STUDIES ON VARIABILITY, PHENOLOGY AND MANAGEMENT METHODS OF THE ALIEN INVASIVE TREE, *SENNA SPECTABILIS* (DC.) IRWIN & BARNEBY IN KERALA, INDIA

Thesis submitted to the University of Calicut in partial Fulfillment of the requirements for the degree of Doctor of Philosophy in Botany

by

Mr. MURALEEKRISHNAN K

Under the guidance of

Dr. Hrideek T.K

Department of Forest Genetics and Tree Breeding
KSCSTE-Kerala Forest Research Institute, Peechi, Kerala, India.

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KSCSTE-Kerala Forest Research Institute

(An Institution under Kerala State Council for Science, Technology and Environment)

Peechi - 680 653, Thrissur, Kerala, India

CERTIFICATE

This is to certify that the thesis entitled **"Studies on Variability, Phenology and Management Methods of the Alien Invasive Tree, *Senna Spectabilis* (Dc.) Irwin & Barneby In Kerala, India"** submitted to the University of Calicut for the award of degree of Doctor of Philosophy in Botany by Mr. Muraleekrishnan. K. is the result of bonafide research work carried out by him under my guidance in the Department of Forest Genetics and Tree Breeding, KSCSTE-Kerala Forest Research Institute, Peechi. Further, I certify this or part thereof has not been the basis for the award of any other diploma or degree either in any institution or university. It is further certified that, that the corrections/suggestions, recommended by the adjudicators have been incorporated in the thesis and that the contents in the thesis and the soft copy are one and the same.

Date: 20.06.2024
Place:Peechi

Dr. Hrideek T. K

Senior Scientist
Department of Genetics and Tree Breeding
KSCSTE - Kerala Forest Research Institute
Peechi,Thrissur, Kerala,India
&
Chief Executive Officer (on Deputation).
State Medicinal Plants Board Kerala, Department of AYUSH

DECLARATION

I, Muraleekrishnan. K hereby declare that the thesis entitled "**Studies on Variability, Phenology and Management Methods of the Alien Invasive Tree, *Senna Spectabilis* (Dc.) Irwin & Barneby In Kerala, India**" embodies the results of bonafide research work done by me under the guidance of Dr. Hrideek T.K, Senior Scientist, Department of Forest Genetics and Tree Breeding, KSCSTE- Kerala Forest Research Institute, Peechi. I further declare that this or part thereof has not been the basis for the award of any other diploma or degree either in any institution or university.

20.06.2024

Kalluvazhi

Mr. Muraleekrishnan. K

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Abbreviations

AIC	Akaike Index Criterion
AR5	Fifth Assessment Report
ASCII	American Standard Code For Information Interchange
AUC	Area Under Curve
BIO1	Annual Mean Temperature
BIO10	Mean Temperature Of Warmest Quarter
BIO11	Mean Temperature Of Coldest Quarter
BIO12	Annual Precipitation
BIO13	Precipitation Of Wettest Month
BIO14	Precipitation Of Driest Month
BIO15	Precipitation Seasonality (Coefficient Of Variation)
BIO16	Precipitation Of Wettest Quarter
BIO17	Precipitation Of Driest Quarter
BIO18	Precipitation Of Warmest Quarter
BIO19	Precipitation Of Coldest Quarter
BIO2	Mean Diurnal Range (Mean Of Monthly (Max Temp – Min Temp)
BIO3	Isothermality (BIO2/BIO7) (×100)
BIO4	Temperature Seasonality (Standard Deviation ×100)
BIO5	Max Temperature Of Warmest Month
BIO6	Min Temperature Of Coldest Month
BIO7	Temperature Annual Range
BIO8	Mean Temperature Of Wettest Quarter
BIO9	Mean Temperature Of Driest Quarter
BT	Bark Thickness
CC	Chlorophyll Content
CCAFS	Climate Change And Agricultural Food Security
CD	Crown Diameter
CIAT	International Center For Tropical Agriculture
CL	Crown Length
CPT	Candidate Plus Tree
CV	Coefficient Of Variation
D	Density
DAPA	Decision And Policy Analysis
DAS	Days After Sowing

DBN	Distribution
DEM	Digital Elevation Model Ecosystem Services
ENM	Ecological Or Environmental Niche Modelling
EROS	Earth Resources Observation And Science
ESR	Rooted Epicormic Shoots
EST	Treated Epicormic Shoots
F	Frequency
FAO	Food And Agriculture Organization
FD	Fruit Dry Mass
FF	Fruit Fresh Mass
FR	Forest Range
GARP	Genetic Algorithm For Rule-Set Production
GBH	Girth At Breast Height
GCMS	General Circulation Models
GD	Germination Duration
GH	Girth At Breast Height
GHCN	Global Historical Climatology Network
GP	Germination Percentage
GPS	Global Positioning System
Hadgem2- ES	Hadley Global Environment Model 2-Earth System Model
HT	Total Tree Height
HT	Total Tree Height
IAPS	Invasive Alien Plant Species
IAS	Invasive Alien Species
IBA	Indole 3 Butyric Acid
IBA	Indole 3 Butyric Acid
INCCA	Indian Network For Climate Change Assessment Institute
IPBES	Intergovernmental Science-Policy Platform On Biodiversity And Ecosystem Services
IPCC	Intergovernmental Panel On Climate Change
IUCN	International Union For Conservation Of Nature
IVI	Important Value Index
KFRI	Kerala Forest Research Institute
KFSC	Kerala Forest Seed Centre
LA	Leaf Area
LC	Leaf Chlorophyll Content

LD Leaf Dry Mass	
LF Leaf Fresh Mass	
LL: Leaf Length	
LPDAAC Land Processes Distributed Active Archive Center	
LW Leaf Width	
Maxent Maximum Entropy Modelling	
MC Moisture Content	
MODIS Moderate Resolution Imaging Spectroradiometer	
NAA Naphthalene Acetic Acid Organization	
PB Primary Branches	
R Correlation Coefficient	
R| Pearson Correlation Matrix	
RCD Root Collar Diameter	
Rcps Representative Concentration Pathways	
RD Relative Density	
Rdo Relative Dominance	
RF Relative Frequency	
RFC Raunkiers Frequency Class	
RL Root Length	
RN Root Number	
ROC Receiver Operating Characteristic Curve	
SAPCC State Action Plan For Climate Change	
SB Secondary Branches	
SD Standard Deviation	
SDM Species Distribution Model	
SE Standard Error	
SEDAC Socioeconomic Data And Applications Center	
SH Shoot Height	
SM Sapwood Moisture Content	
SRTM Shuttle Radar Topography Mission	
STD Standard	
TBA Total Basal Area	
TN Twig Number	
TSS True Skill Statistics	
UNESCO United Nations Educational Scientific And Cultural Organization	
VP Viablity Percent	
WMO World Meteorological Organization	

Chapter 1
Introduction

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Introduction

On a global scale, biological invasions and climate change are the primary factors contributing to the decline in biodiversity and changes in ecosystem services (Vilà and Hulme, 2017). Biological invasions can be defined as the processes through which species having no historical presence in a particular area breach biogeographic barriers, establish new populations, and expand their range, often with human assistance (Richardson *et al.,* 2000). Following habitat destruction, it is widely recognised as the second most significant threat to biodiversity (Miller *et al.,* 2010; Ficetola *et a.l,* 2007). According to the Inter-governmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES, 2019), approximately one-fifth of the Earth's surface, including areas of high biodiversity, is projected to be at risk of biological invasion.

The invasion of alien species seriously threatens both natural and managed ecosystems worldwide (Mack *et al.,* 2000). The proliferation of plant species beyond their indigenous habitats is experiencing a notable increase, and the causes and various means of introduction have been previously studied. Notably, most non-indigenous plant species do not exhibit invasive tendencies, and they may not necessarily be included in the roster of alien invasive plant species. According to the International Union for Conservation of Nature (IUCN), an invasive alien species (IAS) is a species that has established itself outside of its original or current distribution, and its introduction and/or proliferation threaten biological diversity.

Global policy and decision makers have implemented Multilateral Environment Agreements (MEAs) in response to the increasing challenges of Invasive Alien Species (IAS). These agreements, such as the Convention on Biological Diversity (CBD), Ramsar Convention on Wetlands, and Sanitary and Phytosanitary Measures of the International Plant Protection Convention (IPPC), address various aspects of IAS. They encourage Member States to prevent the introduction of alien and invasive species and to manage established populations. The CBD Strategic Plan for Biodiversity 2011-2020, specifically Target 6 of the Kunming-Montreal Global Biodiversity Framework, focused on invasive alien species (CBD 2023-COP-15). Additionally, the Intergovernmental Science Policy Platform on Biodiversity and Ecosystem Services

(IPBES) has recognised the necessity for a thematic global assessment of invasive alien species and their control (Deliverable 3(b)(ii); IPBES 2016).

Based on the findings of the tenth session of the IPBES convened in September 2023, it can be inferred that invasive alien species (IAS) are perhaps the least comprehended among the five primary catalysts contributing to biodiversity decline. These five drivers encompass alterations in land and sea utilisation, direct exploitation of organisms, climate change, and pollution, with IAS being the fifth.

Under the IPBES-2023 Assessment, Invasive Alien Species (IAS) is a subset of Alien Species that have been introduced, naturally, accidentally, or intentionally, into an environment outside their natural habitat. Human activities have facilitated alien species transportation, introduction, establishment, and spread. Human activities have introduced at least 37,000 alien species and have become established in new areas. Approximately 10% of these species have rapidly reproduced and out-competed native species for habitat, food and water, devastatingly impacting native biota and landscapes and spurring extinctions. These constitute IAS, with the proportion of established alien species known to be invasive ranging from 6% of all alien plants to 22% of all alien invertebrates. IAS has been a driver for 60%, and the only driver for 16%, of documented plant and animal extinctions at the global level. Increasing our understanding of IAS is thus crucial to halt and reverse biodiversity loss. IAS also impact food and water security, human health, nature's contributions to people, and a good quality of life. The economic cost is significant, approximating half a trillion USD annually globally.

Biological invasions and natural hazards share similarities in their occurrence and impact dynamics and the challenges associated with predicting and controlling them (Ricciardi *et al.,* 2011). A study conducted by Turbelina *et al.,* in 2023 aimed to quantify the costs of biological invasions compared to natural hazards to raise awareness and exert political influence. By analysing data on the economic losses caused by biological invasions and natural hazards from 1980 to 2019, it was found that the magnitude of financial losses resulting from biological invasions was similar to those caused by natural hazards. For instance, the economic losses from biological invasions amounted to $1,208.0 billion, while losses from storms reached $1,913.6 billion, and losses from earthquakes amounted to $1,139.4 billion. Global warming,

increased human migration for economic and political reasons, and the globalisation of trade and tourism have all been identified as potential factors contributing to a rise in biological organism invasions (Dai *et al*., 2022). Furthermore, these invasions have been linked to the irreversible loss of species (Newbold *et al*., 2015). The effective management and control of invasive plant species, while minimising their impact on ecosystems, rely on advancements in their physiology and ecology (Vantarová, *et al*., 2023). These invasive species exhibit distinctive traits, including rapid reproduction and growth, potent dispersal ability, remarkable adaptability, capacity to thrive in varied ecological zones, tolerance to a broad spectrum of edaphic and climatic conditions, production of abundant seeds that disperse effortlessly, high rate of dispersal, extended periods of flowering and fruiting, development of aggressive root system, short generation time, and extensive native range (Reddy, 2008).

The significance of tree invasions has escalated in recent decades, with more species, more extensive areas of invasion, a wider range of impacts, and an increasingly complex set of management challenges. As a result, tree invasions are now being studied from multiple perspectives. According to Richardson and Rejmanek (2011), there are currently 357 tree species that are invasive in at least one region of the world. Tree invaders can also dominate nutrient cycling, potentially leading to permanent changes in the site (Gomez-Aparico, L, 2008; Virah-Sawmy *et al.*, 2009; Staska, 2014). Exotic tree species have the potential to be more damaging than smaller plants, as they can spread beyond their point of introduction. Additionally, tree invaders create shade, which can act as a barrier to the re-establishment of many native species (Mascaro *et al.,* 2008).

Many alien plant species have been introduced into India, either deliberately or inadvertently, during its early history. Some of these species have become intertwined with the cultural heritage of India. However, among the numerous introduced species, certain ones have become invasive and they pose problems. The introduction of these alien species occurs through various means, including intentional release, accidental escape, transportation with contaminants, transportation through stowaways and corridors, and unassisted natural dispersal (Rajasekaran *et al.,* 2015). It is noteworthy that approximately 40% of the species in the Indian flora are of foreign origin, of which 25% are invasive, as reported by Raghubanshi *et al,* (2005). India, being a mega diverse country, is home to 45,000 wild plant species and approximately

90,000 animal species, with less than 50% of the geographical region surveyed up to 2008, according to the Ministry of Environment and Forests (MoEF, 2008). A study conducted by Khuroo *et al*., (2012) has reported the occurrence of 1,599 alien plant species belonging to 841 genera of 161 families in India. The alien flora thus represents 8.5% of the total Indian vascular flora.

There are several reports available that provide information regarding the invasive flora of a specific region or areas like North Western Indian Himalyas by Kohli *et al.,* 2004; Alien flora of Kashmir Himalyas by Khuroo *et al.,* 2007; Alien flora of Doon valley, northwest Himalaya by Negi and Hajra, 2007; Catalogue of Invasive Alien flora by Reddy, 2008; Alien flora of India by Khuroo *et al.,* 2012. The National Biodiversity Authority (NBA) has compiled a list of 53 invasive alien plant species in terrestrial ecosystems and seven species in aquatic ecosystems (Sandilyan *et al.,* 2018). India is currently grappling with a number of invasive alien plant species, like *Lantana camara* (Murali and Setty, 2001), *Parthenium hysterophorus* (Aneja, 1991; Gunaseelan, 1998; Singh and Kaur, 1997; Sankaran *et al,* 2013), *Mikania micrantha* (Gogoi, 2001; Sankaran and Srinivasan, 2001; Lahkar *et al,* 2011), *Prosopis juliflora* (Anoop, 2010;Dayal, 2007; Kaur *et al*., 2012), *Chromolaena odorata* (Mahajan and Azeez, 2001; Sankaran *et al.,* 2013; Naithani *et al.,* 2017).

With its extensive maritime history, Kerala has facilitated the introduction of numerous invasive species.In the state of Kerala, it has been reported that 82% of invasive alien plant species have been intentionally introduced into the forested areas (Sajeev *et al.,* 2012). The majority of these species are native to the American continent. The primary reason for the intentional introduction of invasive plant species was for ornamental purposes, accounting for more than half of the introduced invasive species. The negative impact of invasive ornamentals on the biodiversity of natural areas has become a severe concern in recent years (Qin *et al.,* 2016).

One such invasive tree species that has been introduced is *Senna spectabilis* (DC.) H. S. Irwin & Barneby, which is native to Central and South America (Satyanarayana and Gnanasekaran, 2013). Previously categorized as a medium risk invasive species (Sajeev *et al.,* 2012), it is now posing a significant threat to native species, particularly in the Wayanad Wildlife Sanctuary where it is spreading rapidly.

Additionally, *S. spectabilis* has the ability to suppress the regeneration of native species, thereby increasing the risk of extinction among the natives.

Senna spectabilis (DC.) H.S.Irwin & Barneby, a tree species listed in the global compendium of weeds as an 'environmental weed', 'garden thug', and 'naturalised weed', (Randall, 2007). Senna grows quickly, flowers and sets seed profusely, and coppices when cut (Mungatana and Ahimbisibwe, 2010). In India, the introduction of *S. spectabilis* in the Western Ghats was carried out without adequate knowledge of its potential to become an invasive species. Subsequently, the species has established itself extensively in new areas, presenting a challenging management task (Vinayan *et al.,* 2020). The proliferation of the invasive alien plant *S.spectabilis* presents a significant peril to the local wildlife and indigenous flora within the forested regions of the Nilgiri Biosphere Reserve, encompassing the Wayanad Wildlife Sanctuary (WWS), a crucial habitat for Asiatic elephants in India. This expansive area, spanning nearly 300 square kilometres, including the Wayanad Wildlife Sanctuary, North and South Wayanad Forest Divisions, as well as the contiguous Mudumalai, Bandipur and Nagarhole Tiger Reserves, has succumbed to the infestation of this invasive species (Hrideek *et al.,* 2020). Moreover, this species has entirely colonised the Western Ghats region, with distinct and fragmented populations now thriving in various regions of peninsular India.

S. spectabilis can rapidly grow, reaching heights of 15 to 20 meters within a short period. Its quick spread results in food scarcity for wildlife populations, particularly herbivores, and exacerbates the man-animal conflict in the district. Additionally, the species impedes the growth of other native tree species and grasses under its thick canopy. Most habitats invaded by *S. spectabilis* are highly disturbed due to human interference. Therefore, it is essential to generate basic information on the distribution and population of this invasive species to understand its precise impact on the forest ecosystem.

In order to enhance the efficacy of management strategies, Species Distribution Models (SDMs) can be employed to anticipate the potential invasive range of prioritized Invasive Alien Species (IAS). This approach facilitates the evaluation of the current spread of Invasive Alien Species in any given region through the utilization of various remote sensing applications. Additionally, the fundamental distribution maps can be utilized to evaluate the influence of climate change on the distribution of

invasive species. Furthermore, field observations on phenology, reproductive biology, and interaction with other plant and animal species will provide valuable insights for developing suitable strategies to manage this invasive species. There is a dearth of studies focusing on species distribution and the influence of climate change on invasiveness. This research gap poses a significant obstacle to the effective conservation and management of plant invasion issues in Kerala. In recent times, the impact of climate change has become evident through alterations in species distribution patterns within the region (Jose and Nameer, 2020).

The success of invasive plants is attributed to their morphological traits. This also study aims to test the hypothesis that genetic shifts in morphological traits or any adaptive variation have occurred in the invasive populations of *S. spectabilis* in Kerala through the use of karyotype analysis and molecular marker methods. Additionally, it is important to note that eradication and management efforts of this species is typically limited and very challenging, it is essential to prioritize the areas of invaded populations that are most important to control. Prioritization should be based on potential impacts of invader and potential for control. Eradication initiatives have been implemented in various regions across the globe, employing manual, mechanical, chemical and biological methods. However, the outcomes achieved thus far have been modest in their effectiveness (Shackleton *et al.,* 2018). Suitable strategies must be formulated for the control, eradication and management of the tree invasive species in relation to the research programme.

The Society for Ecological Restoration International (SERI) has defined ecological restoration as the process of aiding in the recovery of an ecosystem that has been degraded, damaged or destroyed. The restoration of a site that an invasive species has occupied can present unique challenges, as some species may continue to impact the ecosystem even after their removal, hindering the achievement of the desired restoration outcome (Cronk and Fuller, 1995). Developing restoration models for invaded landscapes can be crucial in controlling invasive plants and restoring native vegetation. However, subsequent restoration and effective land use are necessary but often overlooked after removing or eradicating invasive alien plant species. In addition to managing tree invasive experiments, the possibility of incorporating or testing restoration practices where *S. spectabilis* has been eradicated will be explored in this study.

