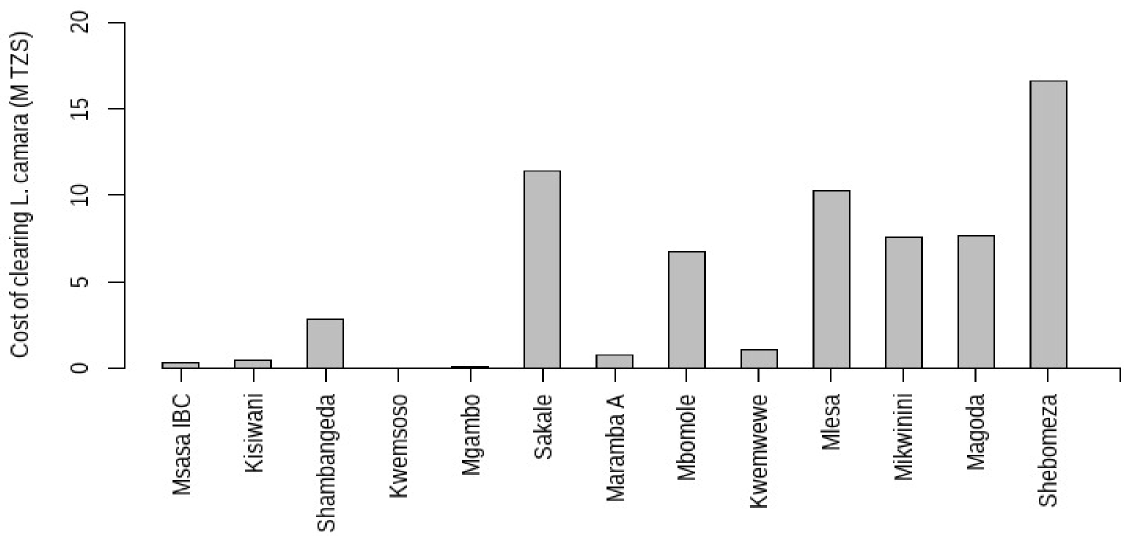

Figure 3.4: Cost of clearing *Lantana camara* in East Usambara

Furthermore, the areas invaded by *L. camara* are mainly cultivated land which means there is economic loss associated with crop production. *Lantana camara* like many invasive plants affect crop growth if it is left to co-exist with crops (Nel *et al.,* 2017) leading to decrease production. The estimated cost of clearing will be justified by the increase in crop production due to the land made available for cultivation and reduced weed-crop competition.

3.3.3 Effect of *Lantana camara* fractional cover on livelihood

Income from crop production is not directly related to *L. camara* factional cover rather through the cost of farm maintenance (p=0.002; Table 3.2). With increased abundance of *L. camara*, the cost of weeding increases and thus it affects overall income from crop production. *L. camara* forms dense thickets, which can be difficult to clear. Moreover, if roots are not properly removed it grows back very quickly and thus a need for weeding more often than usual.

Communities in East Usambara perceive *L. camara* as a weed that has both negative and positive impacts. From the questionnaire analysis, 30% of respondents perceive lantana as having only negative impacts while 32% said it has both positive and negative impacts (figure 3.5). The positive impacts identified were; increase soil fertility, used to make toothbrush, edible fruits for human and birds, source of fuel wood, used as medicine, support for tomato production, kill germs and insects, flowers attract pollinators and food for butterfly breeding. They describe negative impacts as; it spread fast, widely and strenuous, difficult to remove, other plants cannot grow, hinder light, roots hardened the soil, hinder growth of crops, weaken other species, increase crop production cost, reduce productivity of cinnamon and that it is a killer of plants.

Table 3.2: The effects of *L. camara* fractional cover (Lantana FC), cost of clearing *L. camara* at 75%, 50% and 25% cover (CC75, CC50 and CC25), cost of farm maintenance per hectare (Cost FM) and additional working hours due to *L. camara* (Additional WH) on income from crop production. (P < 0.05 is highlighted).Factor	df	F -ratio	P
Lantana FC	1,93	0.29	0.588
CC75	1,93	0.98	0.325
CC50	1,93	0.06	0.814
CC25	1,93	0.02	0.886
Cost FM	1,93	10.1	**0.002**
Additional WH	1,93	0.90	0.345

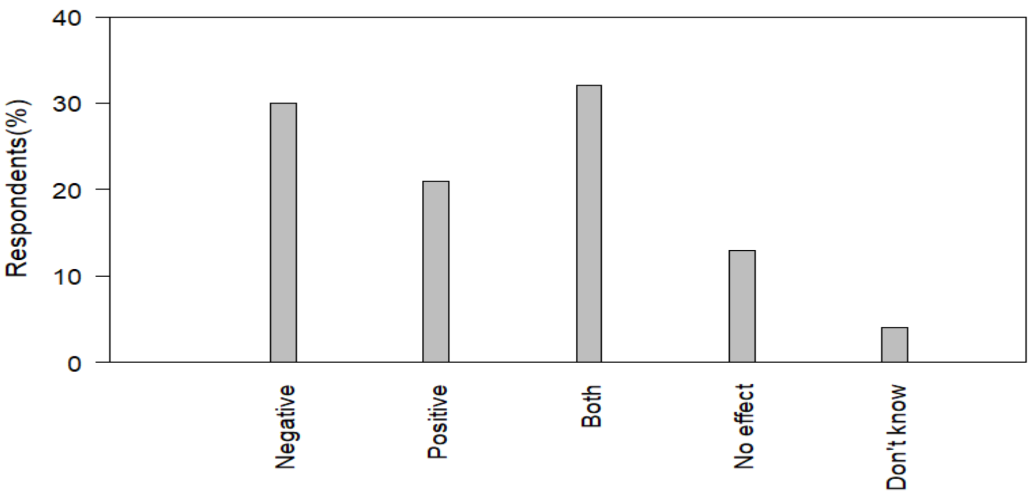

Figure 3.5: Perception of local community on overall effect of *L. camara* invasion

When the respondents were asked to compare the situation before invasion of *L. camara* and present time, 50% of respondents argued that honey production has decreased while 33% said there is no change (Figure 3.6). In addition, 67% said there is a decrease in variety of wild plant species and 25% believed there is increase in wild plant species (Figure 3.6).