Insufficient information is currently available regarding the geographical distribution, general biological characteristics, and response to the eradication or control of this invasive woody plant. Specifically, there is a dearth of knowledge regarding the occurrence and abundance of this invasive species in the Kerala region of the Western Ghats. It would be of great value to laypersons and professionals to gain insight into the types of sites and variables most commonly associated with this invasive tree. Recognition of these characteristics and variables would facilitate the reduction of invasion and aid in eradication. The acquisition of all these forms of information will enhance our capacity to develop invasion theory and effectively manage invasive species.

OBJECTIVES

The objectives of this research are as follows:

1. **To estimate the current distribution and abundance of *Senna spectabilis* in Kerala.**

2. **To study the pollination, seed dispersal, phenology and variability of *Senna spectabilis* populations in Kerala.**

3. **To develop a management protocol for controlling *Senna spectabilis* and restoration protocol using native species.**

It is expected that the diverse range of approaches and inquiries presented in this research on invasive tree species will serve as a productive and thought provoking framework for future studies and research programmes aimed at addressing these invasions.

Chapter 2

Review of literature

Chapter 2

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2.1 Invasion Biology

Globally, the two primary drivers of biodiversity loss and ecosystem service change are biological invasions and climate change, as noted by Vilà and Hulme in 2017. Biological invasions refer to the processes by which species without any historical record in a particular area, breach biogeographic barriers, establish new populations, and extend their range, primarily through human-assisted introductions, as defined by Richardson *et al.,*(2000). This phenomenon is considered the second most significant threat to biodiversity after habitat destruction, as highlighted by Miller *et al.,* (2010) and Ficetola *et al.*, (2007). According to IPBES (2019), one-fifth of Earth's surface, including global biodiversity hotspots, is predicted to be at risk of biologic invasion. The invasion of alien species poses a severe threat to both natural and managed ecosystems worldwide, as emphasized by Mack *et al.,* (2000). The identification of potential future invaders and the implementation of effective measures to prevent their dispersal and establishment pose a significant challenge to both conservation efforts and international commerce (Yang, *et al.*, 2012).

The integration of established self-perpetuating populations into native communities often disrupts their functioning (Richardson *et al.,* 2000). In another perspective, Invasive Alien Species (IAS), by definition, are species that are non-native (or alien) to the ecosystem under consideration whose introduction causes or is likely to cause economic or environmental harm to human health (CBD, 2002; 2004). Invasive plants threaten forest, wetland and agricultural habitats, where they are considered both a cause and a symptom of declining ecosystem function (Beerling, 1995; Mack *et al.*, 2000; Larsson *et al.,* 2001; Scott *et al.,* 2001). It alters the environment they invade. Once invasives have successfully colonized a landscape, they are challenging and expensive to control (Blossey, 1999).

The phenomenon of biological invasion has been causing a homogenization of the world's flora and fauna (Hobbs, 2000). The concept of invaders and invasion was initially introduced in an ecological context by Elton, widely regarded as the "father of invasion ecology", in his seminal book on invasion (Elton, 1958). Charles Elton was the first to acknowledge biological invasions as a significant issue that could

result in global biological homogenization (Elton, 1958). Subsequently, alien species have been identified as one of the most critical threats to global biodiversity, second only to habitat loss (Millennium Ecosystem Assessment, 2005; Pysek and Richardson, 2010). Following the influential work of Elton (1958), 'The Ecology of Invasions by Animals and Plants', narrative reviews and meta-analyses on the mechanisms of biological invasions, along with his other publications, have paved the way for research on invasive alien species worldwide, profoundly influencing our understanding of biological invasions.

Pysek *et al.,* (2010) conducted a study in Europe, which revealed that national wealth and human population density are the most significant predictors of biological invasions. This finding highlights the importance of considering socio-economic factors when developing strategies to manage and prevent biological invasions

Since then, there have been various interpretations and definitions of invaders and invasions. According to the International Union for Conservation of Nature (IUCN, 1999), alien invasive species are those that establish themselves in a natural or semi-natural ecosystem or habitat, act as agents of change, and pose a threat to biological diversity. Kolar and Lodge (2001) defined invasive species as 'non-indigenous species that spread from their original point of introduction and become highly abundant'. The significance of invasive species in today's context is such that Article 8(h) of the Biodiversity Convention recommends taking action to prevent the introduction and control and also for the eradication of these alien species that pose a threat to ecosystems, habitats and species.

The phenomenon of plant invasion can be divided into three distinct stages, namely introduction, colonisation, and naturalisation, as identified by Richardson and Pyšek (2000). Invasive species propagation takes advantage of the ecological disruption caused by either natural or man-made sources, creating an invasion window. During this window, the invasive species is able to out-compete the environmental, reproductive, and dispersion barriers of the invaded area, leading to a rapid spread of its population. It is important to note that the introduced propagule must compete with the already well-adapted native flora present at the site. Consequently, environmental factors that support the establishment of alien

propagules, such as resource availability, are considered to be of utmost importance during the introduction period (Davis *et al*., 2000; Rejmánek *et al*., 2005).

The successful establishment and persistence of introduced species can be attributed to several key determinants. These include uncontrolled vegetative growth, evasion of biotic constraints, prolific seed production, highly effective seed dispersal, successful germination and colonization, as well as adaptive morphological and ecological characteristics. Furthermore, superior propagule attributes that enable increased mobility, as well as the ability to outcompete native flora through resource competition or the manifestation of allelopathic effects, also contribute to their success (Sajeev *et al*., 2012).

Only a limited percentage of introduced exotic species possess the capability to become invasive. It is generally estimated that a mere 10% of introduced species will successfully undergo naturalization, and out of those, only 10% will proceed to become invasive (Pysek and Richardson, 2008). Despite the clear categorization of the invasion process as defined by Richardson, *et al.,* (2000), the process itself is highly intricate. Various other authors have presented different crucial stages, such as arrival, establishment, dispersion and stabilization (Ricklefs, 2005; Davis, 2009; Reise *et al.,* 2006), or introduction, naturalization, facilitation, increased distribution and stabilization (Marchante, 2001). Henderson *et al.*, (2006) have also proposed a sequence of introduction, establishment, naturalization, dispersal, population distribution and dispersal.

The successful invasion of an ecosystem by invasive plants is determined by their competitive advantages or traits in relation to native species. These advantages include disturbance, competitive release, resource availability, propagule pressure, enemy release, and empty niches (Levine and D'Antonio, 1999; Mitchell and Power 2003; Colautti *et al.,* 2004; Jakobs *et al.,* 2004; Gilbert and Lechowicz, 2005; Hierro *et al.,* 2005; Richardson and Pysek, 2006; Theoharides and Dukes, 2007). At the establishment stage, biotic filters including competition from other plants or interactions with herbivores, parasites, pathogens, pollinators and dispersal agents as stated by the enemy release and empty niche theories (Levine and D'Antonio, 1999; Mitchell and Power, 2003; Colautti *et al.*, 2004; Jakobs *et al.,* 2004; Gilbert and Lechowicz, 2005; Hierro *et al.,* 2005) may be the significant barriers to survival,

growth and reproduction for developing self-sustaining, expanding populations at the community scale (Theoharides and Dukes, 2007).

Some other studies also stress these types of invasive plants' characteristics. Important drivers for adaptation and establishment of invasive species include favourable climatic factors, availability of resources such as light, water and soil nutrients as well as ecological factors, pressure of incoming seed propagules (Lockwood *et al.,* 2005; Thuiller *et al.,* 2006) and the presence of habitat typologies that are suitable to their growth (Chytry *et al.,* 2009; Richardson and Pysek, 2006).

Most of the studies were concerned with explaining the causes of biological invasions. In contrast, a smaller but still substantial number of studies were primarily concerned with documenting or testing the impacts of invaders. Those studies that seek to explain the causes of invasions do so by implicitly or explicitly testing or examining hypotheses for the success of the invaders, typically in particular systems (Lowry *et al.*, 2013).

During the 1980s, the Scientific Committee on Problems of the Environment (SCOPE) programme endeavored to bring attention to this significant threat and provided a comprehensive summary of the available information. Consequently, numerous books and proceedings were published (Drake *et al.*, 1989; Duffey, 1988; Groves and Di Castri, 1991; Ramakrishnan, 1991). Williamson's publication "Biological Invasion" (1996) conducted a thorough review and synthesis of invasion research. Exotic species are responsible for substantial economic losses; they pose a threat to native biodiversity and alter ecosystem functions (Vitousek, 1986; 1990; D'Antonio and Vitousek, 1992; Lodge, 1993; Schmitz and Simberloff 1997; Enserink, 1999; Pimentel *et al.,* 2000; Meyerson and Reaser, 2003).

Shigesada and Kawasaki (1997) examined the mathematical model of the spread of invasive species. However, a perusal of the SCOPE-related literature indicates that no serious attempt was made at gathering, analyzing and interpreting the available information on woody plant invaders. For example, in his review, Rejmaneck (1989) included only 32 woody species. Whitmore (1991) produced a similar study for the tropics, chiefly based on evidence from Southeast Asia, and claimed that all tropical moist forests resist invaders. However, several cases had been previously reported (Binggeli, 1990). Cronk and Fuller (1995), Pysek *et al.,* (1995)

and Rejmaneck (1996) provided a much broader, but not comprehensive review of the subject. In the plant kingdom, many invasive species are woody (including sub-shrubs with stems woody at the base).

The human-made introductions in the new habitats are quick and responsible for rapid change within the indigenous communities (Ridenour and Callaway, 2001). The introduction of plant species by humans has experienced a significant increase over the past five centuries, particularly during the twentieth century, as a result of the rapid expansion of global trade and travel. Modern modes of transportation, such as planes, ships and other means, have facilitated both intentional and unintentional movement of species, often leading to unforeseen and occasionally catastrophic consequences (Moore, 2004). In certain instances, these species are introduced into environments that are unsuitable for their growth and establishment, without the ability for the species themselves to select their preferred habitats. The introduction of new species into balanced ecosystems and habitats can disrupt natural processes, resulting in the destruction or loss of biodiversity (Louda *et al.,* 2003). The introduction of *Eucalyptus citriodora* Hook., *Populus deltoides* Marsh. and *Lantana camara* L. species to India serves as examples of human-induced invasions (Kohli *et al.*, 2004; Dogra *et al.,* 2009).

The impacts of natural invasion are similar to those caused by human-made invasions, with the success of the invasion largely dependent on the dispersal abilities of the invading flora and fauna. The time frame for natural invasion can vary from a few years to several years. The sources of natural invasion include birds, animals, water and wind, among others (Herbold and Moyle, 1986). *Ageratum conyzoides* L. and *Parthenium hysterophorus* L. serve as examples of such invasions in India (Kohli *et al.*, 2004; Dogra *et al.*, 2009).

The apparent lack of attention may, to some extent, indicate a lack of concern, as intact tropical forests on the Asian mainland and continental-shelf islands seem to exhibit a certain level of resistance against both plant and animal invasions (Corlett, 2014; Teo, 2003). However, it is important to note that Asian tropical forests are currently facing severe pressures from activities such as clearance, fragmentation, logging and rapid urbanization (Corlett, 2014), all of which are anticipated to render them more susceptible to invasion. The occurrence of spatial and temporal

disturbances may further exacerbate the extent of invasion by alien species in comparison to native species (Maron and Connors, 1996).

Williamson (1996) explained that biological invasions mostly happened in habitats disturbed by human activities, but invasions also happened in natural habitats disturbed by natural processes.

2.2. Studies on Tree Invasion

Until recently, trees have received limited attention in the literature on invasive plants (Richardson, 2014). This is due to their slower growth and longer life cycles compared to most shrubs and herbs, which makes it more challenging to detect and study potential tree invaders. Additionally, there is often a significant time delay between the introduction of tree species and their invasion (Kowarik, 1995; Binggelli *et al.,* 1998).

In forest ecosystems, trees play a crucial role as ecosystem engineers and regulate various ecosystem functions (Reich *et al.,* 2001; Crooks, 2002; Belote and Jones, 2009). Consequently, invasions by woody species can have significant impacts on community functions such as altering primary production, biomass distribution, litter fall and decomposition rates, energy balance, and carbon storage (Richardson and Higgins, 1998; Jackson *et al.,* 2002; Yelenik *et al.,* 2004).

A recent review has presented conclusive evidence that 357 tree species have been identified as invasive in at least one region (Richardson and Rejmanek, in press). The studies conducted thus far have predominantly focused on conifer tree invasions in the Southern Hemisphere (Richardson *et al.,* 1994; Higgins *et al.,* 1996; Richardson, 1998; Simberloff *et al.,* 2010) and broad-leaved deciduous tree invasions in the northern hemisphere (Keay *et al.,* 2000; Rogers and Siemann, 2002; Chabrerie *et al.,* 2008; Cincotta *et al.,* 2009). Furthermore, investigations into tree species invasions have utilized multiple functional traits, including survival and herbivory resistance, germination, growth, biomass accumulation, density and abundance (Siemann and Rogers, 2001; Chaneton *et al.,* 2004). A diverse range of habitats including open fields, grasslands and forests have been invaded by tree species in both the hemispheres and under temperate, subtropical and tropical climates (Lamarque *et al.,* 2011).

When exotic tree species do spread beyond their point of introduction, they have the potential to be more damaging than smaller plants. They create shade, which is a barrier to the re-establishment of many native species (Mascaro *et al.,* 2008) dominate in nutrient cycling, potentially changing the site in a more permanent way (Gomez-Aparico, 2008a; Virah-Sawmy, 2009; Staska *et al.,* 2014) and make the invaded habitat less suitable for arthropods and native vertebrates.

Other reported impacts of some tree invasions include allelopathic suppression of competitors (Gomez-Aparico, 2008b; Lorenzo, 2011), changes in the local fire regime, and reduced stream flow.

Richardson *et al.,* (2014) conducted a study on tree invasion, examining its patterns, processes and the significant challenges faced by researchers and managers. The authors note that tree invasions have become increasingly important in recent decades, with a rise in the number of species involved, the extent of area invaded, the variety of impacts observed, and the complexity of management challenges. As a result, tree invasions are now being studied from multiple perspectives.

Over the past few decades, numerous tree species have become invasive, and many of them are now included in lists of the most widespread and damaging invasive species (Weber 2003; Richardson and Rejmanek 2011). The total area invaded by alien trees in South Africa exceeds 100,000 km^2, which accounts for more than 8 percent of the country's total area (VanWilgen *et al*., 2001). These invasions are primarily concentrated in wetter regions of the country, as well as along river systems, including perennial, seasonal and ephemeral rivers. When considering the major biomes, the fynbos biome, characterized by Mediterranean-type shrub land, is the most heavily invaded by *Pinus, Acacia*, and *Hakea* species in mountainous areas, lowlands and along major river systems (Richardson *et al*., 1997; Cowling *et al*., 1999; Le Maitre *et al*., 2000; Van Wilgen *et al*., 2001). The forest biome is also significantly invaded, although the extent of this invasion has not yet been quantified. In the forest biome, the main invasive species include *Acacia cyclops*, *Acacia mearnsii*, *Acacia saligna*, *Eucalyptus* spp., *Melia azederach*, *Pinus* spp., *Psidium guajava*, *Sesbania punicea*, and *Solanum mauritianum.* (Richardson *et al*., 1997)

The grasslands and savannah biomes are known to be extensively invaded by a variety of species, including *Acacia* spp., *Melia azedarach* and *Jacaranda mimosifolia*, which invade open land, river banks and beds. The *Nama karoo*, a semi-desert shrub land with summer rainfall, is likely the fourth most invaded biome, with *Prosopis* species having invaded at least 18,000 km^2 of low-lying alluvial plains and seasonal and ephemeral water courses. The succulent *karoo*, a semi-desert shrub land with winter rainfall, is also heavily invaded by *Prosopis* spp. (Richardson *et al*., 1997)

The primary reason for the increasing problems with tree invasions worldwide is the rapid increase in human-mediated transport and dissemination of thousands of species for a wide range of purposes, particularly forestry, agroforestry and ornamental horticulture. Vast plantations of alien trees now dominate many regions of the world (Low, 2012). The massive invasion debt created by a century or more of dissemination and planting of exotic trees is now leading to the manifestation and planting of many large-scale invasions. Despite the increasing attention to trees as invasive species worldwide, problems associated with these invasions are becoming more significant and complex (Kowarik 1995).

Lamarque *et al.,* (2011) conducted a quantitative review and meta-analysis to estimate the significance of eight prominent hypotheses in explaining tree invasions. Numerous hypotheses have been proposed to elucidate the success of introduced plants (Hierro *et al.,* 2005). A species may become invasive if its entry into the community is facilitated by native species (theory of facilitation; Bruno *et al.,* 2003), if natural or anthropogenic disturbances impact the community (disturbance hypothesis; Mack *et al.,* 2000), or when resources fluctuate over time and space (theory of fluctuating resource availability; Davis *et al.,* 2000) or vacant niches are present (empty niche hypothesis; Mack, 1996; Levine and D'Antonio, 1999). Conversely, functional traits have been extensively studied to predict invasiveness potential (Sakai *et al.,* 2001). For example, certain exotic grass and tree species have been found to produce a high number of seeds (propagule pressure hypothesis; Williamson and Fitter, 1996; Lonsdale, 1999) or employ allelopathic compounds against native species (novel weapons hypothesis; Callaway and Aschehoug, 2000).

In the absence of natural predators, plant species can also experience a rapid expansion in their distribution and population size (known as the enemy release

hypothesis; Keane and Crawley, 2002). This phenomenon is accompanied by evolutionary changes, as plants allocate their resources previously dedicated to defense towards growth and reproduction (referred to as the evolution of increased competitive ability hypothesis; Blossey and Notzold, 1995). Originally proposed to explain the invasion of weedy plants into grasslands (Maron and Vila, 2001), Lamarque *et al.*, (2011) suggested that these mechanisms may also be applicable to tree invasions. In their study, they examined 90 research papers that investigated 45 invasive tree species. Additionally, they assessed whether certain functional traits of these species (such as growth rate, density/cover, germination, biomass and survival) equally contribute to their invasiveness. The findings indicate that several hypotheses related to invisibility or invasiveness was relevant in explaining tree invasions. Moreover, multiple factors contribute to the success of invasive tree species, as more than one hypothesis was supported for a given species. Furthermore, the growth rate of invasive trees emerged as the most effective predictor of invasiveness, suggesting its potential use in identifying potential alien tree invasions.

2.3. Impact studies on Tree invasion

The impact of alien invasive species on biodiversity has been characterized as significant, subtle and typically irreversible, according to the International Union for Conservation of Nature (IUCN, 2000). Throughout the twentieth century, the scale and consequences of biological invasions have escalated rapidly. These invasive species pose direct and indirect threats to biodiversity and ecosystem processes, thereby affecting human well-being. They not only suppress native biodiversity and contribute to local extinctions but also continue to expand their invasive range, leading to irreversible alterations in ecosystems. Consequently, an increasingly diverse range of functions and processes are being affected by these invasive alien species (Rejmanek, 2000).

Invasion by alien plant species affects the dynamics and composition of soil on a wide scale and greatly impacts ecosystem functions such as soil nutrient cycling. Since these impacts result from differences in traits between the exotic and resident species, novel physiological traits such as nitrogen cycling may cause significant alterations in ecosystem function (Yelenik *et al.,* 2007). In many parts of the world,

introduced woody plants have become invasive and have necessitated some form of management.