Majority of people in East Usambara depend on crop production and other products from natural resources such as forest for their livelihood. Like other rural household in Africa, East Usambara community utilize native plant species for food, medicine, grazing, construction and other traditional uses (Bates, 1985; Chhabra *et al.,* 1990; Rasethe *et al.,* 2013).

The findings of this study indicate that *L. camara* has negative effects on rural livelihood through reduction of land area for cropping, hindering other sources of income such as honey production and reduced availability of wild plant species that the community depend on. The impacts argued by the interviewed households is supported by scientific researches which provide evidence that *L. camara* affects biodiversity and crop production (Day *et al.,* 2003; Romel *et al*., 2007b; Linders, 2019). *Lantana camara* is also reported to negatively affect vegetation diversity, cover and composition where by the effects are more visible in high *L. camara* cover (Ruwanza, 2020). However, the perception by the interviewed community that *L. camara* reduces honey production is in contradiction to some scientific studies that provided evidence that *L. camara* attracts pollinators and contains high volume of nectar (Carrión-Tacuri *et al.,* 2012; Goulson and Derwent, 2004). Reduced honey production may be associated with the community shift to butterfly breeding through projects launched in East Usambara. The community use *L. camara* as a source of food for butterfly.

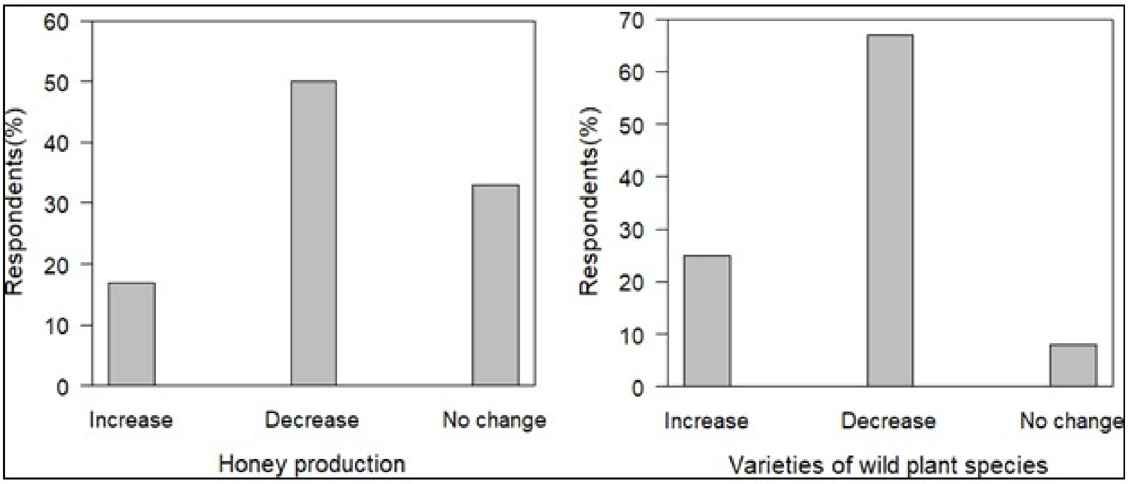

Figure 3.6: Perception of local community on effect of *L. camara* fractional cover on honey production and variety of wild plant species

3.3.4 Conclusions

Result from this study supports previous studies that *L. camara* invades mostly open canopies and that it does not invades dense forests. It is also proven that currently *L. camara* has invaded widely in cropland but is not very dense in East Usambara. The total cost for mechanical removal of currently *L. camara* invaded areas in East Usambara is estimated to be 66.02 Million Tanzanian Shillings. It is recommended that management of *L. camara* is crucial now before it spread widely and densely therefore making its management even more expensive. Findings from this study also indicate that *L. camara* negatively affects income from crop production through increased cost of maintaining the farms; availability of wild plant species and honey production. East Usambara community highly depends on crop production for their income as well as products from forest for their daily life. It is important that the cropping land and forests are managed to avoid too much disturbances which lead to further invasion.

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CHAPTER FOUR

4.0 IMPACT OF *LANTANA CAMARA* ON MAIZE AND CASSAVA GROWTH IN EAST USAMBARA, TANZANIA

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Abstract

The impacts of invasive alien plant species on native plants are generally well documented, but little is known about the mechanisms underlying their impacts on crop growth. A better understanding of immediate as well as legacy effects and of direct and indirect impacts of invasive alien plant species is essential for an improved management of invaded cropland. The study investigated how *Lantana camara* impacts the growth of maize and cassava through competition for resources, allelopathy and the presence of soil microorganisms. Two pot experiments were carried out using soils from invaded abandoned agricultural fields, and invaded or non-invaded cultivated fields. In the first experiment, maize and cassava were grown alone or together with *L. camara*. Half of the pots were treated with activated carbon to suppress allelochemicals. The effect of the soil microbial community on *L. camara* - crop interactions was assessed in an experiment using autoclaved soil with 5% of soil from the three soil types. Biomass of the crops was recorded after four months. It was found that *L. camara* reduced the growth of maize by 29%, but cassava was not affected. There was no evidence of allelopathic effects of *L. camara*. Inoculation of autoclaved soil with microorganisms from all soil types increased biomass of cassava and reduced the growth of maize. Because *L. camara* only caused impacts when growing simultaneously with maize, the results suggest that removal of *L. camara* will immediately mitigate its negative impacts on maize.

4.1 Introduction

Over the last decades, an increasing number of studies have documented the diverse and vast scale of the negative impacts of invasive alien plant species (IAPs) (Charles and Dukes, 2007; Vilà *et al.,* 2011; Pratt *et al.,* 2017; Ipsita *et al.*, 2014). Alien plant species become invasive when they escape from their original place of introduction to other areas and cause economic and environmental damage. For example in crop, pasture and livestock production, or through reductions in biodiversity or access to water resources. Invasive alien plant species are known to establish particularly well in disturbed habitats and farming practices such as shifting cultivation may facilitate alien plant invasions (Petit *et al.,* 2011). Even though impacts of IAPs on crop growth have been well documented, the mechanisms through which these species affect crop growth are not well understood (Ahmed *et al.,* 2007; Skurski *et al.,* 2014). Therefore, it is important to understand the mechanisms of impact of IAPs on crop growth for the development of targeted crop management.

The effects of IAPs may be caused by a variety of mechanisms, which can act immediately while the plants co-occur, or legacy effects may occur after the invader is removed (Grove *et al.,* 2012). Equally, effects can be due to direct interactions, or they can be mediated through changes in biotic or abiotic soil properties induced by the IAPs growing in the soil (indirect effects; (van der Putten *et al.,* 2013). Furthermore, IAPs may indirectly interact with crops and reduce their yield as a result of competition for resources, such as nutrients or light, that would otherwise be available for the crop (Gallandt and Weiner, 2015). For example, rapid growth of IAPs causes shading which can reduce the fitness of neighbouring plants (Page *et al.,* 2010). Allelopathy can inhibit growth of neighbouring plants either indirectly through changing soil chemical properties or microorganism as a result of the root exudates of IAPs, or directly via phytotoxic root exudates (Callaway and Aschehoug, 2000). Phytotoxic root exudates can mediate negative plant-plant interactions only if present at sufficient concentrations to affect plant growth and survival. The ecological relevance of phytotoxic root exudates also depends on the susceptibility of the plants with which the allelopathic plants coexist (Bais *et al.*, 2006). Allelochemicals bind to soil particles and some allelochemicals can persist in soil when the producing plant is no longer there, and may create a legacy effect (Del Fabbro and Prati, 2015). However, such persistent allelopathic effects have rarely been explored (Tian *et al.*, 2007; Grove *et al.*, 2012) and may not persist for a long time after the removal of the IAPs (Del Fabbro and Prati, 2015).

Indirect effects of IAPs may also be mediated by changes in the abiotic or biotic soil environment, i.e. plant-soil feedback (Ehrenfeld, 2010). For example, invasive plants often increase the available N and total N content of the soil (Vilà *et al.*, 2011), which may improve the environment for individuals of the same IAPs and may increase competitive interactions (Osunkuya and Perrett, 2011). Changes in the soil microbial community, cultivated or in response to the allelochemicals released by IAPs, can either be beneficial or detrimental for the growth of individuals of the same IAPs and other plant species growing in the soil. These changes in the soil properties can affect growth of other plants species even after the IAP has been removed or has died (Lankau *et al.*, 2014). The effect of changes in the soil microbial community has been illustrated by Wolfe *et al.* (2008), who found that the invasive *Alliaria petiolata* L. lowered growth of native tree species through phytotoxin-induced reductions in arbuscular mycorrhizal fungal colonization of the tree roots.

Lantana camara L. (Verbenaceae) is native to tropical America and was introduced in different parts of the tropics, e.g. India and Eastern Africa. It became a popular hedge and garden plant in early 20th century due to its introduction in botanical gardens in different European colonies (Dawson *et al.*, 2008d). The species invades a wide range of habitats, but grows best in open, disturbed ecosystems, along forest edges and roadsides (Sharma *et al.,* 2005). *Lantana camara* has become widely established in the East Usambara Mountains of Tanzania, which are dominated by tropical montane forests interspersed with villages and agricultural areas. The farmers in the region associate *L. camara* with lower crop yield if it is present on cultivated land. These claims were also reported by Shackleton *et al.* (2017) who assessed the perception of pastoral and agro-pastoral communities of Uganda claiming 26-50% reductions in crop yield. On the other hand, farmers also associate fields that were invaded with *L. camara* prior to cultivation had better crop yield, compared to non-invaded fields, which may be a result of improved soil quality (Fan *et al.,* 2010; Osunkoya and Perrett, 2011; Sharma and Raghubanshi, 2009). However, there is no experimental evidence that supports these claims.

Mechanisms of impacts, particularly whether the impact occurs in the presence of the alien plant species or represents the legacy of the alien plant species, may affect management decisions with respect to control and prevention of IAPs impacts in agricultural systems. Therefore, the aim of this study was to understand the mechanisms of impacts of the exotic *L. camara* on the growth of maize (*Zea mays* L.) and cassava (*Manihot esculenta* Crantz), in East Usambara. Specifically, the study tested in two parallel pot experiments whether the effect of *L. camara* on maize and cassava growth was mediated by competition for limiting resources, allelopathy or by altered microbial interactions. Also the effect of the native shrub *Whitfieldia elongata* (P. Beauv.) De Wild. &T. Durand (Acanthaceae) was assessed to test if the impact of *L. camara* is different from that of a common native shrub.

4.2 Methods

4.2.1 Study area

The East Usambara mountains comprise a network of 18 forest blocks covering an area of 263 km^2 at an altitude range of 250 – 1506 m above sea level (Burgess *et al.*, 2007a). Among the 18 forest reserves, Amani Nature Reserve is the largest (Hamilton and Bensted-Smith, 1989b). Amani Nature Reserve covers an area of 8380 ha and lies between 300 to 1128 m above sea level. It is a forest ecosystem that has a close interaction with people, since some have settled inside the Nature Reserve and many have settled close but outside the forest (Burgess *et al.*, 2007a; MNRT, 2010). The main economic activity of the inhabitants is cash- and food crop cultivation. The main cash crops include sugar cane and the spices cardamom, cinnamon, cloves and black pepper. The cultivated food crops include cassava, beans, sweet potatoes and maize. While the two main annual crops cultivated locally are cassava and sugar cane, maize and cassava was selected for these studies because the two are main food crops in Tanzania. In Amani, particularly maize fields are vulnerable to *L. camara* invasion as they are left fallow for a certain period after harvesting.

4.2.2 Plant material

Cassava stems from a single variety were obtained from a farmer in the study area in May 2017. The exact variety was unknown, since many farmers propagate cassava from stems when harvesting a field. For the experiment, cassava cuttings of approximately 30 mm diameter and 20 cm length were used. After consulting with the farmers, seeds of the main maize variety used in the study area were purchased (SEED.CO hybrid maize seed SC 719). Cuttings of the native shrub *W. elongata* and of *L. camara* were collected in the wild around the study area at the same time as the cassava cuttings. *Whitfieldia elongata* is a native shrub used as ornamental and widely distributed in Eastern Africa. It grows in open to moderately shaded areas and can be propagated through cuttings like *L. camara*. The cuttings of both shrubs had an average height of 30 cm and a diameter of 10 mm. The cuttings of *L. camara* and *W. elongata* were higher than that of cassava to simulate the field condition, since the shrubs outgrow crops in the field.