Economic loss:

A study conducted by David Pimentel (2005) and others at the College of Agriculture and Life Sciences, Cornell University, US shows that over 1,20,000 non-native species of plants, animals and microbes have invaded just six countries - the US, the UK, Australia, South Africa, India and Brazil. The study concludes that 4,80,000 alien species have been introduced into ecosystems worldwide. While not all alien invasive are harmful - only think of rice or wheat – the study calculates that such species today cause more than US $314 billion worth of damage every year.

The resources needed to control invasive plants can be substantial. For example, in 1991, the Australian Federal Government provided $2 million to mechanically and chemically control *Mimosa pigra* to halt its spread and protect a nearby World Heritage site from invasion (Miller *et al.*, 1992). The risks associated with the invasion of alien species are increasing, with increasingly rapid international exchange and convenient transportation (Chen and Xu, 2001).

Invasive alien species have been found to accelerate the decline of species and genetic biodiversity (Li and Xie, 2002; Wan *et al.*, 2002), disrupt the structure and functioning of ecosystems (Zhang and Ye, 2002), and result in significant economic losses. In fact, the United States has suffered losses amounting to USD 138 billion as a result of invasive alien species by the year 2000 (Pimentel *et al.*, 2000). Another analysis estimated that the total economic and environmental cost of invasive aliens in Southeast Asia was approximately US$33.5 billion per year, with nearly 90% of these costs attributed to the agricultural sector (Nghiem and Soliman, 2013).

In China, the economic losses caused by invasive alien species reached USD 14.45 billion, with direct and indirect economic losses accounting for 16.59% and 83.41% of the total economic losses, respectively by the year 2003 (Xu and Ding, 2003). Oerke *et al.,* (1994) calculated that weeds alone resulted in 13% reduction in global agricultural output (based on eight major crops). Specifically, in maize production, the economic loss due to weeds from 1997 to 1999 amounted to approximately 1.7 billion USD.

Environment loss:

There has been an extensive movement of plant species around the world by humans as a consequence of trading activities. Inconsequently, exotic species form a significant part of the agricultural weed flora, and in natural ecosystems, invasive weeds are almost exclusively alien (Groves *et al.,* 2001). Large parts of the world are currently dominated by human-modified ecosystems that often comprise a greater biomass of introduced than native organisms (Vitousek *et al.*, 1997).

In addition to human actions, there are several other factors that contribute to the successful invasion of alien plants. The climatic and edaphic similarities between the original and new habitats play a critical role in the establishment of alien species (Holdgate, 1986). Therefore, the humid tropics of Asia and Africa, which have highly leached soils, bear resemblance to Latin America, which is the natural habitat for species such as *L. camara*, *A. conyzoides*, *E. odoratum*, *E. adenophorum*; *P. hysterophorus* and *M. micrantha*. These similarities enable these species to invade and colonize suitable locations on these two continents (Ramakrishnan, 1991).

Invasive species have the potential to disrupt native plant community structure, decrease foliage availability for herbivores, reduce species diversity and degrade ecosystem function (Mack *et al.*, 2000; Hejda *et al.*, 2009; Powell *et al.*, 2011; Sankaran *et al.*, 2014; Joshi *et al.*, 2015; Thappa *et al*, 2016; Schirmel *et al.*, 2016; Early *et al.*, 2016; Bellard *et al.*, 2016; Das *et al.*, 2019).

Thousands of alien species are known to establish worldwide, and many more introduced species remain undetected or unrecognized (Ruiz *et al.*, 2000). Their invasion causes a wide range of high-impact and high-profile impacts, including the decline in the population of threatened and endangered species, habitat alteration and loss, increased frequency of fires, shifts in food webs and nutrient cycling, and loss of crops and productive lands. So, plant invasions are a potent force of change, operating globally and affecting many dimensions of society (Wilcove *et al.*, 1998; Ohlemuller *et al.*, 2006). Given the wide range of impacts of plant invasions as mentioned above, comprehensive studies on a long-term basis are required at a global scale.

According to Macdonald and Richardson (1986), invasive species have been linked to the escalation of soil erosion rates. For instance, in South Africa, various

species such as *Acacia mearnsii, A. longifolia, A. saligna, Sesbania punicea and Pinus pinaster* have expedited the erosion of riverbanks. These non-native species are still well-suited to the flooding patterns in the fynbos biome, making them susceptible to being uprooted by swift waters and carrying away native vegetation mats. This process exposes mineral soil and intensifies erosion rates. Additionally, the presence of introduced stands of *Pinus* spp. in South Africa, which typically have sparse ground cover, can also contribute to increased erosion. In certain regions of the United States, it has been noted that the presence of *Centaurea maculosa* (commonly known as spotted knapweed) stands is correlated with an increase in surface run-off and sediment yield, in contrast to areas predominantly occupied by bunch grass (Lacey *et al.,* 1989).

Invasive species have detrimental effects on socioeconomic, cultural and human health aspects as they impact all four categories of ecosystem services: supporting (such as altering succession patterns and soil and nutrient cycling), provisioning (including threats to native species and alteration of genetic resources), regulating (such as changes in pollination services and fire regimes, and acting as vectors of diseases), and cultural services (such as effects on ecotourism and changes in perception of the landscape) (Millennium Ecosystem Assessment, 2005; Pysek and Richardson, 2010; Vila *et al.,* 2010).

Integrated strategies to mitigate the current and future impacts of biological invasions are being implemented in various regions of the world (Pysek and Richardson, 2010). In Europe, the research project DAISIE (Delivering Alien Invasive Species Inventories for Europe), funded by the European Union in 2005, marked the first international effort to compile an inventory of alien species that pose a threat to European terrestrial, freshwater and marine environments (Hulme *et al.,* 2009).

Bambaradeniya *et al.,* (2002) conducted a study to assess the extent of *Prosopis juliflora* within the boundaries of Bundala National Park in Sri Lanka. The researchers documented the detrimental effects of this plant species on the distribution and ecology of large mammals and wading birds. Similarly, Maehr (2005) observed that the prevalence of a hybrid cord grass in the coastal areas of California led to

decline in the population of migrating and resident shorebirds, as it obstructed the tidal mud flats.

In another study by Bambaradeniya *et al.,* (2006), the spread of *Opuntia dillennii* in coastal scrubland and seashore habitats in Sri Lanka was investigated. The invasive nature of these plants resulted in the loss of nesting habitats for globally threatened marine turtles, which annually visited these areas, and hindered the regeneration of coastal vegetation.

Pimentel *et al.,* (2005) conducted research that demonstrated the overall harmful impact of invasive alien species on native flora, fauna and ecosystem functioning in the regions where they were introduced. Additionally, it was observed that these invasive species caused socio-economic damages, including the introduction of diseases to agriculture, alteration of landscapes, and changes in climatic patterns. In summary, invasive species play a significant role in the loss of biodiversity in a complex manner (Blackburn *et al.,* 2019; IPBES, 2019).

As per the findings of Sajeev *et al.* (2012), the consequences of invasive species also encompass the displacement of indigenous plant species, preferential treatment of pollinators towards non-native species leading to reduced reproductive success of local flora, alteration of soil chemical composition, modification of hydrological regimes, increased susceptibility of the new habitat to fire, and reduction in photosynthetic efficiency of local species due to diminished light availability.

Additionally, the introduction of invasive species also results in changes to the phylogenetic and functional diversity of the invaded communities (Blackburn *et al.,* 2019; Brooks *et al.,* 2004; Suarez and Tsutsui, 2008; Ricciardi *et al.,* 2013). However, there remains a dearth of quantitative studies that examine how the impacts of invasive species vary depending on the ecosystem and the specific invader (Levine *et al.,* 2003).

2.4. Studies of Invasive Alien Species in India

In recent decades, a number of Forest Invasive Species (FIS) have been introduced to India, either knowingly or unknowingly, without fully comprehending the potential consequences. These FIS can be classified into three categories: floral (including weeds and plants with national and regional distribution), entomological

(insects) and pathogenic (fungi). Sankaran and Suresh (2013) have compiled a comprehensive dataset on invasive plants found in the region's forests, identifying a total of 111 FIS falling within the aforementioned categories.

In India, similar to other regions across the globe, invasive species have made their way into the ecosystem through various means. It has been observed that the majority of alien plant species that are known to be invasive in Protected Areas in India were initially introduced into the country as garden ornamentals (Sankaran and Suresh, 2013). Furthermore, invasive alien species were intentionally introduced to meet fuel wood requirements, prevent desert spread, and for commercial cultivation purposes (Randerson, 2003).

The most extraordinary, and potentially unverified instance of a deliberate introduction is that of *Mikania micrantha*, a climbing plant renowned for its rapid growth in humid tropical habitats. It is believed that *M. micrantha* was intentionally introduced by the Allied Forces during World War II to conceal airfields constructed along the Indo-Burmese border as a defensive measure against the advancing Japanese Forces (Randerson, 2003). On the other hand, an example of an inadvertently introduced species that has become invasive is *Parthenium hysterophorus*. Reports indicate that it entered India as a contaminant of imported wheat in the mid-1950s, although there is evidence suggesting its presence in India as early as 1810 (Paul, 2010).

Reddy (2008) has compiled a comprehensive inventory of invasive species in India, many of which have become naturalized. These species have been utilized for various purposes, including medicinal, religious, furniture and composting applications.

Abraham and Abraham (2005) conducted a pioneering study on the spread and control of invasive plants in Kerala. Utilizing a risk assessment protocol and field surveys, Sajeev *et al.*(2012) identified 38 alien invasive species in Kerala's forests, with 10 posing high risk, 12 posing medium risk, 10 posing low risk, and six being insignificant. The alien invasive species in Kerala's forests include five trees, 11 shrubs, four sub-shrubs, 12 herbs, and six climbers.

Invasive species commonly exhibit high levels of success and abundance, while numerous native species tend to be scarce. Additionally, invasive species often possess distinct characteristics in comparison to non-invasive species. For instance, many invasive plants demonstrate broad climatic tolerances and extensive geographic ranges (Rejmanek 1995; Goodwin *et al.*, 1999; Qian and Ricklefs, 2006), which can potentially influence their responses to climate change. Furthermore, invasive plant species frequently possess traits that facilitate swift range shifts, including low seed mass and rapid maturation (Rejmanek and Richardson, 1996).

2.5. Patterns and process of the spread ecology of invasions

Numerous woody plant species have been introduced to various tropical regions. However, only a small percentage (approximately 1%) of these species has successfully spread into new habitats, and even fewer have become significant weeds. Weber *et al*., in 2008, conducted an analysis of the diversity and ecological insights of current invasive alien plant species in China, identifying 270 species belonging to 59 families. Akter and Zuberi (2009) conducted a study of invasive alien species in Northern Bangladesh, recording the 21 most abundant species in the region.

Pysek (1998) conducted a study of the alien and native flora of Central European urban regions, reporting an average of 259 alien species and 386 native species. The author also found that city size, human activities and temperature were factors that contributed to the growth of invasive flora. Lowe, *et al*, (2000) recorded the world's worst invasive alien species, identifying *Lantana*, *Luecaena, Mimosa, Wedelia, Mikania* and others as the most problematic invasive plant species globally**.**

According to Wright (2011), the proliferation of invasive species poses a significant risk to both global and local beta diversity. Furthermore, beta diversity is widely recognized as a crucial element of ecosystem well-being, and the presence of invasive species leads to a decline in beta diversity, resulting in the homogenization of species within ecosystems.

Several studies have been done to characterize invasive plants from different regions of India (Khuroo *et al.,* 2012). India hosts hundreds of different invasive species; recent inventories of Indian alien flora have identified more than one-third of them as originating in South America (Khuroo *et al*., 2012). Invasive alien plants have been established in several habitats and in different climatic regimes across India

(Khuroo *et al.,* 2012). Their persistent nature and adaptability has made it easy for them to grow and spread rapidly across diverse ecosystems in India (Khuroo *et al.*, 2011).

In their study, Reddy *et al.,* (2008) examined the invasive flora of India and documented a total of 173 invasive alien species, which were classified into 117 genera across 44 families. The majority of these species were found to be adaptable to various habitats, including forests, agricultural lands, wastelands, plantations, gardens and roadsides. The author posited that the impact of invasive alien species on native biodiversity surpassed that of environmental pollution, and was identified as the primary factor contributing to global biodiversity decline.

Singh *et al.,* (2010) conducted a study on the invasive alien flora of Uttar Pradesh, examining their history of introduction and uses. The researchers documented a total of 152 invasive alien species, which were classified into 109 genera under 44 families. In a separate study, Negi and Hajra (2007) investigated the alien flora of Doon Valley in the Northwest Himalaya. They recorded a total of 308 woody and 128 herbaceous exotic species from the Mega Himalayan 'Hotspot Belt'. Sood *et al.* (2011) focused their research on the distribution and ecology of alien plants in the temple courtyards of Himachal Pradesh. The authors documented 159 species, which were classified into 133 genera under 63 families. Similarly, Sekar (2012) recorded 190 invasive alien species from the Himalayan Region. These species belonged to 112 genera, which were further classified into 47 families.

Aravindhan and Rajendran (2014) conducted a study on the diversity of invasive plant species in Boluvampatti Forest Range. They recorded a total of 90 invasive alien species, which were classified into 74 genera under 37 families. Shah and Reshi (2014) reported the alien aquatic flora of Kashmir Himalaya. 129 alien plant species belonging to 68 genera and 42 families were recorded. Among this, 69.1% were emergent, 15.6% were rooted floating leaf types, 10.1% were submerged, and 7% were free floating types.

Srivastava *et al.,* (2014) studied invasive alien species in the terrestrial vegetation of North-Eastern Uttar Pradesh and recorded 149 species belonging to 100 genera under 41 families. Naidu *et al.,* (2015) studied the invasive alien plant species

in the Eastern Ghats of Northern Andhra Pradesh. The authors documented 87 invasive alien plant species under 73 genera, belonging to 32 families.

In their study, Prabhu *et al.,* (2014) examined the eco-physiological characteristics of *Mikania micrantha* at two distinct elevations in the tropical regions of the Western Ghats. Their findings revealed a significant level of phenotypic plasticity in this invasive weed, with the species exhibiting greater success at lower elevations.

Wijesundara (2010) highlighted the potential impact of climate change on the expansion of invasive plant species. Mihulka (1998) conducted research on the influence of altitude on the patterns of plant invasions. The author observed a decrease in both the number and proportion of alien plants as altitude increased.

Chytry *et al.,* (2005) reported that the presence of alien species was relatively low in habitats characterized by dense vegetation, while habitats with open vegetation exhibited a higher number of alien species.

The Western Ghats biodiversity hotspot region is particularly vulnerable to these invasive species due to its climatic similarity with the Central American landscape (Kohli *et al.* 2006; Rao and Sagar, 2012), which makes it remarkably easy for species from the Central and South American continents to establish (Khuroo *et al.,* 2012). Other studies on invasion in this region have reported vulnerability to disturbance, such as forest fragmentation (Joshi *et al.,* 2015) and roads (Prasad, 2009).

According to Amor and Stevens (1976), the frequency of alien plants in forests is directly correlated with the reduction in diffuse light. Gentle and Duggin (1997) proposed that resource availability is enhanced by canopy openings, which modify the microclimate, in line with the disturbance patch invasion model. Denslow *et al.* (2001) explained that most invasive alien species require light for growth, and therefore, canopy openings in forests promote the spread of such species.

Sharma and Ragubanshi (2010) reported that *Lantana* cover decreases with increasing canopy cover, and they observed no *Lantana* growth in areas with high tree canopy cover. The availability of light on the forest floor was also identified as a crucial factor that promotes the growth of invasive plants.

The spread of invasive species has stimulated interest in predicting their distribution. Many studies have been undertaken to map the distribution of invasive species (Anderson *et al.*, 1993; Mc Cormick, 1999). Typically, such maps display the presence and absence of an invader. Ecological knowledge is required to assess their density and impact. Distribution maps have been used for a long to acquire such knowledge. For example, species climate models have been used to assess the risk of further spread of invaders (Rouget *et al.*, 2004). Studies show that spatial information is needed to develop policies aimed at invasive species management (Wittenberg and Cock, 2001). Effective species management must be based on a thorough knowledge of these species' locations and distributions, modes and rates of spread, potential and known effects, and control methods. Random surveys on current distribution and abundance is an important tool of impact assessment (Crosier and Stohlgren, 2004).

Estimating the potential distribution and abundance of an invasive species necessitates the utilization of comprehensive data integrated with a geographic information system (GIS). By employing these tools, a "habitat suitability map" can be created to depict the susceptibility of habitats to invasion by the target species (Chong *et al.,* 2001).

Medeiros (2004) conducted an investigation into the key life history attributes of three invasive species, encompassing phenology, seed dispersal, seed predation, and establishment sites in the Hawaiian Islands and other locations. This study provided crucial information for establishing priorities and devising strategies for the control of invasive plants. In essence, it contributes to a more comprehensive comprehension of the life history attributes of invasive plants.

2.5.1. Determining the Distribution and Density of Invasives

The initial step in the development of an effective management programme should involve the mapping of the current and potential range of the species, as well as its density and size class distribution. However, it is important to exercise caution in allocating excessive time and resources to this step, as it may allow the population to expand significantly before control measures are implemented. The level of details required for this assessment should be determined by its purpose.

Ludwig and Reynolds (1988) and Magurren (1988) have established standardized methodologies for conducting vegetation analysis in forest ecosystems.

To assess the extent of invasive species in a forest area, a vegetation study is necessary. Data will be analyzed to determine the frequency, density and abundance of the species, drawing upon the studies of Curtis and McIntos (1950) and Mishra (1968). Ground vegetation analysis in the Muthanga and Tholpetty forest ranges of Wayanad in Kerala was conducted using a random systematic design and grad sect (Barbour *et al.,* 1999; Sing and Sing, 1992). Satellite images can be particularly useful on a large scale, especially in cases where an alien woody plant has heavily invaded extensive areas of treeless vegetation. When studying the current and potential spread of a species, it is necessary to model the geographic distribution of the species. In the case of this study, where the species is invasive, geographical modeling is crucial in order to halt its ongoing spread.

2.5.2. Species Distribution Models (SDMs)

In recent times, predictive modelling has advanced rapidly. Several tools and techniques have emerged which show robust model development using multiple input parameters, as well as prove effective predictors of the species to be modelled (Austin 2002; Gallien *et al.,* 2012; Jiménez-Valverde *et al.,* 2011; Thuiller *et al.,* 2009). Environmental (or ecological) niche modelling (ENM), habitat modelling, predictive habitat distribution modelling, and range mapping are examples of SDMs (Elith and Leathwick, 2009) commonly used in ecological and biodiversity conservation research to represent how species are distributed worldwide throughout a geographic area. These models accommodate the tools that incorporate known species occurrences with environmental data (Philips *et al.,* 2006).

There are two types of species distribution models (SDMs), namely correlative and mechanistic. Correlative SDMs aim to predict the impact of climatic variations on the spatial distribution of data (Thomas *et al.,* 2004). These models analyze statistical records of environmental associations with species abundance and occurrence in order to identify the factors that impede the species' spread.

According to Moilanen and Wintle (2007), correlative SDMs are considered superior to other SDMs due to their simplicity and ability to describe complex environmental interactions with limited data. On the other hand, mechanistic SDMs, also referred to as biophysical models or process-based models, seek to establish the

connection between a species' physiology and its surrounding environment, which in turn influences its abundance and distribution (Kearney and Porter, 2009).

Understanding and managing invasive species is crucial for determining their current and potential distribution. The development and implementation of preventive measures to address invasive species is a top priority in the conservation of biodiversity (Hulme, 2006). Preventive measures are more cost-effective than measures aimed at controlling or eradicating invasive species (Leung *et al.*, 2002). In this context, Ecological Niche Models (ENMs), also referred to as Bioclimatic Models, Climate Envelopes, Habitat Models, Species Distribution Models, Range Maps and Resource Selection Functions (Elith and Leathwick, 2009), have been utilized to forecast the potential distribution of exotic species (Jimnez-Valverde *et al.,* 2011).

ENMs are developed using data from a species' native habitat. They are employed to identify suitable areas for the establishment of invasive species in a new region (Peterson and Vieglais, 2001). Models can also be constructed using data from both the native and invaded areas to predict the potential distribution of invasive species (Broennimann and Guisan, 2008).

These models are created using various modeling techniques and integrate species occurrence records (geographical coordinates of the occurrence records) with a set of predictor variables (such as climate, land use type and salinity). Models are used to forecast suitable habitats in which species can maintain a population and persist over time (Guisan and Thuiller, 2005; Mateo *et al.,* 2011).

Modelling methods can be categorized into two groups based on the type of occurrence-records input utilized for creating the models. The first group consists of methods that rely on presence-only records, such as BIOCLIM and DOMAIN. The second group includes methods that utilize both presence and absence records, such as logistic regression and generalized additive model (GAM) (Tsoar *et al.,* 2007). Some methods, such as Genetic Algorithm for Rule-set Production (GARP) and Maximum Entropy (MAXENT), employ pseudo-absence data for model construction. However, despite the use of pseudo-absence data, these methods are still considered to fall under the category of presence-only records, as absence records are not incorporated in the model construction process (Tsoar *et al.,* 2007).

2.5.3. Maximum Entropy Modelling (MaxEnt)

MaxEnt is a general purpose machine learning method with accurate mathematical computations introduced by Phillips *et al. (*2006) for modelling the spatial distribution of species. For modelling species habitat, it uses the maximum entropy method. MaxEnt uses a set of environmental variables as input, such as temperature, precipitation, etc., along with the species occurrence data and obtains a range of given species, i.e., it executes by finding out the maximum spread (maximum entropy) by estimating the probability distribution for the species in a geographic dataset to the 'background' environmental layers (Philips *et al.,* 2006).

MaxEnt is used for modelling the species distribution and the range using the presence-only data utilizing both continuous and categorical data. MaxEnt estimates the suitability of each grid cell as a function of the grid cell's environmental variable. The grid with a high value is likely to have optimal conditions for species occurrence. MaxEnt outperforms the other modelling approaches (Elith *et al.,* 2006; Hernandez *et al.,* 2006; Phillips *et al.,* 2006; Ortega-Huerta and Peterson, 2008).

MaxEnt, according to Phillips *et al.,* (2006), adopts the maximum entropy distribution. For estimating species distribution, the data were subjected to the constraint that the expected value of each environment parameter (interactions) in the estimated distribution matched its empirical average. It approximated the most uniform distribution using background locations and data-derived constraints (Phillips *et al.*, 2004; Phillips *et al.,* 2006).

If presence-only species data were used, the complexity of the fitted functions could be chosen in this model. According to Pearson *et al.,* (2007), MaxEnt has a higher success rate than other algorithms, and it could identify differences even with limited sample sets. When sample sizes were artificially reduced, the model performance worsened.

MaxEnt models have demonstrated the ability to project a wider range of suitable conditions, including the capability to anticipate areas that are typically excluded (Pearson *et al.,* 2007). However, it has been observed that the performance of MaxEnt models can be enhanced through species-specific parameter tuning (Radosavljevic and Anderson, 2014). MaxEnt has the capacity to generate intricate and non-linear response curves by utilizing various feature classes such as linear,

quadratic, threshold, hinge, product and categorical, which are determined by the number of presences by default (Syfert *et al.,* 2013). In addition to the feature class, the Regularization Multiplier is another adjustable parameter in MaxEnt. This parameter imposes constraints on the model, acting as a penalty. The primary objective is to minimize over-complexity and over-fitting by adjusting the intensity of the selected feature classes used in model creation (Morales *et al.,* 2017).

Several researchers have reported the variability in predictions that might result from different MaxEnt background samples, with a particular focus on the extent of the site from which they are chosen (Baasch *et al.,* 2010; Giovanelli *et al.,* 2010; Barve *et al.,* 2011). The raw output, which is interpreted in terms of occurrence rate, the cumulative output, interpreted as omission rate, and the logistic output are the three types of outputs derived from MaxEnt models. However, the difference in scaling between these three types of outputs is critical in creating various prediction maps (Merow *et al.,* 2013).

In general, these modelling methods combine species locality data (geo-referenced coordinates of latitude and longitude from confirmed presence) with environmental variables to create a model of species requirements for the examined variables (Anderson *et al.,* 2003). The resulting model is then projected onto a GIS map (termed a habitat suitability map) of the study region, showing the potential geographic distribution of a species. For invasive species management, habitat suitability maps identify areas where (1) invasive species may actually be present (but are as yet undetected), and (2) where invasive species may disperse in the future, thus providing assistance for planning and prioritizing areas for surveillance. Such information can also assist in determining the extent, cost and likelihood of success of a control programme.

Predictive model of species distribution is an important tool for managing invasive species (Anderson *et al.,* 2003). Recent advancements in geospatial and statistical modelling methodologies, along with the growing availability of species data, have enabled Species Distribution Models (SDM) to increasingly tackle a range of pressing ecological problems, such as managing rare and endangered species and predicting species responses to climate change and human modifications of habitat structure (Guisan and Thuiller, 2005).

Due to globalization and extensive land transformations that facilitate the transfer and establishment of non-native organisms, SDM methods are also being increasingly used to predict spatial patterns of biological invasions and prioritize locations for early detection and control of invasion outbreaks (Peterson and Vieglais, 2001; Fonseca *et al.*, 2006; Lippitt *et al.,* 2008; Meentemeyer *et al.,* 2008; Strubbe and Matthysen, 2009).

Invasive species distribution models face several challenges because the ecological theory and assumptions underlying SDMs typically do not apply to invasive species. One of the challenges is that, by definition, the assumption of equilibrium between organisms and their environment is violated, and potential dispersal limitations of the invader are often ignored. As most SDMs implicitly rely on ecological niche concepts (Grinnell, 1917; Hutchinson, 1957), they assume that the species occur at all locations where the environmental conditions are favourable and that dispersal is not a limiting factor (Jeschke and Strayer, 2008). However, invasive species are often absent at particular locations not because of low habitat quality but because the species has not dispersed to that site due to stochastic events, geographical barriers and dispersal constraints (Higgins *et al.,* 1999; Araujo and Pearson, 2005; Araujo and Guisan, 2006). Although dispersal limitations, more than biotic interactions, stochastic events or abiotic factors, are known to play a major role in the spread of invasions (Hastings *et al.*, 2005; Soberon and Peterson, 2005; Araujo and Guisan, 2006), only a few studies to date have tested empirically the benefits of including dispersal constraints in invasive SDMs (Meentemeyer *et al.*, 2008).

The effectiveness of species distribution models can be summarized into two categories: first, these models can be used to detect the occurrence of rare species in remote regions where systematic surveys had not been conducted (Elith, 2002; Pearce *et al.,* 2001); second, habitat change mapping can be very crucial in assessing the direct impact of anthropogenic pressure on existing habitats in terms of land use, land cover and climate change (Johnson *et al.,* 2004). These models could also be used to predict future species distributions under various climate change scenarios (Jeschke and Strayer, 2008; Sinclair *et al.*, 2010), potential expansion of introduced species in newly colonised areas (Jimenez-Valverde *et al.,* 2011; Jeschke and Strayer, 2008), and reserve planning (Thorn *et al.*, 2009). Stolhgren *et al.* (2010) advocate that species distribution modelling can help with risk assessment and conservation. Guisan

and Zimmermann (2000) also discussed a range of species distribution modelling approaches that can be used to predict a species' potential suitable habitat.

The environmental conditions are defined using known species distributional information, resulting in identifying geographical regions with similar environments and modelling species distribution (Pearson and Dawson, 2003). If the spread of a species is accurately mapped, environmental variables such as climate could be correlated to its presence or absence (Crick, 2004).

Species distribution modelling studies (Shreshta and Shreshta, 2019; Shrestha *et al.,* 2018; Thapa *et al.,* 2018) the changing climate would create additional climatically suitable areas for IAS in Nepal in the future. Furthermore, the study of Adhikari *et al.,* (2019) also predicted a further and continuous increase in the current and future potential habitats for invasive plant species in the different provinces of the Republic of Korea due to climate change. A significant niche expansion was observed in the study of Banerjee *et al., (*2019), which suggested that the species may be able to colonise new areas in India, which were also consistent with the results of the SDM study of invasion hotspots of Adhikari *et al.*, (2015).

Peterson and Vieglais, (2001) used ecological niche modelling to address the difficulties in anticipating possible species invasions. Wan *et al.,* (2018) have modelled 36 invasive alien plant species identified as the world's worst invasive species. Numerous studies have examined the risks of invasive plant species spreading across a region with a lot of plant diversity (Bradley *et al.,* 2010; O'Donnell *et al.,* 2012; Adhikari *et al.,* 2015; Peknicova and Berchova-Bimova, 2016). Jimenez Valverde *et al.*, (2011) investigated the use of niche models in the risk assessment of invasive species.

2.5.4. Advancements in Ensemble Modelling

The employment of an ensemble approach is anticipated to diminish model uncertainty and enhance its robustness in accurately modeling species distributions, as stated by Marmion *et al.,* (2009) and Thuiller (2003). However, Kaky (2020) has demonstrated that MaxEnt can perform and predict comparably well over an ensemble method that combines several well-known and highly regarded algorithms to highlight important habitats for Egyptian medicinal plant conservation in his study. These findings do not necessarily imply that MaxEnt is a superior technique to other

approaches, and there are still instances in which it is ineffective, as noted by Guillera-Arroita *et al.,* (2014). Nevertheless, when modeling species distributions from insufficient data, MaxEnt may be considered one of the most efficient and accessible methods, as supported by Abdelaala *et al.,* (2019); *Fois et al.,* (2018); Kaky and Gilbert (2019), *Kaki et al.,* (2020); Phillips *et al.,* (2006); Koo *et al., (*2015); Dullinger *et al.,* (2017); Deb *et al.,* (2017); Lamsal *et al.,* (2018); Thapa *et al.,* (2018) and Shrestha and Shrestha (2019).

MaxEnt possesses numerous advantages over other models. For instance, it allows for the utilization of presence points as input species data, accommodating both categorical and continuous environmental layers. Even when working with small sample sizes, MaxEnt consistently and reliably produces predictions with a high degree of accuracy. It is capable of predicting the distribution of threatened species and generating spatially explicit maps that depict habitat suitability in a manner that is easily interpretable. Additionally, MaxEnt permits the execution of replicated runs to assess the robustness of the model. Irrespective of the threshold rule employed, the jackknife test can be employed to ascertain the significance of each environmental variable. The MaxEnt model, also known as the bioclimatic envelope model, can be employed to project into the future under climate change scenarios, enabling the prediction of habitat losses and gains within species ranges. This, in turn, facilitates the planning of appropriate conservation measures. (Elith *et al.,* 2011; Fois *et al.,* 2018; Pearson *et al.,* 2007; Phillips *et al.,* 2006; Padalia and Bahuguna, 2017; Abdelaala *et al.,* 2019).

2.6. Phenology and Reproductive Biology of Invasive Species

In order to establish self-sustaining populations, exotic species must demonstrate successful reproduction in novel environments. As such, the reproductive characteristics and success of such species are critical factors in their ability to invade. Notably, certain trends in plant traits have been identified as being associated with invasive plants. Specifically, traits related to seedling emergence, growth form, growth rate, breeding system, dispersal, and environmental tolerance are key in predicting the likelihood of a given species becoming invasive (Thuiller *et al.,* 2006; Pysek and Richardson, 2007; Van Kleunen and Johnson, 2007).

In general, research indicates that phenological studies pertain to the timing of recurring biological events. In the case of plants, these events encompass reproductive processes such as bud formation, flowering, fruiting and seed germination, as well as vegetative processes such as leaf flushing and shedding. The impact of plant phenology on animal populations is often significant, as it can result in temporal changes in resource availability. Furthermore, phenological schedules may be influenced by biotic factors such as competition, herbivory, pollination, and seed dispersal, in addition to various climatic variables (Sakai *et al.,* 1999).

Floral display (i.e. flower number and arrangement) and phenology are important factors determining the attractiveness and composition of pollinator guilds (Kunin, 1997; Feldman, 2006). Congeneric pairs of invasive and native species are good models to investigate competition for pollinators, as they often share floral and/or ecological traits. Such models have already been used to disentangle biological or life history traits linked to invasion success (Goodwin *et al.,* 1999; Gerlach and Rice, 2003; Sans *et al.,* 2004).

Invasive plants are exotic species introduced in new areas that reproduce and disperse efficiently to such an extent that they spread rapidly, and it is increasingly clear that interactions with resident biota often play a role in the invasive success (Richardson *et al.,* 2000). Pollination has received less attention among these interactions (Bjerknes *et al.,* 2007). For insect-pollinated exotic species, reproductive success critically depends on attractiveness to local pollinators. Some authors proposed that some exotic plants became invasive due to their high attractiveness to pollinators (Brown *et al.,* 2002).

A study of invasive Australian Acacias explained that this species share several reproductive traits and that may contribute to their invasiveness: massive and long-lasting floral displays, generalist pollination syndromes, precocious production of a large number of long-lived and highly viable seeds resulting in massive seed banks, seed dispersal adaptations and a positive response to disturbance (e.g.: resprouting ability or mass germination) (Milton and Hall, 1981).

Invasive species generally have a high sexual reproductive capacity, the ability to reproduce asexually, rapid growth from seed to sexual maturity, a great dispersal and colonization efficiency, a high tolerance to environmental heterogeneity and

disturbances, a high adaptation to environmental stress (phenotypic plasticity) and a greater competitive capacity than native species (Sakai *et al.,* 2001; Vila and Weiner, 2004, Werner *et al.,* 2009).

An investigation into the primary means of dispersal of the alien plant species should be undertaken to ascertain whether this presents the optimal opportunity to manage the invasion. Numerous traits impact the survival and reproductive fitness of plants, but flowering phenology is particularly valuable for examining natural selection and constraints on adaptive evolution in novel and changing environments for at least two reasons. Firstly, natural variation in flowering phenology is frequently strongly associated with plant fitness (Munguia-Rosas *et al.,* 2011). Therefore, understanding how the timing of key phenological events interacts with climatic and biotic factors to influence survival and reproduction is of fundamental interest for comprehending plant adaptation. Secondly, flowering phenology is determined by various growth and life-history characteristics, which can restrict an adaptive response to selection (Rathcke and Lacey, 1985; Bernier and Périlleux, 2005). Given that flowering phenology is subject to strong natural selection, it is unsurprising that numerous species exhibit phenological shifts that are correlated with climate change (Fitter and Fitter, 2002; Cleland, 2007).

The evolution is immediate and unimpeded by genetic drift, gene flow or a lack of standing genetic variation. These factors can constrain evolutionary responses to selection (Yeaman and Guillaume, 2009) and may favour invasive species over native species with sparse populations (Munguía-Rosas, 2011).Thus, incorporating delayed evolutionary responses would potentially slow the evolution of flowering time and reduce reproductive fitness in native populations more than in introduced populations.

Previous studies have shown that successful invaders generally display early flowering or long blooming periods (Goodwin *et al.,* 1999; Pysek *et al.,* 2003). Also, in alien–native comparisons, many authors have found that invasive alien species flower earlier than natives (Cadotte and Lovett-Doust, 2001; Lake and Leishman, 2004). Those results suggest that invasive species capitalize on an early blooming strategy to increase their reproductive success since the chance to acquire improved

fitness via effective pollination visits is also increased (Goodwin *et al.,* 1999; Pysek, *et al.,* 2003).

Multiple studies have established a positive correlation between the flowering phenology of non-native species and their potential for invasiveness (Goodwin *et al.,* 1999; Pysek *et al.,* 2003; Lake and Leishman, 2004; Cadotte and Lovett-Doust, 2001). However, the differences in flowering phenology between invasive species and native species are merely a result of varying historical introductions orchestrated by humans. The findings demonstrate that invasive alien species, when subjected to the same climatic conditions in geographically distinct regions, do not exhibit a uniform flowering phenology pattern. Instead, their flowering occurs earlier, later, or concurrently with native species, depending on the climatic conditions in the region where they originated.

The attainment of reproductive success is a crucial factor in the colonization of novel territories and the establishment of sustainable populations over the long term. In light of this, self-compatible plants possess an advantage in successfully establishing themselves in new ranges, as their reproductive capabilities are less restricted by population size and pollinator availability. Consequently, self-compatible plants are anticipated to exhibit greater invasiveness than obligate outcrossing plants, as per the findings of Baker (1955) and Gibson *et al.,* (2011). While out-crossing may prove advantageous for the evolution of invasive plants, when feasible, the ability to produce autonomous seeds, which does not necessarily preclude out-crossing, is likely to be indispensable during various stages of the invasion process, as suggested by Van Kleunen and Johnson (2007).

2.6.1 Pollination Ecology

Pollen is a highly coveted floral reward that serves as a crucial source of sustenance for numerous insects, particularly Apidae larvae, various beetles, flies (notably syrphids and anthomyiids), thrips, springtails, and select orthopteroid and butterfly species (Anderson, 1996). This nutrient-rich substance contains both essential and non-essential amino acids, making it highly nutritious (Haydak, 1970). Additionally, starch and lipids may serve as alternative energy reserves utilized during pollen germination and pollen tube growth (Inouye, 1980).

In entomophilous plants, self-incompatible species rely entirely on pollinator services and the availability of mating partners to achieve sexual reproduction. Conversely, self-compatible species have the ability to self-pollinate (either autonomously or not) and ensure seed production even in the face of poor pollination services and/or limited mate availability (Eckert *et al*., 2006).

The capacity to produce seeds after self-fertilization, even at a low rate, is significant in the early stages of naturalization and invasion because it reduces the need for pollinators and compatible plants (Baker's law) (Baker, 1955; Davis *et al.,* 2004). Indeed, numerous invasive plants have been described as self-compatible in the introduced ranges (Rambuda and Johnson, 2004; van Kleunen and Johnson, 2007; Stout, 2007; Rodger et al., 2010; Hao *et al.*, 2011), and this has been proposed as an advantage for successful invasion (Williamson and Fitter, 1996; Pannel and Barret,1998) and also, plant species that are pollinator generalists may have an advantage in the introduced ranges compared with specialist species, because they can be pollinated by a wide variety of floral visitors (mutualist facilitation hypothesis) (Richardson *et al.*, 2000).