4.2.3 Soil material

The soil used in the pot experiments was collected from fields that differed in their history of *L. camara* invasion and agricultural management: 1) “Invaded” soil was collected from cultivated fields that had not been weeded during the previous three months and that had ca. 40% *L. camara* cover; 2) “Abandoned” soil was collected from fields that had been fallow for three years with a *L. camara* coverage >75%; and 3) “Non-invaded” soil was collected from cultivated fields that had not been weeded during the previous three months and contained no *L. camara* plants. Soil from each type was collected at six different farms. To avoid a bias caused by the previously grown crop type, for each soil type soil was sampled from three locations previously cultivated with maize, the other three with cassava. Soil was collected by first removing the litter and then collecting loose soil from the top soil layer down to about 50 cm depth using a shovel. The soil from each location was then mixed and stored in woven plastic bags. Samples from each soil type and location were analyzed for pH (pH-KCL), total nitrogen (Kjeldahl), organic carbon, carbon-nitrogen ratio, potassium, phosphorous (Bray and Kurtz 1), calcium, magnesium and sodium content according to standard laboratory protocols (Ministry of Agriculture and Fisheries, 2016). The soil samples to be autoclaved were obtained from a single location without *L. camara* to keep the nutrient content constant and transferred to the Mlingano laboratory, Tanga District, for autoclaving at 121°C for 30 minutes. Autoclaving was repeated once per day for three days. The soil samples were collected seven days prior to setting up the experiment to give time for autoclaving.

4.2.4 Study 1 – Effects of shrubs and allelochemicals

To assess the effects of the three soil treatments, the presence of *L. camara* or *W. elongata* and allelochemicals on the growth of the two crops, a full-factorial experiment was set up in 2L pots. The six locations where the soil samples were collected were used as replicates, which made a total of 216 pots. One third of the pots were filled with soil from each type. Activated carbon (2 % v/v; Charcoal activated Art. 2690, Loba Chemie, Mumbai, India) was added to half of the pots by mixing the required amount of activated carbon with the soil prior to filling the pots. Activated carbon can absorb organic compounds including allelochemicals and can reduce or even eliminate the negative effects of allelochemicals (Callaway and Aschehoug, 2000). One cutting of *L. camara* each was then planted in one third of the pots and one cutting of *W. elongata* in another third of the pots, while the remaining pots were not planted with any shrub. The weight of the cuttings was recorded prior to planting. Half of each of the pots (n=108) were then planted with three seeds of maize, the other half for each pot with one cutting of cassava. The weight of cassava cuttings was measured before planting. After one month, the maize plants were reduced to one per pot. The pots were placed in a completely randomized order in an experimental garden at the Amani Botanical Garden, Tanga District, Tanzania (5°06′2.5″ S, 38°37′46″ E; altitude 924 m a.s.l.) in mid-May 2017 and were watered daily. After four months, the crops and shrubs were harvested. Since shrub and crop roots were entangled, the roots were carefully disentangled using water to completely remove the soil and loosen the roots. Each crop and shrub from a pot was then stored in a separate paper bag and labelled. Aboveground and belowground fresh biomass were measured within one day from harvesting. Crops were then taken to the laboratory and oven dried at 80° C for 24 hours. Then the total dry biomass of the crops was measured, because not many roots were formed, after drying the root weight was insignificant.

4.2.5 Study 2 – Effects of microbial community

To assess the effects of soil microorganisms on the growth of maize and cassava four soil treatments were used: 100 % autoclaved soil, 95 % autoclaved soil plus 5 % of soil from abandoned, invaded or non-invaded fields. Adding 5 % soil that was not autoclaved was done to introduce microorganisms from the three soil sources, while other soil components such as nutrients remained largely unchanged (Shaw *et al.*, 1999). The addition was done by mixing the autoclaved and untreated soil samples before filling the pots. The six fields for each soil type were used as replicates. In half of the pots for each treatment, one cassava cutting was planted and in the remaining pots three maize seeds were sown as in Study 1, resulting in a total of 48 pots. The weight of each cassava cutting was recorded prior to planting. The pots were placed in the experimental garden in completely randomized order during first week of June 2017 and watered daily thereafter. After a month, maize plants were reduced to one per pot. After four months, the crops were harvested and the same measurement procedure as in Study 1 was applied.

4.2.6 Statistical analysis

Soil chemical properties were analysed using generalized linear models with the individual chemical parameters as response variables and the soil type as explanatory variable. Fresh and dry biomass values of each crop species were compared using Pearson correlation, which were positively correlated for maize ($r=0.71$, $P<0.05$) and cassava ($r=0.72$, $P<0.05$). The effects of soil type, allelochemicals and the presence of shrubs on biomass of maize and cassava (Study 1) were tested using a generalized linear mixed effects model with soil type, activated carbon, crop type and shrub treatments as fixed factors. The location where the soil was collected was included as random effect. Generalized linear models with dry biomass as response variable and crop species and soil chemical parameters were used to test the effects of the soil chemistry on crop biomass. For this analysis, pots where AC was added and where *L. camara* or *W. elongata* was present were excluded. Relative Efficiency Index (REI) was calculated (Connolly, 1987) as a measure of the relative competitive ability of the crops; the higher REI the stronger (higher relative competitive ability). REI was calculated as (ln (Final crop weight)-ln (Initial crop weight))-(ln (Final shrub weight)-ln (Initial shrub weight)). The effects of shrub type on REI was analysed using similar linear mixed effects models as above, but for the crops separately, as the data showed a bimodal distribution as a result of the differences in crop weight. The relationship between the increase in crop weight and the change in shrub weight was assessed using a Pearson correlation.

The effects of microbial community composition of the soil types on crop growth (Study 2) were analysed using a generalized linear model with biomass as response variable, the four soil treatments and crop type as fixed factors, and locations where the soil was collected as random effect. Least square means was used to test linear contrast within treatments if significant effects were found among treatments.

4.3. Results

4.3.1 Soil characteristics

The carbon to nitrogen ratio (C:N) was higher in non-invaded soil than in the soil samples collected from abandoned and invaded fields (18.5 ± 0.6, 14.2 ± 0.6, and 13.9 ± 0.5, respectively; $F_{2,40} = 7.4$, $P = 0.006$). The C:N ratio in soil samples of abandoned and invaded fields was not significantly different ($P > 0.05$). Similarly, the organic carbon content of soil samples of non-invaded fields was higher than that in soil samples of invaded and abandoned fields (0.95 ± 0.02, 0.80 ± 0.03 and 0.70 ± 0.03 %, respectively; $F_{2,40} = 7.5$, $P = 0.006$). Total nitrogen content did not differ among the three soil types ($P > 0.1$), indicating that the difference in the C:N ratio was due to differences in carbon content. A positive correlation between the organic carbon content and C:N ratios of the soils was found ($t = 3.98$, $P = 0.001$). The other parameters (P, K, Ca, Na, Mg, pH and EC) did not differ significantly among the three soil types (all $P > 0.1$).