Invasive plants are widely acknowledged to have detrimental effects on the ecosystems they invade, leading to a loss of biodiversity and changes in ecosystem functioning (Mack *et al.*, 2000). However, the existing studies have not universally supported this theory; the impacts are likely to be highly dependent on the specific context and can vary based on the characteristics of the invasive species and the community being invaded (Pysek *et al.*, 2012).

In the past years, there has been a particular emphasis on examining the interactions between invasive plants and their mutualistic partners, particularly pollinators (Stout and Morales, 2009; Schweiger *et al.*, 2010). The majority of studies conducted thus far on the mutualistic relationships between invasive plants and pollinators have focused on the indirect effects of plant invasion on native plant pollination. These studies have tested hypotheses regarding competition and facilitation between native and invasive plants (Traveset and Richardson, 2006; reviewed in Morales and Traveset, 2009).

In a study of Acacia species, Stone *et al.,* (2003) assert that pollination followed by successful seed production is crucial for plant invasion; however, these aspects remain largely unexplored for most Acacia species.

In general, these studies have primarily demonstrated the negative impacts of invasive plants on the reproduction of co-flowering native species, especially when the invasive plants are more abundant (Morales and Traveset, 2009; Dietzsch *et al*., 2011). However, these effects vary depending on the characteristics of the invader (Thijs *et al.,* 2012), the scale of the studies conducted (Jakobsson and Traveset, 2009), and the specific context. Fewer studies have focused on the direct effects of invasive plants on native flower visitors or the role played by native pollinators in plant invasion.

The pollination ecology of a few species has been studied in both native and invasive ranges, for example, *Hedysarum coronarium* in Spain, in populations where it is native on the mainland, and where it has been introduced to the Balearic islands (Montero-Castano *et al*., 2014); *Rhododendron ponticum* in native populations in Spain and where it is introduced and highly invasive in Ireland (Stout *et al.,* 2006); and *Nicotiana glauca* in its native habitats in South America and across its introduced range globally (Ollerton *et al*., 2012). In all cases, pollination in the non-native range is carried out by resident native pollinators belonging to similar functional groups as those pollinating plants in their native ranges.

Hedysarum coronarium is pollinated by certain species of Hymenoptera and Coleoptera, including the honey bee *Apis mellifera*, in both its native and introduced areas (Montero-Castaño Vila and Ortiz-Sánchez, 2014). *R. ponticum* is pollinated by the large bees *Xylocopa violacea* and *Bombus* spp. in Spain, and *Bombus* spp. in Ireland (Stout *et al.,* 2006). *N. glauca* is pollinated by hummingbirds in its native range and specialized sunbirds in the parts of its invasive range where they occur (Ollerton *et al.,* 2012). However, there is still much to learn about the pollination ecology of most species in both native and non-native regions, even those that have received significant scientific and public attention. For instance, *Impatiens glandulifera*, a well-known invader in northern temperate regions, is pollinated by various bee species in its invasive range (Valentine, 1978; Stary and Tkalcu, 1998; Nienhuis and Stout, 2009). However, little is known about its pollinators in its native

regions of India, Pakistan and Nepal, except that *Bombus* spp. visit it for food (Saini *et al*., 2013). The role of pollinators in facilitating invasion in non-native ranges is not well-studied. The mutualist facilitation hypothesis suggests that the introduction of lost mutualists, including pollinators, from a plant's native range to its non-native range is crucial for the establishment and spread of invasive plants (Richardson *et al.,* 2000). Invasive plants such as *Carpobrotus* spp., *Mimosa pigra*, and *Lantana camara*, which offer abundant floral resources and have extensive or prolonged floral displays, can have a significant negative impact on the reproductive success of native plants if they are chosen by pollinators (Trveset and Richardson, 2006).

Seed production is essential for establishing self-sustaining populations and the subsequent naturalization of introduced species. However, seed production depends on pollination ecology, the introduced plants' breeding system, and the recipient area's environmental conditions (Richardson *et al.,* 2000). Thus, floral traits linked with the functioning of the flower and (in) dependence on the pollinator, as well as with pollinator attraction, will determine the final reproductive success of the plant.

A study of Stout (2017) explains that the impacts of invasive plants are likely to be plant species specific and ecological context-specific; however, by designing appropriate studies (Kumschick *et al.,* 2015), and incorporating more knowledge of plant and insect species traits (including plant breeding system, pollination syndrome, nectar chemistry, insect body size and diet breadth), better prediction of impacts may be achieved. Thus, more studies of invasive plants and flower visitor ecology are required before generalizations about direct impacts can be made. Many studies have addressed the attractiveness of invasive species for bees (Parker, 1997; Detzel and Wink, 1993).

The study conducted by Harmon-Threatt and Kremen (2015) demonstrates that honey bee responses to plant invasions, including behavior, health and abundance, exhibit significant variability. This variability, whether negative, positive, or neutral, poses challenges in comprehending and predicting the impact of these invasions on pollination networks and bee conservation. On one hand, invasive plants have the potential to harm bee populations by altering the diversity and relative abundance of native plant species. Certain bee species may be unable to forage or

thrive on alternative plants, including invasive species, due to behavioral and physiological limitations such as flower handling, host recognition, toxin occurrence, and nutrient deficiency.

On the other hand, invasive plant species can become integrated into the diet of bees, such as *Impatiens glandulifera*, thereby providing new resources. The local abundance and diverse presence of generalist bee species on some invasive plants can be attributed to their extensive, accessible and visually appealing floral displays. However, foraging on a new host plant, particularly one that becomes dominant in the plant community, can also impact the overall quality of the bee's diet and disrupt the cost-benefit balance of their foraging activities. This impact is contingent upon various bee traits, including physiological abilities such as digestion and social behavior (Harmon-Threatt and Kremen, 2015)

2.7. Genetics of Invasive Alien Species

Biological invasions present interesting evolutionary problems because they are stochastic events often involving small populations that can survive rapid habitat transitions (Kruger and May 1991). Genetic effects of invasive species are defined as alterations to the gene pools of native species, usually through hybridization or introgression (Kruger and May, 1991).

Hybridization is the interbreeding of individuals from distinct species or genetically distinct populations. The young produced through hybridization may or may not be sterile. If they are sterile, they waste native gametes and reduce native reproduction. If they are not sterile, they may compete with the native parent species and reduce their survival or reproduction. Non-sterile hybrids may also backcross, leading to introgression, producing gene flow between species. The impact of most concern comes from combining hybrid vigor and preferential backcrossing of the hybrid young with the invasive parent species. In this context, the hybrid has increased fitness over the native and the slow trickle of reproduction that the native does manage is increasingly tainted by genes from the invader. This can lead to the loss of genetic integrity and the extinction of the native genotype one gene at a time. Loss of genetic diversity during introduction and subsequent range expansion might limit local adaptation possibilities in the introduced region (Lambrinos, 2004).

Evolutionary processes, such as genetic drift, migration, and/or multiple introduction events, could form novel genotypes and increase standing genetic variation on which natural selection could act, leading to the evolution of locally adapted genotypes (Kolbe *et al.*,2004,;Shirk *et al.*, 2014). Several studies have shown that invasive species are often able to adapt very rapidly (Maron *et al.*, 2004; Montague *et al.*, 2008; Monty *et al.*, 2013) and expand their range (Colautti and Barrett, 2013). Adaptation to local environmental conditions results in genetically based phenotypic differentiation (Kawecki and Ebert, 2004). Also, along latitudinal gradients, populations are more scattered, and gene flow between populations is likely low. Limited gene flow and increased inbreeding often lead to higher genetic differentiation (Kawecki and Ebert, 2004). Because of genetic differentiation, morphological and phenological traits of native plant populations show clinal variation to associated environmental variation along latitudinal gradients. When invasive populations show clinal variation parallel to that of native populations, it is suggested that adaptive evolution occurs in the invasive species (Kollmann and Banuelos, 2004; Maron *et al.*, 2004; Colautti and Barrett, 2013).

Hybridization plays a crucial role in shaping the dynamics of invasive populations. The inter-or intraspecific hybridization of invasive populations with native or other non-native populations has the potential to mitigate the loss of additive genetic variance during founder events and generate novel genotypes (Ellstrand and Schierenbeck, 2000). Numerous studies have provided evidence of the positive effects of hybridization on invasibility, including accelerated growth, increased size, and heightened aggression. These effects may be attributed to the increased genetic variance, new gene interactions, masking or elimination of deleterious recessive alleles, or the transfer of advantageous genes. Hybridization can also facilitate adaptation, such as the acquisition of herbicide resistance in weeds from genetically engineered crops (Snow *et al.,* 1999) and potentially the transfer of cold tolerance. However, it is important to note that hybridization can be detrimental in many cases, and successful invasions are likely occur through the selection of numerous hybrid combinations (Ellstrand and Schierenbeck, 2000).

Information on the invasive populations and their level of genetic diversity compared to source populations is needed to test different evolutionary and ecological hypotheses used to explain the success of invasive species (Muller-Scharer *et al.*,

2004; Bossdorf *et al.*, 2005; Prentis *et al.,* 2008). Identifying the origin of introduced populations is important for determining if invasions arose from single or multiple sources. At the same time, quantifying differences in genetic variation between the ranges will establish if a genetic bottleneck occurred during introduction. Genetic data can also provide means to test whether admixture among genetically differentiated source populations following multiple introductions or founder effects following genetic bottlenecks can facilitate or inhibit invasiveness.

Determining both the origin of introductions and the amount of genetic diversity present also has implications for management such as the selection of biological control agents (Garcia-Rossi *et al.,* 2003; Mu¨ller-Scha¨rer *et al.*, 2004; Goolsby *et al.,* 2006; Zalucki *et al.,* 2007). Populations of invasive species suffer from genetic bottlenecks after introduction because the number of initial colonists is often small (Nei *et al.*, 1975). Moreover, sequential bottlenecks during colonization and introduction would further reduce genetic diversity (Barrett and Kohn, 1991; Clegg *et al.*, 2002, Estoup *et al.*, 2004). Thus, introduced populations are typically less genetically diverse than their source populations (Amsellem *et al.*, 2000, Estoup *et al.*, 2004, Dlugosch and Parker, 2008). This, along with high risk of inbreeding depression resulting from a population-level bottleneck, may limit the population growth (Allendorf and Lundquist, 2003) and the introduced species' ability to adapt to new environments (Kinziger *et al.*, 2011). In spite of this, many invasive species that have gone through population bottlenecks are successful, a phenomenon commonly referred to as the genetic paradox of invasive species (Allendorf and Lundquist, 2003; Frankham, 2005).

Molecular genetic investigations have also demonstrated that invasions can result in rapid adaptive evolution despite facing significant bottlenecks (Amsellem *et al.,* 2000; Dlugsch and Parker, 2008). Furthermore, it has been observed that successful invasions may involve "bridge head effects," wherein widespread secondary invasions originate from a specific successful invasion population (Lombaert *et al.,* 2010).

The capacity of a population to adapt to changing environmental conditions is determined by its genetic variability (Fisher, 1930; Sakas *et al.,* 2001). Therefore, genetic variability plays a crucial role in determining the potential of a population to

become invasive (Lee, 2002; Kolbe *et al.,* 2006). Reports have indicated an increase in genetic diversity in invasive species such as *Phalaris arundinacea* L. in North America (Lavergne and Molofsky, 2007), *Geranium carolinianum* in China (Shirk *et al.,* 2014), and *Ambrosia artemisifolia* in France (Genton *et al.*, 2005).

2.7.1. Molecular Marker Studies in Invasive Alien Species

Molecular markers can be classified into two categories: homozygosity, also known as dominant markers, and heterozygosity, also known as co-dominant markers (Hartl, 1988). Dominant DNA markers, such as Inter-simple sequence repeats (ISSR) (Zietkiewicz *et al.,* 1994), Random Amplified Polymorphic DNA (RAPD) (Williams *et al.,* 1990), Microsatellites (SSR) (Akkaya *et al.,* 1992), arbitrarily primed polymerase chain reaction (AP-PCR) (Welsh and McClelland, 1990), and Amplified Fragment Length Polymorphisms (AFLP) (Becker *et al.,* 1995), have been commonly used in genetic diversity studies in plants.

The Inter simple sequence repeat (ISSR)-PCR technique is a molecular method that utilizes microsatellite sequences as primers in a polymerase chain reaction to generate multi-locus markers. This technique combines the advantages of microsatellites (SSRs) and amplified fragment length polymorphism (AFLP) with the universality of random amplified polymorphic DNA (RAPD). ISSR markers are highly polymorphic and have proven to be valuable in various areas of research, including genetic diversity, phylogeny, gene tagging, genome mapping, and evolutionary biology (Reddy *et al.,* 2002).

According to Sanger *et al.*, (2001), the ISSR technique addresses the limitations of RAPD, such as low reproducibility, and the high cost associated with AFLP. Additionally, ISSR allows for the development of species-specific primers for SSR polymorphism by utilizing flanking sequences. Tikunov (2003) further states that ISSR primers, which are longer (15 to 20 bp), have a higher annealing temperature, resulting in better repeatability compared to RAPDs. In China an alien invasive plant Mexican sunflower (*Tithonia diversiflia*) showed high level of genetic diversity compared to native species and lowest genetic diversity estimates were represented by two regions by using ISSR markers and karyotype analysis (Yang *et al.*, 2012).

Karyotyping is the technique to identify and evaluate the size, shape and number of chromosomes in a sample. Chromosomal variation is widespread in plants

and animals. It often contributes to the genetic barriers to gene flow that exist between species and hence its role in species diversification has been heavily debated (White, 1978; King, 1993; Rieseberg, 2001).

2.8. Morphological Variation Studies of Invasive Alien Species

Most invasion ecology studies relate traits of alien species to their capacity to invade, with the overall aim of unraveling aspects of the invasion process and aiming to predict future invasions. However, not all the observed plants traits identified as being associated with invasiveness in aliens really confer invasiveness, since other causes often underlie the observed pattern (Goodwin *et al.*, 1999; Pysek *et al.*, 2003).

Important traits associated with invasiveness of plant species listed in the literature are physiological traits and say that aliens have higher photosynthetic capacity, more efficient nitrogen and water use, longer flowering period, and higher specific leaf area (SLA). Aliens have lower root-shoot ratios, *i.e.*, they put more resources into above ground biomass, their growth rate is high, are taller and have higher biomass, have larger and/or fleshy fruits, and have higher values for traits related to number of flowers or seeds, germination, survival, and/or mortality (Lake, *et al.,* 2004, Van Kleunen *et al.,* 2010; Lamarque *et al.,* 2011). In the case of specific traits like the ability to release allelopathic compounds that are novel to native habitats and in terms of biotic resistance aliens are more successful. They are resistant to herbivory can reproduce vegetatively and they show high phenotypic plasticity and as a result they can acclimate to changing environments (Wilsey, 2006; Pysek *et al.,* 2006; Hufbauer, 2007). Successful invasive plants often enhance their performance and competitive ability by improving resource acquisition and use in their introduced ranges (Zou *et al.*, 2007; Feng *et al.*, 2009; Chen *et al.*, 2013; Liu *et al*., 2017; Petruzzellis *et al.*, 2019).

Several studies have carried out comparisons between leaf morphological physiological and biochemical traits of plants from native and introduced populations (Feng *et al.,* 2009; Angelo and Pau, 2017; Liu *et al.,* 2017; Petruzzellis *et al.,* 2019). In a study conducted by Li *et al.,* (2020), an invasive Chinese tallow tree (*Triadica sebifera*) was examined. The researchers analyzed the concentrations of carbohydrates (soluble sugar, sucrose, fructose, starch, and cellulose) as well as the mass of roots, stems and leaves. They also investigated root water potential and the colonization of

arbuscular mycorrhizal fungi (AMF) in soil-cultured *T. sebifera* seedlings from 10 native (China) and 10 introduced (United States) populations in a common garden setting. The results showed that introduced populations had significantly greater stem and leaf mass compared to native populations, while their root masses did not differ. Additionally, introduced populations exhibited higher soluble sugar concentrations but lower starch and cellulose concentrations in their leaves, stems and roots. These populations also displayed more negative root water potentials and higher AMF colonization. The findings suggest that invasive plants alter their carbohydrate allocation, resulting in faster growth and a greater above ground allocation strategy. The higher AMF colonization and more negative water potential in invasive plants likely contribute to more efficient water absorption by the roots. Therefore, the researchers propose that such physiological variation in root characteristics may play a role in the success of plant invasions.

According to Dematteis *et al.,* (2020) extent of morphological variation in invasive populations across a diverse range of environments plays a crucial role in determining their performance and shaping forces. In their study, they investigate the impact of natural selection on invasive populations of *Senecio madagascariensis* in Argentina. The researchers conducted morphometric analyses on the invasive populations of *S. madagascariensis*, specifically measuring leaf area, head number, and the length of internodes and inflorescence, using material collected over a span of 54 years (1962-2016). They then compared these measurements between populations at the edge of the range and those in established areas.

The results revealed that while there were differences in all the measured plant traits among the sampled areas, only leaf area showed statistically significant variation. This indicates that the populations in the edge and established ranges exhibit different responses to the same environmental pressures. The lack of significant differentiation among the areas can be primarily attributed to gene flow, which acts to homogenize allelic frequencies and restrict adaptation to a heterogeneous environment (Coulleri, 2010).

The expression of phenotypes is a consequence of the interaction between genetic elements and environmental pressures. It is expected that neighboring areas will exhibit greater morphological similarity due to the exchange of genes between

them (Dematteis *et al.,* 2020). The phenotypic diversity of invasive species can be influenced by the number of times they are introduced to a new region, as this increases the genetic variations (Kramer and Kozlowski, 1960; Walker, 2003). For instance, studies have shown that the number of introductions of *Phalaris arundinacea* L. to North America's invasive range has led to an increase in both phenotypic and genetic diversity. These introduction events have stimulated rapid evolution and phenotypic adaptability, facilitating the expansion of their range (Durka, 2005). Similarly, the multiple introductions of *S. madagascariensis* in Brazil serve as a genetic resource that can contribute to higher variability in dispersal traits among the invasive populations (Mäder, 2016).

2.9. *Senna spectabilis,* A tree invasive and a growing threat of forest and wild life

Senna spectabilis (DC.) H.S. Irwin & Barneby (Fabaceae; Caesalpinioideae) is commonly utilized in the semi-arid region of northeastern Brazil, where it is referred to as "canafístula" or "cassia," for the purpose of providing fodder to sheep and goats. Additionally, it is utilized as a source of fuel wood and timber. *Senna spectabilis* has been identified as a casual, spontaneous, exotic species that is capable of surviving outside of cultivation, but does not form self-replacing populations, and instead relies on repeated introduction or limited asexual reproduction for persistence (Chong *et al.,* 2009).

The Wayanad Wildlife Sanctuary is recognized as one of the aggressive growth habitats of *S. spectabilis*. It was initially introduced to the sanctuary in the early 1980s and has since expanded to encompass approximately 23% of the sanctuary's total area over the course of 40 years (Anoop *et al.,* 2021). A survey conducted by the Wildlife Trust of India and the Forest department has revealed that *S. spectabilis* is a significant presence in the Muthanga, Sulthan Bathery, and Tholpetty range of forests within the sanctuary (APFISN, KFRI, 2014). Despite having escaped from Trinidad and Tobago and invaded the northern parts of Orinoco in Venezuela (Irwin and Barneby, 1982), *S. spectabilis* is not recorded in the Global Invasive Species Database (2021).

This is a tree of medium to large size that originated from tropical America. It grows up to 60 feet of height, although it is often smaller. The subfamily Caesalpinioideae (Fabaceae Lindi.) holds a basal position in phylogenetic trees and

exhibits significant diversity in floral form and ontogeny, as reported by Doyle *et al.,* (2000) and Bruneau *et al.,* (2001). This subfamily is typically classified into five tribes, namely Ercideae, Caesalpineae, Cassieae and Detarieae, with Macrolobieae (derived from Detarieae) recently added, according to Tucker (2003). This classification has been supported by various taxonomical studies of these genera, including analyses of seed proteins (Guareeth *et al.,* 1999), morphological, vegetative and reproductive characters (Irwin and Barneby, 1981; 1982; Gottsberger and SilberBauer-Gottsberger,1988; Owens and Lewis,1989; Dulberger *et al.,* 1994; Boonkerd *et al.,* 2005), ontogenetic characteristics (Tucker, 1996), molecular systematics (Doyle *et al.,* 2000; Bruneau *et al.,* 2001), and cytogenetics (Goldblatt,1981; Biondo *et al.,* 2005a; 2005b).

The chromosome number of Senna species are 2n = 22, 24, 26 and 28 (Irwin and Turner 1960; Bandel, 1974; Coleman and Demenezes, 1980; Goldblatt, 1981, Irwin and Barneby, 1982; Souza and Benko-Iseppon, 2004; Biondo *et al.,* 2005a; 2005b). Cytogenetic studies of Senna described chromosomal number, karyomorphology and meiotic behavior of some species (Souza and Benko-Iseppan, 2004; Biondo *et al.,* 2005a; 2005b). Nevertheless, little is known about the intra and interspecific variation for these cytological characteristics. In Minas Gerais state Brazil, a group carried out investigation (Katia *et al.,* 2013) and recorded the difference in chromosome number (n= 12, 13, 14 and 28). Mohanty *et al.,* (2006) conducted a study on the karyotype analysis of a native specimen of *Senna spectabilis* collected from the Royal Botanical Garden, KEW. The results revealed that the chromosome number of this specimen is 2n = 28.

This particular species is known for its rapid growth, abundant flowering and seed production, as well as its ability to readily re-sprout when cut (Mungatana and Ahimbisibwe, 2010). It was introduced as an ornamental plant in botanical gardens in India, but has been reported to escape cultivation and establish itself in forest areas of Sikkim and Mysore (Singh, 2001). Its distribution in India includes Karnataka (Mysore), Kerala (Wayanad), Sikkim (Rishikha), Tamil Nadu (Coimbatore and Sathyamangalam), and West Bengal (Howrah) (Satyanarayana *et al.,* 2013).

In Australia, *Senna spectabilis* is considered naturalized and has been observed as a weed in natural environments. It has also been found to escape

cultivation and is labeled as an invasive species due to its ability to rapidly spread and form monocultures (Randall, 2007). The species exhibits early flowering and seed production, with the seeds remaining viable for up to three years. Additionally, it possesses strong coppicing ability (Kerala Biodiversity Board, 2012). When cut, the species quickly, abundantly and persistently re-sprouts. While *S. spectabilis* is non-nodulating, it efficiently accumulates nitrogen and can even deplete soil nitrogen reserves, leading to its classification as a nitrogen-fixing tree (Kerala Biodiversity Board, 2012). However, a study by Ladha *et al.,* (1993) concluded that *S. spectabilis* is unable to fix atmospheric nitrogen and instead relies on its extensive root system to extract nitrogen from deep soil horizons.

It is suspected that the plant in question possesses allelopathic properties, although it does not demonstrate allelopathy towards maize or rice. Additionally, the growth and regeneration of indigenous tree species are impeded by *S. spectabilis*, as documented by Wakibara (1998) and Wakibara and Mnaya (2002).

S. spectabilis thrives in areas with a mean annual temperature of 19-22^{0}C, with a mean maximum temperature of the hottest month ranging from 23-32^{0}C and a mean minimum temperature of the coldest month ranging from 14-17^{0}C. The species requires full sunlight and an annual precipitation ranging from 800-2000 mm (CABI, 2021).

According to Mungatana and Ahimbisibwe (2010), *S. spectabilis* is considered an invasive alien species in Uganda, posing a significant threat to the native flora. Similarly, Oviedo-Prieto *et al.,* (2012) classify it as an invasive species in Cuba. In the Mahale Mountains National Park in western Tanzania and the Budongo Forest Reserve in central Uganda, *S. spectabilis* has successfully expanded beyond its boundaries, infiltrating the protected areas and establishing itself as an invasive species. It now dominates both the tree canopy and understory in these forested regions, as documented by NARO (2009).

James V Wakibara, (1998) a researcher from Kyoto University in Japan, conducted a study on the ecological implications of the invasion of *S. spectabilis*. The study focused on comparing the effectiveness of girdling and total cutting as control methods for *S. spectabilis* at Mahale National Park. This invasive species has a

tendency to easily establish itself in arborescent forests, especially in disturbed habitats.

In a separate study, Richard (2011) aimed to improve the preservation of pollen germination in *S. spectabilis*. The researchers chose this invasive tree species because it plays a significant role in the bee population and is at risk of extinction in the Adamawa region of northern Cameroon. The study also aimed to assist beekeepers in their efforts. The researchers tested the possibilities of *in vitro* germination and storage of pollen. The results indicated that the pollen of *S. spectabilis* prefers to germinate in a medium enriched with 25% sucrose at a concentration of 38.36 BK (Brewbaker medium). Furthermore, the stored pollen remained viable for up to 22 weeks when stored at temperatures of 10 and -20 degrees Celsius.

Zembele and Ngulubeto (2022) conducted an evaluation on the impact of pre-treatment methods on the germination of *Senna spectabilis* seeds. Additionally, they sought to determine the influence of these pre-treatment methods on the growth of *Senna spectabilis* seedlings, specifically in terms of height and root collar diameter. This particular tree species exhibits dormancy caused by its seed coat, resulting in delayed germination. Given its classification as an agroforestry species and its wide range of applications, it serves as a valuable source of energy (in the form of firewood and charcoal), forage for honeybees, and provides shade and shelter in homesteads and other agroforestry systems. The shade it provides also aids in improving soil water conservation in certain agroforestry system applications. The study conducted by Zembele and Ngulubeto revealed that pre-treatment methods significantly enhance the germination process and seedling growth of *Senna spectabilis* seeds. Due to its hard seed coat, *Senna spectabilis* requires more time to germinate, resulting in a lower germination percentage during nursery establishment. However, effective pre-treatment methods can ensure the successful germination and growth of this agroforestry tree species among farmers. Among the various treatments applied in the experiment, the most effective method for *Senna spectabilis* was found to be nicking and hot water soaking for 24 hours, as it resulted in faster germination and accelerated seedling growth.

In a laboratory study conducted by Prajitha and Sudhabai (2022), it was demonstrated that the seeds of *S. spectabilis* exhibit a high level of dormancy. However, through the process of scarification using concentrated H_2SO_4 or water incubation for 24 hours, germination could be induced. Subsequent examination of soil samples from the invaded region indicated that the soils possess an acidic nature and possess a significant capacity for retaining water. The findings of this study highlight the substantial risk posed by the invasion of *S. spectabilis* to the native plant diversity. Additionally, the edaphic factors present in the invaded areas contribute to the extensive sprouting observed.

2.9.1. Genetic Diversity Studies on *Senna* Species

Roberts *et al.,* (2016) conducted a study in which they developed SSR markers for *Senna spectabilis* var. *excelsa* with the primary objective of assessing the genetic diversity and structure of this species. Two types of microsatellite libraries were constructed: an Inter Simple Sequence Repeat (ISSR) library and a microsatellite-enriched library. Primers were designed using Primer3Plus software (Untergasser *et al.,* 2007) based on specific criteria, including a product size range of 100-300 bp, primer melting temperature (Tm) of 50-60°C (with a maximum difference of 2°C), and GC content of 40-60%. PCR was performed using the appropriate primers, with the reaction conditions consisting of a pre-melt at 94°C for five minutes, followed by 35 cycles of denaturation at 94°C for 15 seconds, annealing at 55°C for 35 seconds, and extension at 72°C for 90 seconds. A final extension step at 72°C for seven minutes was then carried out. The resulting amplicons were visualized using agarose gel electrophoresis (1.5%) with a 100-bp ladder. The SSRIT software was utilized for the identification of simple sequence repeats, and the microsatellites were analyzed using GeneMapper 4.0 (Applied Biosystems). The polymorphism information content (PIC) was calculated using online tools provided by the Centre for Genomic Research at Liverpool University. The researchers successfully developed polymorphic microsatellite markers for *S. spectabilis* var. *excelsa,* which could also be employed for evaluating the genetic diversity of germplasm collections.

A study conducted by Mohanty *et al.,* (2010) revealed that the primers SSR, ISSR, RAPD and SSR were used to evaluate genetic variability and evolutionary

relationships between twenty eight Cassia species. The presence or absence of DNA bands in the gel may be used as RAPD markers to study inter- and intra-specific genetic variations, genetic closeness, and identification of specific genes and patterns of gene expression. In the RAPD analysis, 21 selected primers were used for PCR amplification and PCR performed. The reaction condition was: initial denaturation for seven minutes at 94^0C, followed by 45 cycles of 30 secs at 94^0C, 45 secs at 52^0C and 2 min at 72^0 C, with a final extension at 72^0. In ISSR analysis, microsatellite markers have been produced and PCR has been done using different ISSR sequences. In SSR analysis, both forward and backward (20- and 20-mer) primers were designed. These primers are flanking sequences of the SSR repeat. In this experiment they concluded that the amplification products produced by either RAPD, SSR or ISSR primers were not common in all the species. Based on SSR, RAPD and ISSR data they constructed three dendrograms. Due to diversification and genetic polymorphism, some species of Cassia show wide range of adaptation.

Mao *et al.,* (2017) conducted a study on the identification of seeds of *Senna obtusifolia* and *Senna occidentalis* using molecular markers and secondary metabolites. This study helped to determine the differences and similarities between *S. obtusifolia* and *S. occidentalis* seeds and their characteristics were evaluated by high performance liquid chromatography (HPLC) and three molecular markers. Dendrogram was constructed based on bands. They concluded that, intra-specific similarity was 99.79 & 100.0% for 20 *S. obtusifolia* and 16 *S. occidentalis* and inter-specific similarity was 89.58%. Lal *et al.,* (1998) collected 49 accessions of Senna from Uttar Pradesh, Gujarat, Maharashtra, Tamil Nadu, Andhra Pradesh and Karnataka states of India and these accessions were grouped into seven clusters based on genetic variability. De Britto *et al.,* (2010) conducted genetic diversity of Cassia populations from different locations of Tirunelveli District of Tamil Nadu using RAPD markers. Eight primers were used for this experiment. In these, five primers were found to be the most informative and produced thirty-five polymorphic bands.

RAPD markers were utilized to assess the genetic diversity of *Senna spectabilis* accessions (Santos *et al.,* 2013). The genetic diversity between eight accessions of *S. spectabilis*, which were present in the forage germplasm collection of Embrapa Meio-Norte, was evaluated using RAPD markers. These accessions were identified as CAN.1, CAN.2, CAN.4, CAN.5, CAN.6, CAN.7, CAN.8, and CAN.9 in

the germplasm collection of Embrapa Meio-Norte. Initially, 100 RAPD primers were analyzed in this study, out of which 15 primers exhibited high degree of polymorphism and generated a total of 107 bands. Among these bands, 59 (55.14%) were identified as polymorphic bands. This study provided valuable insights into the genetic distance between the accessions. The primers OP-A07, OP-A12, OP-M04, OP-M05, OP-M07 and OP-M15 generated the highest number of polymorphic loci.

2.10. Possible Control Methods for Tree Invasives

The control of invasive species is a challenging and costly endeavour that demands significant time and resources. In the United States alone, billions of dollars have been expended over the past decade to prevent the harm caused by invasive plants. Article 8(h) of the Convention on Biological Diversity (CBD) stipulates that each contracting party should endeavour to prevent the introduction, to control, or to eradicate alien species that pose threat to ecosystems, habitats, or species. The CBD, established in 1992, sets global priorities and guidelines and coordinates international action on invasive alien species, while also collecting information. From a conservation standpoint, addressing climate change is futile if the biodiversity we seek to protect has already been lost to invasive species. The rapid increase in the number of invasive plant species and the extent of invasions at multiple scales has raised concerns about the stability of ecosystems. Therefore, the management of invasive species necessitates a hierarchical approach that prioritizes prevention, followed by control and eradication, based on preference, resources, and capacity.

Some invasive trees provide valuable products or services to society, and when eradication is not feasible, management options should be identified that balance their positive and negative aspects. The invasion of alien species could be further exacerbated by a lack of awareness and insufficient information on the species. Typically, the management of invasive species necessitates repeated treatments and long-term monitoring to ensure that the invasive species does not reestablish itself after treatment. Extensive research has been conducted on the negative impacts of invasive plants on native flora diversity and production. When a species is already established and naturalized, three management strategies may be appropriate, as outlined by Carter (2000) and Grice (2009): 1) eradication for recently established species or species with a limited distribution; 2) containment for species which are

beyond eradication (or where eradication has been rejected as a goal) but still in an early stage of invasion and expanding their range; and 3) control for large and extensive populations, which may include biological control, referred to as "sustained control" (*sensu* Parkes and Panetta, 2009) or "maintenance control" (*sensu* Hulme, 2006). Alternatively, doing nothing is also an option. Eradication is defined as the "removal of all individuals of a species from an area to which reintroduction will not occur" (Myers and Bazely, 2003) or the "permanent removal of discrete populations" (Parkes and Panetta, 2009). Eradication is dependent on the area over which the weed is distributed and must be repeatedly searched for following control (gross infestation area), as well as constraints such as site accessibility, plant detectability, the species' characteristics, control efficacy, and funding support (Panetta and Timmins, 2004; Parkes and Panetta, 2009).

Containment and control are sometimes combined because their common aim is reduction of the density of the target species or its rate of spread. Whether plant eradications are successful depends on the life history traits of the species, including growth rate, reproductive capacities, and dispersal abilities (distance and speed). A major obstacle for plant eradication is the existence of a soil seed bank, which can persist for several years or more. A variety of techniques can be used to remove exotic species from reserves or restoration sites. These most commonly include hand removal, mechanical removal, herbicides, fire, or some combination of the above (Masters and Nissen, 1998).

Accurate spatial and temporal characterizations of pest risk are crucial for making strategic and tactical decisions regarding the management of invasive alien species. The options available for managing biological invasions include prevention, eradication, containment and suppression, as discussed by Venette and Kosh in 2009. Various chemical, mechanical and biological methods are employed to manage invasions, as highlighted in the works of Czarapata (2005), DiTomaso (2000) and Hobbs and Humphries (1995). However, the effectiveness of these methods in removing specific plant invaders while minimizing impacts on native species varies significantly, as demonstrated by the studies conducted by Flory and Clay (2009) and Miller and Miller (2004). Moreover, the use of herbicides, which is often the preferred method for controlling plant invasions, is limited in many systems due to environmental, economic, or social concerns, as noted by Guynn *et al.,* (2004). For

instance, herbicides can have unintended effects on native plants and animals, including those that are threatened or endangered, and the costs associated with large-scale herbicide application are often prohibitive. Therefore, it is imperative to conduct studies that evaluate nonchemical methods, such as fire, hand removal, or mechanical techniques, for removing invasions and assess their impact on native plant communities, as emphasized by Simberloff *et al.,* (2005).

Several other studies have also documented the use of various methods to control invasive alien species, including mechanical, chemical and biological approaches (D'Antonio and Meyerson, 2002; Musil *et al.,* 2005; Witkowski and Garner, 2008; Flory, 2009). In certain dynamic habitats with sporadic natural soil disturbance, uprooting can be a viable strategy (Pickart *et al.,* 1998 a, b). Additionally, uprooting can be combined with the burial of invasive plants in the subsoil, particularly in dunes, scree slopes, and gravel bars. This method is most effective in areas with a limited number of distinct patches of invasive alien plants and easy access for heavy machinery. Furthermore, it is crucial that the target species does not possess a long-lived seed bank that could be activated by uprooting (Ussery and Krannitz, 1998). The benefits of this approach include the absence of herbicide usage, the avoidance of plant material removal, prevention of re-sprouting through sediment layering, and the restoration of habitat dynamics, which would ultimately benefit native species.

Exotic species pose a persistent threat to the preservation of native assemblages for park and reserve managers. This is due to their ability to consume native species, infect them with diseases for which they have no resistance,outcompete them, or disrupt ecosystem functions. Consequently, restoring the ecosystem to its previous, often more desirable, state becomes challenging and costly (Vitousek *et al.,* 1997). In order for decision makers to determine whether management is necessary and, if so, which approach is most suitable, they must be aware of the potential ecological, economic, or social impacts that a species may have on areas outside its native range. Species with the potential to cause harm in a specific area are classified as pests, and their potential can be estimated through pest risk assessment conducted by analysts (FAO 2007).

Developing a management frame work to eradicate *S. spectabilis* and restoring the study area needs an integrated manner of controlling methods which include physical and chemical methods. A study of assessment of the effectiveness of two control options for *Senna spectabilis* in Budongo forest reserve conducted by Peter Beine in 2015 tries to compare ring-barking and the use of Tordon herbicide as possible ways to kill Senna trees *in situ*, with minimal disturbance to the surrounding trees. This is based on the earlier studies on management of invasive trees. In the southern Cape forests, *Olea capensis* and *Canthium obovatum* were killed effectively by ring-barking (Goodland, 1998). Wakibara (2002) working in Tanzania, concluded that girdling worked as a control measure for *Senna spectabilis* but it was costly and time demanding. On the use of chemical control, Gillespie (1991) reported that frill girdling with the herbicide Tordon (2,4-D and picloram) was more effective at killing *Pittosporum undulatum* than cutting and the application of Tordon to the cut stump.

One of the findings that mechanical control of alien plant invasions is by far the most common control method is also surprising as it is presumably also the most expensive. Nunez and Pauchard (2010) argues that developed countries are in the position to allocate funds for sophisticated control methods, while developing countries might have fewer funds but abundant low cost labour, which is a major advantage. On the other hand, Kull *et al.,(*2011) and Wilson *et al.,* (2011) state that control of alien invasions in developing countries is often in direct conflict with uses of invasives (e.g., for restoration of degraded lands or as a resource for poor communities).

Mowing and cutting are effective strategies for reducing seed production and curbing the growth of weeds, particularly in the case of annuals that are cut before they reach the flowering and seed-setting stage (Hanson, 1996). Nevertheless, it's important to note that some species exhibit strong resprouting capabilities when subjected to cutting, replacing one or a few stems with numerous new ones that can rapidly produce flowers and seeds. For instance, yellow starthistle (*Centaurea solstitialis*) can be effectively managed by mowing at the onset of flowering, typically when around 2 to 5% of the seed heads are in bloom. Mowing at an earlier stage can have negative impacts on native species, and yellow starthistle may even resprout (Benefield *et al.,* 1999). It is crucial to consider the specific biology of the weed before implementing a cutting strategy.