4.3.2 Study 1 - Effect of shrubs, soil source and allelochemicals

The dry weight of cassava as expected was significantly larger than that of maize ($P < 0.001$; Table 4.1). Furthermore, a significant interaction between the crop and shrub treatments was found ($P = 0.048$), indicating that the crops responded differently to the presence of the two shrubs. Hence, the two crops were analyzed separately and found that the difference between crop and shrub treatments was significant in maize, while it was not in cassava ($F_{2,70} = 3.46$, $P = 0.037$ and $F_{2,78} = 0.56$, $P = 0.573$, respectively): Presence of *L. camara* suppressed maize growth by 29% ($P = 0.05$) compared to when maize was growing alone or in the presence of *W. elongata* (Fig. 4.1). Presence of *W. elongata* did not have a significant effect on maize biomass ($P = 0.7$). Additionally, the relationship between crop and shrub growth was assessed by comparing the change in shrub weight with crop biomass. No relationship between crop and shrub growth was found (Pearson correlation: $P > 0.2$) and no significant differences between shrub species were found for REI in both crops ($P > 0.1$; overall mean for maize and cassava 3.47 ± 0.18 and 0.37 ± 0.04, respectively). There was no difference in crop biomass among the soil types ($P = 0.367$; Table 4.1) and the addition of AC had no significant effect on biomass ($P = 0.428$). A significant negative relationship between the C:N of the soils and crop biomass was found ($F_{1,14} = 4.84$, $P = 0.045$), but no relationship with organic carbon content. Marginally significant positive relationships between crop biomass and Total N and the N:P ratios of the soils were found ($P = 0.090$ and $P = 0.077$), suggesting that N was the limiting nutrient for growth of the two crops.

Table 4.1: The effects of crop, activated carbon (AC), the presence of shrubs and soil type on dry biomass. Shown are degrees of freedom (denominator, numerator), F-ratios and P-values (P < 0.05 is highlighted).

Factor	df	F -ratio	P
Crop	1,168	626	**<0.001**
AC	1,168	0.63	0.428
Shrub	2,168	2.63	0.075
Soil type	1,16	0.86	0.367
Crop x AC	1,168	0.02	0.903
Crop x Shrub	2,168	3.08	**0.048**
AC x Shrub	2,168	0.1	0.901
Crop x Soil type	1,168	0.01	0.912
AC x Soil type	1,168	0.19	0.665
Shrub x Soil type	2,168	0.19	0.826
Crop x AC x Shrub	1,168	0.69	0.504
Crop x AC x Soil type	2,168	1.32	0.253
Crop x Shrub x Soil type	2,168	0.35	0.703

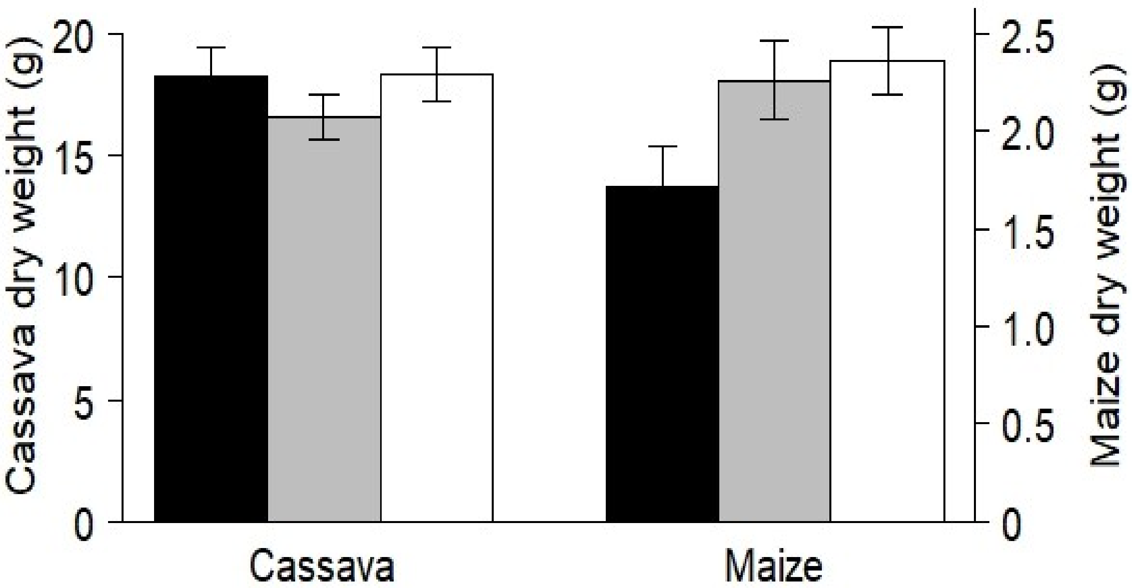

Figure 4.1: Dry weight (g) of maize and cassava, grown in pots together with *L. camara* (black bar), Whitfieldia elongata (grey bar), and no shrub (control, white bar). Error bars indicate one standard error

4.3.3 Study 2 - Effect of microorganisms

Biomass of the two crops was affected differently by the soil treatments ($P < 0.001$; Table 4.2). Maize biomass decreased by 90% when microorganisms were introduced in autoclaved soil ($P < 0.001$). In contrast, cassava biomass increased by on average 35% when microorganisms were introduced in autoclaved soil, but the effect of microorganisms was only marginally significant ($P = 0.07$; Table 4.2, Fig. 4.2). When autoclaved soil was omitted from the analysis a non-significant difference in maize and cassava biomass among the invaded and non-invaded soils was found ($P > 0.1$), indicating that the soil microbial community in the soil types had no different effect on the growth of the two crops.

Table 4.2: The effects of crop and soil treatment on dry biomass. Shown are degrees of freedom (denominator, numerator), F-ratios and P-values ($P < 0.05$ is highlighted).