Certain plant species can be eliminated by severing or injuring the carbohydrate storage structure situated at the base of the plant. This structure may take the form of a root corm, storage rhizome (tuber), or taproot, depending on the species, and is typically located at the base of the stem and beneath the soil. By cutting off access to these storage structures, certain species can be effectively weakened or starved.

According to the Weed Control Methods Handbook by Tu *et al.,* (2001) published by The Nature Conservancy, various techniques have been compiled for the control of invasive plants. Stabbing has been utilized in Michigan to control the growth of baby's breath (*Gypsophila paniculata*), as reported by McGowan-Stinski (2000). On the other hand, girdling is commonly employed to control trees or shrubs with a single trunk and has been found to be effective against certain species of pines, oaks and maples. This method requires less labour than cutting and removal, and only targets the intended plant, leaving no residue except for the standing trunks. Additionally, a dead standing tree (snag) can provide valuable wildlife habitat, and if left to decay, allows the nutrients of the tree to be returned to the system rather than being removed and deposited elsewhere.

However, it is important to note that a few species, such as black locust (*Robinia pseudoacacia*) and tree of heaven (*Ailanthus altissima*), should not be girdled as they respond by producing many fast-growing root and stem sprouts. Therefore, before employing girdling, it is advisable to determine if the target species responds by re-sprouting. If so, an alternative control technique, such as hack and squirt herbicide applications, should be used. Alternatively, if girdling is chosen, it is recommended to return at 1 to 4 month intervals to cut, burn, or herbicide all re-sprouts for at least 2 years.

Girdling has been successfully used on preserves in New York State to control quaking aspen (*Populus tremuloides*) and bigtooth aspen (*Populus grandidentata*). It can also be combined with herbicides for effective control. For instance, black locust (*Robinia pseudoacacia*) and quaking aspen (*P. tremuloides*) in New York and Wisconsin, respectively, were successfully controlled using girdling with herbicide. However, this method was not successful in controlling tropical ash (*Fraxinus uhdei*) on the Kamakou preserve on Molokai, Hawaii.

Soil solarization is the technique of placing a cover (usually black or clear plastic) over the soil surface to trap solar radiation and cause an increase in soil temperatures to levels that kill plants, seeds, plant pathogens and insects. In addition, when black plastic or other opaque materials are used, sunlight is blocked which can kill existing plants (Katan *et al.,* 1987). Soil solarization however, can cause significant biological, physical and chemical changes in the soil that can last up to two years, and deter the growth of desirable native species. Soil solarization is used in horticulture and for a few high value agricultural crops like strawberries. This method has not been used extensively for weed control in natural settings. The effectiveness of soil solarization depends, in part, on how susceptible weed seeds are to temperature increases. It is the most effective against winter annual weeds that germinate under cool conditions (Elmore, 1990).

Summer annuals and other species adapted to higher temperatures, which germinate during warmer parts of the year, are less susceptible. Soil solarization is the most effective during the summer months, and may be less effective in cooler climates (DeVay, 1990). The higher the temperature, the more quickly a kill is achieved. Solarization is effective only if done in wet soil. Where soils are typically dry, they must first be irrigated until soil from the surface to 50 to 60 cm deep is at field capacity (Grinstein and Hetzroni, 1991). However, solarization leaves an open substrate that can be readily invaded by new organisms, both native and non-native once the plastic is removed (Stapleton, 1990).

In situations where the water level of a wetland or riverine system can be manipulated, flooding can be used to control some plant species. Some species, however, have vegetative buds or underground storage organs that can survive several months or more under flooded conditions. In Vermont, flooding was used successfully to kill seeds and seedlings of common buckthorn (*Rhamnus cathartica*). Flooding was also used in combination with herbicide to successfully control the spread of autumn olive (*Elaeagnus umbellata*) and reed canary grass (*Phalaris arundinacea*) in Ohio. At Wertheim NWR on Long Island, NY, *Phragmites australis* was controlled by burning and then flooding with several feet of water in impounded areas (Tu *et al.,* 2001).

According to Lockwood (2000), biological control, commonly referred to as biocontrol, involves the utilization of animals, fungi, or other microbes to feed upon, parasitize, or otherwise interfere with a targeted pest species. Although touted as a less hazardous option, biocontrol has not proven successful in actual field conditions. The concept of biological control is straightforward: locate a predator of a pest or weed in its original habitat and introduce it to the invaded region. However, stringent parameters must be met, such as host-specificity, to ensure that the predator does not prey on other plants or animals. For example, in 1889, the Australian vedalia lady beetle was introduced to California, USA, to control the cottony cushion scale insect that threatened citrus orchids, and it completely eradicated the pest. Similarly, the introduction of *Dactylopius opuntiae* to control the prickly pear in India has been highly successful. Nevertheless, determining host-specificity is challenging, and numerous examples exist of biocontrol agents themselves becoming pests. The most notorious case is the release of the predatory snail *Euglandin arosea* to control the alien African giant snail in the Hawaiian Islands, which resulted in the decimation of most of the 41 indigenous snail species, while the African giant snail still roams Hawaii's forest floors. Successful biocontrol programmes significantly reduce the pest's abundance; but in some cases, they merely prevent the damage caused by the pest, such as by preventing it from feeding on valuable crops, without reducing pest abundance.

Biocontrol is often viewed as a progressive and environment friendly way to control pest organisms because it leaves behind no chemical residues that might have harmful impacts on humans or other organisms, and when successful, it can provide essentially permanent, widespread control with a very favorable cost-benefit ratio. However, some biocontrol programmes have resulted in significant, irreversible harm to untargeted (non-pest) organisms and to ecological processes. Of course, all pest control methods have the potential to harm non-target native species, and the pests themselves can cause harm to non-target species if they are left uncontrolled. Therefore, before releasing a biocontrol agent (or using other methods), it is important to balance its potential to benefit conservation targets and management goals against its potential to cause harm (Newman *et al.,* 1998).

Classical biocontrol is by far the most common approach for plant pests. Conservation and augmentation approaches show great promise on their own and

especially for enhancing the impacts of classical biocontrol and other weed control measures as researchers and managers focus on managing to maximize native biological diversity in invaded ecosystems (Newman *et al.,* 1998).

Successful classical biocontrol programmes lead to the permanent establishment of the control agent(s), resulting in a long-term reduction in the abundance or at least the damaging effects of the weed within its introduced range. It should be noted that classical biocontrol does not aim to completely eradicate the pest species, and it often takes a considerable amount of time, sometimes even decades, after the initial release of control agents before their effects become evident. There are various reasons why classical biocontrol programmes may fail. Some biocontrol agents may fail to establish themselves, or it may require multiple releases to establish viable populations. Additionally, some biocontrol agents may become established but have minimal or undetectable impact on the targeted pest (Greathead, 1995).

Biocontrol agents typically possess the ability to persist and spread to areas far from their release sites. They may also undergo genetic or behavioral changes that enable them to feed on new hosts. Despite the associated risks, the use of biocontrol has the potential to be an extremely effective tool for managing invasive species. Currently, biological control is being explored as means of controlling several exotic plant species that negatively impact the value of wildland habitats (DeLoach, 1997).

To date, there exist only a limited number of instances of its implementation in wildland management, with the exception of its use in controlling rangeland and aquatic weeds. In certain cases, reserves have benefited from the introduction of biocontrol agents to combat weeds in nearby agricultural settings. For instance, the introduction of biocontrol agents was found to reduce the prevalence of the noxious weed *Senecio jacobaea* (tansy ragwort) in Redwoods National Park, although the agents had been released in the region for the purpose of controlling *S. jacobaea* on rangelands, rather than within the park itself. The release of biocontrol agents in U.S. parks and reserves has been a contentious issue. Advocates of weed biological control argue that there is little evidence to suggest that control agents cause unexpected damage (Simberloff and Stiling, 1996).

Simberloff and Stiling (1996) have noted that only a small percentage of all biocontrol releases have been subjected to rigorous study. Rather than advocating for

the abandonment of biological control, they insist that control agents should not be deemed safe until research supports this conclusion. Another biological method involves the use of microbial insecticides, such as those derived from the bacterium *Bacillus thuringensis* (Bt). This microorganism occurs naturally in the soil and on the surface of leaves. During sporulation, the bacterium produces a toxic protein crystal that can be utilized as an insecticide. The toxin produced by one strain, Bt, is specific to Lepidoptera; the gene for the Bt toxin has been genetically engineered into several major crops, including maize and cotton.

According to Aarssen and Epp (1990), chemical control is a highly effective method for eradicating or managing invasive alien species. Biocides, a term synonymous with pesticides, encompass all chemicals utilized to control noxious and invasive organisms, including insecticides for insect control, herbicides for plant control, and rodenticides for vertebrates such as rats and mice, or poisons for larger animals like foxes and rabbits. The origins of chemical control date back to the advent of agriculture, when it became necessary to preserve stored grains between seasons. The Sumerians used sulfur around 2500 BC, while in China, chalk, wood ash, and botanical products were employed for the treatment of stored grain from approximately 1200 BC. Arsenic was utilized as an insecticide in the second century BC. After centuries of use in traditional folklore, the insecticidal properties of certain botanical products, such as pyrethrum, nicotine, and derris, were recognized from the sixteenth century onwards.

The early twentieth century witnessed the standardization of petroleum oils and botanical products, as well as the initiation of investigations into the correlation between chemical structure and biological activity. The significant advancement in chemical insecticides occurred around 1940, with the discovery of the insecticidal properties of DDT (1,1,1-dichloro diphenyl trichloroethane) and BHC (benzene hexachloride). Numerous chemicals were subsequently tested for their insecticidal properties, leading to their widespread use worldwide. However, in the early 1950s, concerns arose regarding the toxicity of these chemicals to vertebrates, including humans, as well as the presence of pesticide residues in food. Additionally, the emergence of resistance among target species to commonly used insecticides posed a new challenge. Consequently, it is imperative that chemicals employed for control purposes exhibit specificity towards the target species, while also being safe, non-

persistent, and non-accumulative in food chains. Furthermore, strict adherence to the manufacturer's guidelines is essential when utilizing chemicals. The creation of extra-strength solutions and the mixing of chemicals can pose risks to both users and the environment. Proper disposal of unused chemicals and their containers is crucial, as they should not be dumped or poured into drains or waterways. When implementing control methods for animals, it is imperative to prioritize humane practices while ensuring effective control.

Numerous studies have indicated that various methods of removing invasive plants, such as hand-weeding or herbicide application can have an impact on native communities. These impacts can arise from soil disturbance or unintended effects of herbicides (Aarssen and Epp, 1990; Campbell *et al.,* 1991; McLellan, 1995). However, previous experiments that have evaluated the responses of native communities have typically focused on a single removal method (D'Antonio *et al.,* 1998; Alvarez and Cushman, 2002; Hulme and Bremner, 2006).

For instance, removal treatments like hand-weeding often disturb the soil, which can disrupt the root systems and mycorrhizal networks of non-target plants (McLellan, 1995). However, hand-weeding can also facilitate the establishment of both native (Biggerstaff and Beck, 2007) and non-native (Mack and Lonsdale, 2002; Ogden and Rejmanek, 2005; Mau-Crimmins, 2007) plants.

Chemical treatments can have significant effects on a specific group of plants if the herbicide is designed to target only monocotyledonous or dicotyledonous plants (Pavlik *et al.,* 1993; Cione *et al.,* 2002). Additionally, these treatments may leave behind toxic residues that inhibit the recruitment of native plants. Pre-emergent herbicides may also hinder the germination of non-target species. Other control methods, such as grazing, fire, mowing, or shading, may also have unforeseen or unintended consequences.

To develop effective techniques for managing invasive plants and promoting the recovery of native communities, it is necessary to conduct studies that test multiple removal methods for specific plant invaders. Black wattle (*Acacia mearnsii* De Wild) holds significant economic value for South Africa, but also poses a significant threat to their natural ecosystems when it spreads beyond plantations into the surrounding grasslands and watercourses. This invasive species often forms dense,

impenetrable thickets that displace native vegetation and impede water flow. In South Africa, the primary method of controlling black wattle is through chemical means (Campbell, 1987). Traditional control methods involve applying the registered selective, systemic herbicide Garlon 4 to the foliage of young plants using knapsack sprayers before they reach a height of 1 meter (Vermeulen *et al.,* 1996). However, due to the extensive nature of the problem, the project lacked the capacity to address it in a timely manner. Therefore, an urgent need arose for a more efficient solution to prevent previously cleared areas from being overrun by black wattle once again. Consequently, the objective of this study was to assess the effectiveness of backpack mist blowers and various concentrations of Garlon 4 in controlling extensive, dense stands of black wattle seedlings that typically emerge following fire events and significant rainfall.

A study, control of black wattle (*Acacia mearnsii* De Wild.) seedlings with Garlon herbicide applied by backpack mist blower from South Africa by Viljoen and Stoltsz (2008) reported that the long-term success of any large-scale black wattle clearing operation depends on the ability to prevent re-invasion of seedlings from the soil seedbank following initial clearing. The larger the scale of the clearing operation, the bigger the risk of seedlings escaping timely follow-up operations and the area reverting back to jungle infestation, making the original problem many times worse. The high incidence of wild fires in grassland areas necessitates a rapid and cost-effective method to combat the extensive seedling flushes that follow. The results of this experiment support the registration of 0.125% Garlon 4 when applied by backpack mist blower for this purpose. Knapsack sprayers would still be useful for spot spraying any survivors or late germinators. Considering that effective black wattle control relies on frequent follow-up control operations until the massive soil seed bank is exhausted, the registration of such a low rate of Garlon 4 should improve the cost benefit of future control operations, with less negative impact on the environment.

In the dunes of north-western Europe, particularly in the Denmark region, *Rosa rugosa* Thunb.(commonly known as Japanese rose and belonging to the Rosaceae family) poses a significant problem. This invasive species poses a threat to coastal ecosystems and suppresses the natural vegetation that holds high conservation value (Reddersen, 2006; Jorgensen and Kollmann, 2009). The control of this species

through mechanical means is challenging as it exhibits vigorous regeneration even after cutting, grazing, mowing, harrowing and burning (Weidema, 2006; Weidema *et al.,* 2007). While herbicides have proven to be effective, their use is often not recommended in plant communities where this species becomes a pest. Additionally, biological control methods may be problematic due to the introduction of foreign organisms (Bruun, 2006).

In the case of *Miconia calvescens* DC.(Melastomataceae), which is considered one of the most destructive plant invaders in the native forests of pacific islands, the situation differs from that of "agricultural weeds" such as grasses, herbs, vines, shrubs and aquatic plants, which are targeted for eradication in regions like California, USA (Rejmanek and Pitcairn, 2000), Australia (Woldendorp and Bomford, 2004; Parkes and Panetta, 2009), or New Zealand (Harris and Timmins, 2009). *Miconia,* being a small tree, possesses the ability to reproduce prolifically and maintain a persistent seed bank. Consequently, it can invade species-rich, intact rain forests and cloud forests, leading to the destruction of native biodiversity. Control methods consist of manually uprooting seedlings and saplings, chemically treating the reproductive (or mature) trees on cut-stumps or bark, and carefully targeting spraying from helicopter (Later it was restricted only in Hawaii). For this species, volunteers for short-term control operations or long-term funded teams, or both, have been involved and public awareness campaigns have been conducted in all island groups (Conant *et al.,* 1997; Medeiros *et al.*, 1997; Meyer and Malet, 1997; Meyer, 2010). Eradication success depends on: 1) the number and size of infestations, 2) the accessibility of infestations, 3) detectability of the species, 4) the biological characteristics of the species (or its invasiveness), and 5) effectiveness of the control (Panetta and Timmins, 2004). Furthermore, the most cost effective strategy against invasive plants is early intervention and eradication during a “lag phase” when populations remain small and localized (Hobbs and Humphries, 1995; Loope and Stone, 1996).

Weed risk assessment, quarantine regulations, and other biosecurity and phytosanitary measures form a first barrier to plant invasion. Eradication of invasive species depends on consequent monitoring and follow-up treatments (Witkowski and Garner, 2008).

2.11. Studies on Restoring Invaded Ecosystems

Invasive species are difficult to eradicate once established (Benz *et al.,* 1999; Manchester and Bullock, 2000; Wilson, 2002; Price and Weltzin, 2003). Methods of preventing or constraining their spread have been difficult to develop (Keane and Crawley, 2002; Perry and Galatowitsch, 2003; Perry *et al.,* 2004) but would be ecologically valuable. Ecological restoration is 'the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed' (Society for Ecological Restoration, 2002) and often involves the planting of native species associated with later successional stages.

Restoration requires both the removal of invasive plants and re-establishment of native plant communities (Diaz *et al.,* 2003; Hulme, 2006). Much research has focused on developing effective chemical, mechanical and biological removal methods (Paynter and Flanagan, 2004; Judge *et al.,* 2005a; Simmons *et al.,* 2007).

Ecological restoration is widely acknowledged for its positive impacts on soil stabilization, wildlife habitat and biodiversity. A study conducted by Bakker (2004) suggests that another advantage of ecological restoration is its ability to limit biological invasion. Other studies have also demonstrated that ecological restoration can decrease the prevalence of established invasive species (Benz *et al.*, 1999; Price and Weltzin, 2003). A study by Bakker (2004) specifically reveals that ecological restoration not only hinders the spread of invasive species but also facilitates the establishment of native species. In this way, ecological restoration acts as a filter, allowing native species to thrive while curbing the expansion of invasive species. Consequently, restoration efforts can effectively address the challenging issue of invasion management (Keane and Crawley, 2002).

A comprehensive analysis of published restoration studies indicates that invasion control is not as frequently studied as other forms of restoration, such as forest restoration. However, a closer examination of the subset of invasion-restoration studies reveals a clear connection between invasion and restoration ecology in the scientific literature. Restoration of a site colonized by an invasive species can present a unique challenge because some species can continue to affect a system after their removal, preventing attainment of the desired restoration outcome (Cronk and Fuller, 1995).

A major goal of restoration practitioners is to return a habitat to a more desirable condition involving a particular species composition, community structure, and/or set of ecosystem functions (Noss, 1990). The importance of restoration for the management of alien plant invasion is reflected in the study of Mirijam (2012). The author opines that 65% of the studies with the aim of controlling the invader and promoting native species adopted measures other than the removal of the invader. At the same time, invasives play an important role in restoration studies that have other objectives (e.g. forest restoration), interfering with restoration actions by hindering the establishment of native species. Interesting is the finding that in some cases alien species are even used in the process of restoration (Lavoie *et al.,* 2005; Jurado *et al.,* 2006).