Factor	df	F -ratio	P
Crop	1,17	145	**<0.001**
Soil treatment	3,19	0	0.992
Crop x soil type	3,17	19.9	**<0.001**

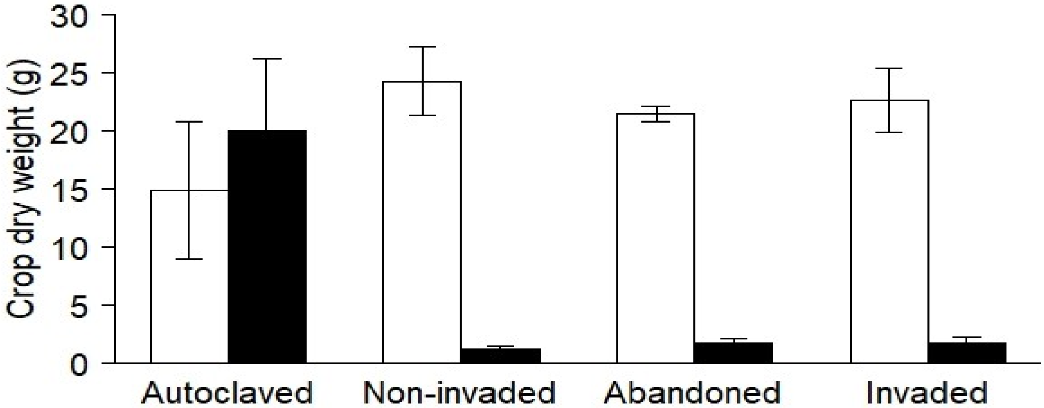

Figure 4.2: Dry weight (g) of cassava (white) and maize (black), grown in pots with autoclaved soil together with 5% (v/v) untreated soil from abandoned,

invaded or non-invaded fields Autoclaved indicates pots to which untreated soil was not added. Error bars indicate one standard error

4.4. Discussion

4.4.1 Immediate effects

The growth of maize (Experiment 1) in the presence of *L. camara* was reduced by 29% compared to maize grown alone or in combination with *W. elongata*. To our knowledge, this is the first study to demonstrate and quantify the impact of *L. camara* on crop growth in semi-natural conditions. This is important, because the economic impact of IAS is understudied, especially in Africa (Diagne *et al.*, 2021, Eschen *et al.*, in prep.). Ccompetition for limiting resources and allelopathy was considered as potential mechanisms for this immediate *L. camara* effect, but did not find support for either.Ssoil nutrient content and crop biomass had a weak relationship. In particular, the weak positive relationship with nitrogen and N:P ratio suggests that nitrogen was the limiting nutrient for growth of maize and cassava in these soils. While nutrient addition studies would be necessary to confirm that no competition for nutrients occurred, the absence of a relationship between crop and shrub growth or differences in REI suggests that competition for soil nutrients did not cause the reduction in maize biomass. In addition, one would expect that growth of cassava would also have been affected if competition played a role, and, it would also have occurred in the presence of the native *W. elongata*, which was not the case. Moreover, the results suggest that the growth of the shrubs (and perhaps cassava) was not much affected by the nutrient content of the soil. The weight of the shrub cuttings decreased during the experiment, even though leaves and roots were produced by all cuttings. This weight reduction may have been due to investment of resources by the planted cuttings in the formation of roots and leaves, but the study did not assess the biomass of new shoots or roots after planting separately. Drying of some parts of the plant may have reduced the shrub weight. Light and water availability are unlikely to have been limiting for growth of any of the plants, as no shading occurred due to the small size of the shrubs and the pots were watered every day. Hence, findings were not able to tell with certainty if competition for resources is the cause of reduction in weight, because it appears that the development of the shrubs was not entirely dependent on resources contained in the soil, but also on resources present in the planted cuttings.

The presence of activated carbon in pots with *L. camara* growing together with the two crops had no significant effect on their biomass, which indicates that no immediate allelopathic effects occurred. This is in contrast to studies that found that *L. camara* has an allelopathic effect. But many of those studies were experiments conducted in Petri dishes or pots using *L. camara* leaf extracts (Ahmed *et al.*, 2007; Ipsita *et al.*, 2014; Mishra, 2014) or dry shoot residue (Mersie and Singh, 1987) to assess allelopathic effects of *L. camara* on germination and crop growth. Previous studies have shown that *L. camara* leaf extracts can have an inhibitory effect on shoot and root elongation, as well as biomass of various crops and weed species (Ahmed *et al.*, 2007). *Lantana camara* dry shoot residues were found to strongly reduce growth of maize in these artificial conditions (Mersie and Singh, 1987) and in some cases increased crop growth in pots, which could be a fertilizing effect of adding the residues. Some studies indicate that Petri dish experiments may exaggerate the allelopathic effects of the studied plants (Hierro and Callaway, 2003) and don’t necessarily reflect natural conditions (Inderjit and Weston, 2000; Inderjit *et al.*, 2005). This study was conducted in a semi-natural environment by growing crops in pots filled with soil samples from agriculture fields, which may explain why there was no evidence of allelopathy. Moreover, the pots were free of litter and no decomposition of *L. camara* seems to have occurred during the study period. Hence, results suggest that allelopathic effects may be less common in competitive interactions between *L. camara* and crop plants than often stated or that such effects are context dependent.

4.4.2 Legacy effects

Assessment was done to validate whether *L. camara* changes soil properties, such as soil nutrients and microorganisms, of soil samples with different histories of *L. camara* invasion and consequently may have lasting effects on crop growth. However, there was no differences in soil properties that could be attributed to history of *L. camara* invasion.

Farmers in the study area claim that *L. camara* affects soil nutrients by fertilizing the soil (Hamad, personal observation). The C:N ratio and organic matter content were higher in non-invaded soils than in soils where *L. camara* was growing, while total N, P, K and other chemical parameters did not differ. Hence, these results do not support farmers' claims about the fertilizing effect of *L. camara* and this was confirmed by the absence of higher crop growth on invaded soils. The high organic matter content, and related higher C:N content in non-invaded soils in this study could be a result of the use of organic manure in these fields, that were actively cultivated, and does not appear to have been affected by *L. camara*. Similarly, a study in Australian sub-tropic rainforest and open Eucalyptus stands revealed no differences in soil N, P and K between *L. camara* invaded and non-invaded soils (Osunkoya and Perrett, 2011). However, some other studies have revealed differences in soil nutrient content between invaded and non-invaded sites, which were associated to *L. camara* invasion. Higher pH, Mg, Ca and K were found in *L. camara* invaded areas of forest, shrub-grassland and riverine habitat compared to uninvaded areas in Nairobi national park (Simba *et al.*, 2013). Higher N content was also found in *L. camara* invaded shrubland and grassland in a dry deciduous forest compared to non-invaded areas in India (Sharma and Raghubanshi, 2009). The effect of plant invasion on soil is often site and species specific, which may explain the different findings (Ehrenfeld, 2003). Moreover, it may be that *L. camara* may not have been the driver of these changes reported by other authors, and its establishment at the studied sites may have been the consequence of a disturbance, which can also result in changes or differences in soil properties (MacDougall and Turkington, 2005).

If allelochemicals bind to soil particles, and thus persist in the soil for an extended period, they may affect plants growing in the soil after the alien plant has been removed (Del Fabbro and Prati, 2015). However, allelochemicals may decompose and be utilized by soil microbes and consequently the allelopathic effects may disappear over time (Zeng, 2014). The addition of activated carbon to the soil that was previously invaded by *L. camara* had no significant effect on crop biomass. The results therefore indicate an absence of an allelopathic legacy of *L. camara*. These findings are similar to those of a study using soil from areas previously invaded and non-invaded by eleven different invasive species in European countries, where activated carbon was added to suppress allelochemicals (Del Fabbro and Prati, 2015).

Soil microorganisms can have many effects on plant growth. For example, presence of plant growth promoting bacteria and mycorrhizal fungi may improve soil properties and organic matter content (Hayat *et al.,* 2010; Nadeem *et al.,* 2014), but the soil microbial community may also contain organisms that are detrimental to the growth of certain plant species (Ehrenfeld, 2003). Changes in the microbial community or the abundance of soil microorganisms are therefore likely to affect the growth of crops. This was clearly illustrated in this study (experiment 2), where a 30% increase in cassava biomass and a 90% decrease of maize biomass was observed when comparing autoclaved soils with soil with 5% non-autoclaved soil, which added microorganisms from the eighteen fields where the soil was collected. The effects of microorganisms on crop growth was irrespective of the soil types with different *L. camara* invasion histories, indicating that

L. camara does not affect growth of maize and cassava through changes in the soil microbial community. The contrasting responses of maize and cassava to the presence of microorganisms suggest different relationships between these crops and the soil microbial community. One reason for these differences could be the presence of plant pathogens that affect maize, but not cassava or less so. Another reason may have been a higher cost associated with the formation of mycorrhizas for maize than cassava. Cassava depends on mycorrhizal fungi, but larger cuttings grow faster than small cuttings, apparently as a result of the larger reserves (Habte and Byappanahalli, 1994). Since large cuttings were used when planting cassava and seeds for planting maize, it is possible that the maize plants had to invest earlier in mycorrhizas than the cassava plants, which then reduced their growth rate. It seems unlikely that the mechanism was competition for resources between the plants and the soil microorganisms, as this would have led to growth reductions in both crop species.

4.4.3 Conclusions

Most previous studies of the impacts of *L. camara* were done under conditions that do not reflect realistic interactions between *L. camara* and native plants, such as addition of aqueous extracts or ground plant material to Petri dishes or soil. This study is unique, because it assessed the effect of the presence of live *L. camara* on crop crops growing in the same soils. Because of the methodological difference it is unsurprising that results from this study do not confirm the results of previous studies that indicated allelopathic effects of *L. camara*. However, results from this study demonstrate the need for realistic experimental conditions when assessing impacts of IAS on crop growth: although results were unable to identify the mechanism underlying the observed impact, the experimental approach allowed for identification of when impacts occur under semi-natural conditions, which indicates when management could be undertaken to minimise these effects.

The two studies aimed to disentangle immediate and legacy effects, as well as direct and indirect effects of *L. camara* on the growth of two major staple crops in East Africa, which may have implications for the management of *L. camara* in agricultural fields. Farmers state that increases soil nutrient content, resulting in a fertiliser effect once the plants are removed, but also that *L. camara* reduces crop growth. To our knowledge there is no study that has shown the beneficial effect of *L. camara* alleged by the farmers in East Africa (Shackleton *et al.*, 2017). The study also did not reveal beneficial effects of *L. camara* on soil nutrients or crop growth, but the results revealed that *L. camara* can have significant direct negative impacts on the growth of some crops: while maize biomass was significantly reduced in the presence of *L. camara*, the biomass of cassava was not affected. Since there were no legacy effects, the results suggest that removal of *L. camara* in croplands is sufficient to remediate their negative effects. Future studies where *L. camara* is experimentally removed from crop fields are needed to confirm this.

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CHAPTER FIVE

5.0 KEY CONTRIBUTION, CONCLUSIONS AND RECOMMENDATIONS

5.1 Introduction

This chapter highlights the key contributions the present study has made to the body of existing knowledge. It also outlines the areas that need further studies. The chapter further presents the conclusions drawn from this study and highlights a number of policy recommendations.

5.2 Key Contribution of the Present Study

The study has provided the spectral library for 18 IAPS which will be a baseline information for future detection, differentiation and spatial assessment of *L. camara, Clidemia hirta, Rubus rosifolius, Maesopsis eminii* among others using remote sensing techniques. The study has also contributed in identifying optimum spectral ranges (SWIR: 1 778-1 790nm and Ultraviolet: 350-383nm) for the species differentiation. Furthermore, the optimum band for species differentiation for sensors available and soon to be launched were identified. Optimum band for Worldview-2 sensor was NIR2 (860-1 040nm) and Sentinel sensor was SWIR cirrus (1 360-1 390nm) while for EnMAP sensor was band 1 (423nm).

This study provided the current distribution and fractional cover maps of *L. camara* which led to cost estimation of clearing currently 1 277 hectares of *L. camara* invaded areas in East Usambara which amounted to TZS 66.02 million. This information is particularly important to policy makers, planners and researchers in planning for management of this specie.

This study is unique, because it assessed the effect of the presence of live *L. camara* on crops growing in the same soils. Most previous studies of the impacts of *L. camara* were done under conditions that do not reflect realistic interactions between *L. camara* and native plants, such as addition of aqueous extracts (Ahmed *et al.,* 2007; Ipsita *et al.,* 2014; Mishra, 2014) or ground plant material (Mersie and Singh, 1987) or dry shoot residue to Petri dishes or soil. The study has empirically shown that maize growth is reduced by 29% if grown in the presence of *L. camara* and that it does not have any legacy effect. This study generally provides information that is important to the government, NGOs and other development actors in policy development for the management of the specie.

5.3 Conclusions

5.3.1 Optimum spectral ranges for identification of a range of invasive alien plant species in East Usambara mountain

The objective is presented and discussed in details in chapter two. Results suggest optimum bands for IAPS differentiation for currently available multispectral sensors depends on band availability for the particular sensor and bandwidths. Shortwave infrared band is proved to be optimum in differentiating IAPS using Sentinel sensor, however it is not present in Worldview 2 sensor. Therefore for cases like Worldview 2, the NIR band is best. Although EnMAP has SWIR band it contains very narrow bands in the visible region which made it possible to differentiate species in that region as compared to broadbands of Worldview 2 and Sentinel.

Most available sensors on board have bands starting at the visible region (400-700nm). They do not contain the ultra violet (UV) region (350-399nm) as was the case for the field spectrometer. Results from random forest show that the UV region (350-383nm) is the best region in differentiation of the species. In the absence of this region the SWIR region (1 778-1 790nm) is the second-best region for the species differentiation. It is evident that one needs a high spectral resolution satellite data to be able to discriminate these invasive species occurring in the same space. However, factors like the size of invaded areas, type of plant species and the co-occurring plant species may affect the choice of sensor. While it is important when choosing images for detection of the studied and similar species it should contain the SWIR bands, it is not possible to have both very high-resolution sensor and SWIR region. This is due to high signal to noise ratio in the bottom of the atmosphere. It is also important for future satellite missions to include UV wavebands so as similar species as the 18 in this study can be detected and differentiated. It would also be vital for invasive species studies to have sensors such as Worldview with high spatial resolution but also including the SWIR bands.

5.3.2 Current distribution and fractional cover of *L. camara* in East Usambara Mountains

This objective is presented and discussed in details in chapter three. Worldview 3 image was able to map *L. camara* successfully with accuracy of 86% utilizing the NIR 2 band as advised in chapter two of this study. Results from this study supports previous studies that *L. camara* invades mostly open canopies and that it does not invades dense forests. About 11.3% of the study area is currently invaded with *L. camara* which is approximately 1761 hectares. It is also proven that currently *L. camara* has invaded widely in cropland, forest edges, tea plantation edges and in eucalyptus plantations. *L. camara* abundance is still very low in East Usambara with 78% of the invaded areas (1 379.5 ha) having abundance of less than 25%. Even though *L. camara* does not seem to invade large areas of the East Usambara and is not present abundantly, continuous disturbances and human activities may intensify the spread of invasion and increase its abundance.

5.3.3 Impacts of *L. camara* on selected ecosystem services and livelihoods in East Usambara Mountains

This objective is presented and discussed in details in chapter three and chapter four. East Usambara community highly depends on crop production for their income as well as products from forest for their daily life. However, findings from this study indicate that *L. camara* negatively affects income from crop production through increased cost of maintaining the farms (weeding); availability of wild plant species and honey production thus affecting their livelihood. It is estimated that the total cost for mechanical removal of 1 277 hectares of *L. camara* invaded areas in East Usambara to be 66.02 million Tanzanian Shillings. Management of *L. camara* is crucial now before it spread widely and densely therefore making its management even more expensive. It is important that the cropping land and forests are managed to avoid too much disturbances which lead to further invasion.

Lantana camara can have significant direct negative impacts on the growth of some crops: while maize biomass was significantly reduced in the presence of *L. camara*, the biomass of cassava was not affected. Since there were no legacy effects, removal of *L. camara* in croplands is sufficient to remediate their negative effects. Future studies where *L. camara* is experimentally removed from crop fields are needed to confirm this.

Results demonstrate the need for realistic experimental conditions when assessing impacts of IAS on crop growth: although this study was unable to identify the mechanism underlying the observed impact, the experimental approach allowed for identification of when impacts occur under semi-natural conditions, which indicates when management could be undertaken to minimize these effects.

5.4 Recommendations

i. Study found that the ultraviolet region of the electromagnetic spectrum is the most suitable for species differentiation. However available sensors currently do not contain this region therefore future satellite sensors should include UV region so as to better differentiate similar IAPS as in this study that are co-existing together.

ii. The study discovered that currently available multispectral sensors the SWIR band is suitable in differentiating the IAPS. Therefore, when choosing sensors for the studied species detection it should contain the SWIR band.

iii. *Lantana camara* was found to have direct impact when growing with crops hence it does not seem to have legacy effects. Therefore, removal of *L. camara* is sufficient to remediate its negative effects on ecosystem services and livelihood.

iv. Since this study showed that *L. camara* is not yet widely distributed, therefore management strategies for *L. camara* in East Usambara should primarily focus on reducing the weed in invaded areas as well as prevention and Early Detection and Rapid Response for the non-invaded areas. Moreover, regional or national spatially explicit management strategy for this species should emphasize prevention alongside Early Detection and Rapid Response and control. Also, awareness of the negative impacts of woody IAPS should be raised among the general public and other stakeholders in regions that are still uninvaded to avoid further invasions.

5.5 Areas for Further Studies

Due to some limitation in this study, areas for further research were identified, so as to gain more insight onto the distribution of *L. camara* and its impacts on ecosystem services and livelihood.

i. The study (Chapter 2) tested by resampling the field hyperspectral reflectances to three sensors Worldview2, sentinel and EnMAP and evaluated the important bands for the 18 invasive alien plant species discrimination. Result suggests that the visible and SWIR are important bands in species discrimination for EnMAP and Sentinel sensors. Furthermore, Worldview does not have SWIR band, therefore the visible and NIR2 were important in discrimination of the species. Further studies should be done on attempt to map distribution of the 18 species in Tanzania using these bands with these sensors in order to inform management of these invasive alien species.

ii. The study (Chapter 3) supported previous studies that *L. camara* invades mostly open canopies and that it does not invades dense forests. It is not very dense in East Usambara. This study did not capture *L. camara* in tree plantations like cloves, eucalyptus and teak, therefore future studies should develop methods to detect this species in tree plantation so as to have better estimation of cost of management of this species.

iii. The study (Chapter 4) assessed the effect of the presence of live *L. camara* on crops growing in the same soils. Although the study could not identify the mechanism underlying the observed impact, the experimental approach allowed for identification of when impacts occur under semi-natural conditions, which indicates when management could be undertaken to minimise these effects. Results demonstrate the need for realistic experimental conditions when assessing impacts of IAS on crop growth. Therefore, future studies where *L. camara* is experimentally removed from crop fields is recommended so as to confirm this study.

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