In the case study conducted by Suresh Babu et *al.,* (2009), the authors present the 'Ecological Restoration of *Lantana* Invaded Landscapes in Corbett Tiger Reserve, India'. This paper demonstrates the successful eradication and restoration of two sites invaded by lantana in the reserve. A method for eradicating lantana was developed based on an understanding of its ecology, and subsequently, the landscapes were restored to productive grasslands and mixed woodlands using native species. The restoration of these areas to grassland communities has effectively prevented secondary invasions by lantana and other weeds, thereby enhancing the habitat quality for herbivores. The presence of these herbivores is crucial for the survival of top carnivores such as the tiger (*Panthera tigris corbetti*). The restoration technique employed by the authors consisted of four important steps. Firstly, reference ecosystems were identified through field ecological surveys in relatively undisturbed areas of the park that shared similar ecological characteristics. Secondly, target conditions were defined, with the requirements of the park management being taken into consideration as stakeholders. The third step involved the establishment of field nurseries and the sequential introduction of selected species into the lantana-free landscapes. The fourth and most significant step involved monitoring and follow-up activities, including manual weeding under perching trees and along drainage channels. As a result, the experimental plots are now fully covered with a diverse range of grass species and legumes, interspersed with scattered trees. Any restoration programme implemented in a similar setting would also need to consider various factors, such as the dispersal mechanism of the invader, soil and climate conditions,

the current state of the ecosystem, the condition of neighboring ecosystems, and most importantly, the requirements of the stakeholders.

According to Amit *et al.,* (2009), one of the primary issues currently faced by numerous protected areas in India is the degradation and fragmentation of habitats, which leads to the colonization of these disturbed habitats by invasive species. The extensive areas of forest and community lands covered by invasive weeds present two-fold challenges: the control and eradication of these weeds, and the restoration of weed-free landscapes to prevent further invasions and improve habitat conditions. The authors assert that the eradication of lantana and subsequent restoration of grasslands in Corbet Tiger Reserve offer several benefits. These include enhanced biological productivity, particularly in terms of palatable species of grasses and legumes, as well as improved retention of soil moisture, prevention of soil erosion, enrichment of native biodiversity, increased frequency of wildlife sightings, and enhanced recreational values.

According to D Antonio and Meyerson's (2002) work on 'Exotic Plant Species as Problems and Solutions in Ecological Restoration: A Synthesis', landscapes that have been degraded by invasive species are often extensive. Even as weeds are eradicated from one area, the propagule pressure can persist from other areas. Therefore, a well-coordinated removal programme followed by restoration at suitably large scales is imperative to tilt the scales in favor of native species.

Invasive exotic species may play a role in the restoration process in several ways. Firstly, their presence or dominance at the site may be part of the condition leading to the assessment that restoration is needed. In the best-case scenario, restoration may be as simple as removing founding individuals of an exotic species. Secondly, exotic species may be the first species to recolonize after disturbances associated with removal. Thirdly, exotic species may be the first to colonize after a planned disturbance (power line right-of-way, pipeline corridor, etc.) even if they were not present in the pre-disturbance community. This may interfere with restoration efforts or alter successional processes that would otherwise lead to a native assemblage. Fourthly, they may leave behind a legacy after removal that makes long-term restoration of the site difficult or challenges management goals. This legacy may be in the form of a buried seed bank or chemical or physical alteration to the habitat. Finally, exotic species may be used by managers to restore particular functions if native species are not available. The success of restoration strategies is contingent

upon a comprehensive understanding of the fundamental processes associated with the regeneration of invasive plants (Clewell and Rieger, 1997; Byers *et al.*, 2002; Emery and Gross, 2005; Tassin *et al.,* 2007).

The restoration of sites that have been degraded by exotic species and soil erosion presents a particular challenge. Following an invasion, the topography may no longer resemble its pre-invasion state, and the removal of the exotic species may lead to further erosion, particularly if there is a delay between removal and the establishment of native plants. Highly degraded sites may no longer have the capacity to support the desired species assemblages. In such circumstances, it may be necessary to adopt a gradual approach and stabilize the soil using synthetic or biodegradable materials, or establish native vegetation prior to removing the exotic species. This planting could either resemble the desired species assemblage upon completion of the restoration or serve as an interim fallow planting to enhance site quality (D' Antonio and Meyerson, 2002).

According to several studies conducted by Planty-Tabacchi *et al.* (1996), Wiser *et al.* (1998), Levine and D'Antonio (1999), Stohlgren *et al.,* (1999), and Symstad (2000), research has shown a positive correlation between the diversity of native species and the diversity of exotic species. This finding contradicts previous theories that suggested communities with low species diversity are more prone to invasion due to their lower "biotic resistance" (Levine and D'Antonio, 1999).

Although small-scale experimental studies indicate that high native diversity can reduce vulnerability to invasion, larger-scale investigations suggest that regional factors influencing diversity overshadow these local effects (Levine and D'Antonio, 1999; Levine, 2000). Consequently, natural areas that are valued for their high species diversity may be particularly susceptible to invasion and therefore require increased vigilance against harmful invaders.

Chapter 3

METHODOLOGY

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Based on the alarming nature of the distribution of *Senna spectabilis* populations reported in different habitats of Kerala state of India, appropriate experiments were designed and carried out to estimate the current distribution and abundance of *Senna spectabilis* in the study area, to study the pollination, seed dispersal, phenology and variability of the species in the area and also to develop appropriate management protocol for its effective control and also for the restoration of the habitats using native species.

3.1. Estimation of the current distribution and abundance of *Senna spectabilis* in Kerala

Populations of *Senna spectabilis* were identified in Kerala through reconnaissance survey, forest working plans and floristic records. The survey was conducted from 2016 to 2019 and it covered all major and minor roads, natural forests, plantations, wetlands, vacant lands and other valuable sensitive areas.

A grid-based sampling design was adopted for data collection on this invasive speciespresence. The size of the individual grid was 1km x 1km. *S. spectabilis* populations were marked with Garmin Etrex 20x GPS device. The device was used to archive location coordinates, and only one record was collected within each grid. The accumulated points were georeferenced with Google Earth to obtain accuracy in coordinates for mapping. Using GIS technologies (ArcMap ver.10.7.1), mapping was carried out.

A comprehensive ecological survey was carried out using 20m x 20m sample plots laid out randomly in the selected areas and various growth stages like seedlings, saplings and mature trees occurring in the quadrats were recorded and subsequently grouped them into different girth classes and the population structure determined, following Jayakumar and Nair (2013). Worked out the density, abundance and frequency of the studied species and associate species using the formula (Misra,1968):

$$\text{Density (No. of individuals/ ha)} = \frac{\text{No. of individuals encountered}}{\text{Total area sampled in m}^2} \times 10{,}000$$

Studies and reports revealed that Wayanad Wildlife Sanctuary is an aggressive growth habitat of *S. spectabilis*. From this point of view, distribution pattern and spread of *S.*

spectabilis, characteristics and distribution analysis of the entire vegetation in the sanctuary was carried out. A pilot observation study was conducted to determine the population and spread of *S. spectabilis* and also GPS points were taken for mapping the distribution during the 2017-2018 periods.

Wayanad Wildlife Sanctuary is contiguous to the PAs of Nagarhole and Bandipur of Karnataka on the north-east and Mudumalai of Tamil Nadu on the south-east. It is located from $11^0$34'39.9" N to $11^0$58'12.6" N and 75^0 59'22.7" E to $76^0$26'11.04" E. It has an extent of 344.44 km^2, and is rich in biodiversity and is an integral part of the Nilgiri Biosphere Reserve. The sanctuary has four ranges: Sulthan Bathery, Muthanga, Kurichyad and Tholpetty. It is a stretch of moist deciduous forests along the state border towards the east. It was established in 1973. Dry deciduous forests are lying along this track of the sanctuary. The total extent of the area is divided into two discontinuous portions with revenue lands in between.

The Northwest portion of the sanctuary has only one range, viz., Tholpetty with an area of 77.67 km^2. This range is contiguous to Rajiv Gandhi National Park in Nagarhole in the northeast, Kakkankotte Reserve Forest in the north and Brahmagiri Hills of North Wayanad division in Kerala in the east. The southern portion of the sanctuary comprises an area of about 266.77 km^2. It is contiguous to the Mudumalai Wildlife Sanctuary of Tamil Nadu in the east and Bandipur Tiger Reserve (BTR) of Karnataka in the north and northeast. These areas are moist deciduous forests that gradually descend to the dry deciduous forests and join with the Deccan plateau. There are, however, a few patches of West-Coast semi-evergreen forests within this contiguous forest belt. About 110 km^2 of the sanctuary is under different plantations.

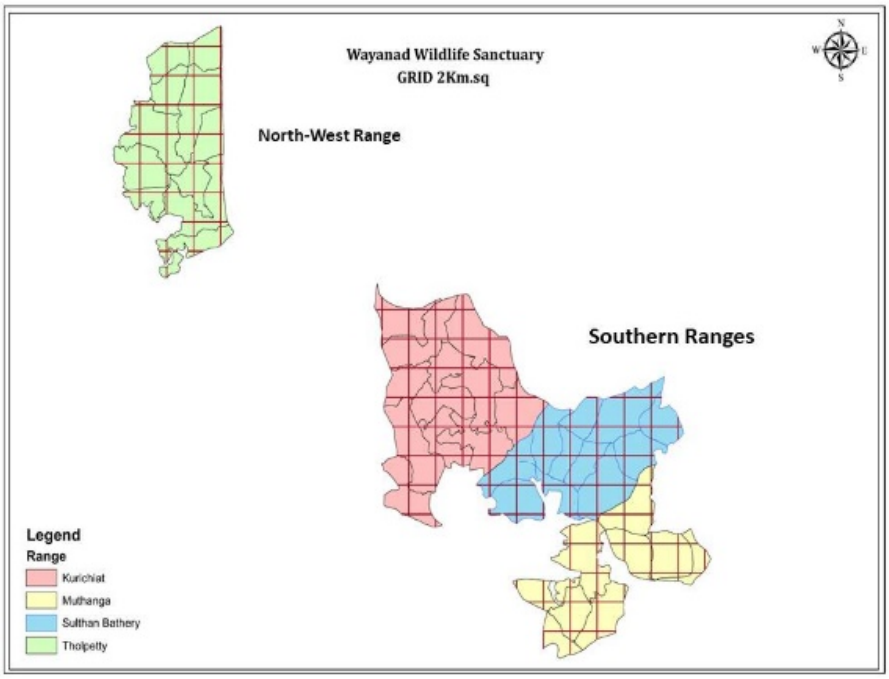


Fig. 3.1. Study area: Map of Wayanad wild life sanctuary

The vegetation survey was conducted in Wayanad Wildlife Sanctuary during the pre-monsoon and post-monsoon from 2018 to 2019. Detailed field surveys were adopted for recording the floristic composition of the communities in these sites. The survey covered all the major and minor roads, natural forests, plantations, wetlands, vacant lands and other valuable, sensitive areas. For the study purpose, based on some distinguishing factors, the study area for vegetation analysis was divided into two: 1) The southern ranges (Muthanga, Sulthan Bathery and Kurichyad forest ranges) and 2) The north-west range (Tholpetty forest range).

Southern Ranges: Study area, one is referred to as the Southern ranges, including Muthanga, Sulthan Bathery and Kurichyad forest ranges. The forest types here are classified into the following categories (Champion and Seth, 1968): Southern moist, mixed deciduous, southern dry mixed, deciduous interspersed with teak plantations and softwood plantations. Also, some patches of west semi-evergreen forest are also there. Moreover, tree invasive species like *Senna spectabilis*, *Mesopsis eminii* and *Eucalyptus* are spreading in this area.

North-west Range: In this study site, two areas of the Tholpetty forest range are included: 3840.5 Ha area having teak plantations interspersed with the tree invasive *S. spectabilis*. Moist deciduous forest patches dominate these areas (Champion and Seth, 1968).

The land cover map of the study area was divided into grids comprising of 2 km^2 and marked in the topo sheet of the study area. Sample plots of size 20 m x 20 m were laid out in each grid. This exercise was repeated five times randomly in the same grid. Thus, a total of 350 quadrats: 265 in southern ranges and 85 in northwest range, were established (for enumerating seedlings, four consecutive 5 m x 5 m subplots were taken in each 20 m x 20 m plots) (Ludwig and Reynolds, 1988; Magurren, 1988)

To assess the extent of this species in the forest area, it was categorised based on different strata: tree layer, shrub layer and herb layer. Individuals above 31.5 cm CBH (the circumference at breast height, i.e., 1.37 m above ground level) were considered trees; those between 10.5 cm to 31.5 cm CBH were recorded as shrubs and individuals less than 10.5 cm CBH were considered as herbaceous plants (Knight, 1963). The seedlings were considered herbs and saplings as shrubs. GPS recorded the site's elevation/altitude, longitude and latitude at 5-6 accuracy.

Data collection was undertaken through a floristic survey. The plant samples were collected and processed following the routine method of plant collection and herbarium technique (Lawrence, 1951; Judd *et al.*, 2002). The species were identified with the help of Flora of the Presidency of Madras (Gamble and Fischer, 1915-1935) and also were identified with the use of some online resources like invasive species databases. CABI's Invasive Species Compendium (http://www.cabi.org/isc/), Flowering Plants of Kerala ver. 2.0-DVD, India Biodiversity Portal (http//indiabiodiversity.org) was accessed whenever required.

3.1.1. Data Analysis

The data were quantitatively analysed. The Importance Value Index (IVI) is a better expression of the relative ecological importance of the species than the single absolute values such as frequency, density, basal cover, etc. (Cain *et al.*, 1956). The IVI is calculated by summing the relative values, viz., relative frequency, relative density and relative dominance (Curtis, 1959). These relative values are calculated following Phillips (1959) and Mishra (1968). The abundance (A)/ frequency (F) ratio (Whitford, 1949) was used to interpret the distribution of the species. The distribution was classified as regular (<0.025), random (0.025-0.05) and contagious (>0.05) following the method of Curtis and Cottam (1956). The following formulae were used to calculate the quantitative parameters:

Frequency % (F) = $\frac{\text{No. of quadrats of occurrence of a species}}{\text{Total number of quadrats studied}} \times 100$

Density (D) = $\frac{\text{Total No. of individuals of a species in all the quadrats}}{\text{Total number of quadrats studied}}$

Abundance (A) = $\frac{\text{Total No. of individuals of a species in all the quadrats}}{\text{Total number of quadrats in which the species occurred}}$

Relative Frequency (RFR) = $\frac{\text{Total number of occurrence of the species}}{\text{Total number of occurrence of all the species}} \times 100$

Relative Density (RD) = $\frac{\text{Total number of individuals of the species}}{\text{Total number of individuals of all the species}} \times 100$

Relative Dominance (RDom) = $\frac{\text{Total basal area/Basal cover of the species}}{\text{Total basal area/ Basal cover of all the species}} \times 100$

IVI= Rel. Frequency + Rel. Density + Rel. Dominance

Distribution = $\frac{\text{Abundance}}{\text{Frequency}}$

The species types are divided into five frequency classes depending on Raunkier's (1934) frequency classification is as follows.

Class A= 1 to 20%

Class B= 21 to 40%

Class C= 41 to 60%

Class D= 61 to 80%

Class E= 81 to 100%

Each species is classified into the classes as mentioned above. A histogram is drawn with the percentage of the total number of species on the Y-axis and frequency classes A to E on the X-axis.

It is compared with the law of frequency as follows. A>B>C>or<D<E

3.1.2. *Senna spectabilis* Distribution Modelling

In addition to the above objective, we studied the impact of projected climate change on the spread and distribution of *Senna spectabilis* in Wayanad. For that, the potential habitat suitability of *S. spectabilis* had been modelled by the species distribution modelling under current and future climatic conditions, using Maxent species distribution models methods.

For occurrence data, the field visit was done in the Wayanad district of Kerala and carried out in March 2019. A grid-based sampling design was adopted to systematise data collection on the presence of selected invasive species. The size of the individual grid was 1 km x 1 km. Systematic sampling was done in the area. Garmin Etrex 20x GPS device was used to archive location coordinates, and only one record was collected within each grid. The data records were mainly from the roadsides, open area presence points and the presence points inside the wildlife sanctuary. A total of 374 presence records were compiled from the field survey. Data were refined for the occurrence record in Microsoft Excel to remove duplicates. The spatial auto-correlation between the occurrences of *S. spectabilis* was rectified using the package "spThin" (Aiello-Lammens *et al*., 2015) in the R platform. The incidence records were reduced to 94 from 394 after eliminating spatial auto-correlation and multiple records.

3.1.2.1. Preparation of environmental variables

The bioclimatic predictor variables were obtained from the WorldClim database WorldClim version 2.1 (https://www.worldclim.org) at 30 arc-second scales (accessed the data on 23/11/2020). These bioclimatic variables are the derivatives of monthly rainfall and temperature values for 1970–2000 (Fick and Hijmans, 2017). These variables represent annual trends, seasonality and extreme or limiting environmental factors. The 19 variables are as follows:

- BIO1 = Annual Mean Temperature (degree Celsius)
- BIO2 = Mean Diurnal Range (Mean of monthly (max temp - min temp) (degree Celsius)
- BIO3 = Isothermality (BIO2/BIO7) (×100) (dimensionless)
- BIO4 = Temperature Seasonality (standard deviation ×100) (degree Celsius)
- BIO5 = Max Temperature of Warmest Month (degree Celsius)
- BIO6 = Min Temperature of Coldest Month (degree Celsius)

- BIO7 = Temperature Annual Range (BIO5-BIO6) (degree Celsius)
- BIO8 = Mean Temperature of Wettest Quarter (degree Celsius)
- BIO9 = Mean Temperature of Driest Quarter (degree Celsius)
- BIO10 = Mean Temperature of Warmest Quarter (degree Celsius)
- BIO11 = Mean Temperature of Coldest Quarter (degree Celsius)
- BIO12 = Annual Precipitation (mm) 30
- BIO13 = Precipitation of Wettest Month (mm)
- BIO14 = Precipitation of Driest Month (mm)
- BIO15 = Precipitation Seasonality (Coefficient of Variation) (Fraction)
- BIO16 = Precipitation of Wettest Quarter (mm)
- BIO17 = Precipitation of Driest Quarter (mm)
- BIO18 = Precipitation of Warmest Quarter (mm)
- BIO19 = Precipitation of Coldest Quarter (mm)

The temperature unit is '^{0}C' and precipitation is 'mm'. 30 arc-seconds (approximately 1 km^2 at the equator) resolution data were used for current and future conditions. In addition, these were in the latitude/longitude coordinate reference system under the datum WGS84.

According to Fick and Hijmans (2017) and Hutchinson and Xu (2013), the WorldClim database was created by interpolating average monthly data from weather stations worldwide, excluding Antarctica. Besides, these climate data or the weather station data were compiled from many sources and databases that comprise long-term average values for creating 'climate surfaces'. For future climate projections, the future bioclimatic data at a spatial resolution of 30 arc-seconds (~1 km) were downloaded (on 22/11/2020) from CCAFS on the Climate Change and Agricultural Food Security (CCAFS) climate data archive (http://ccafs-climate.org/).

These datasets are a part of the Decision and Policy Analysis (DAPA) program's climate change downscaled data from the International Centre for Tropical Agriculture (CIAT), according to which these future bioclimatic raster data 31 were downscaled from IPCC general circulation models (GCM) from the IPCC's fifth report (IPCC 2013, future climate projections) and reprocessed using thin-plate spline interpolation algorithm anomalies and the current distribution of climates from the WorldClim version 1.4 database developed by Hijman *et al.,* (2005).

The unit of temperature is '^{0}Cx10'and precipitation is 'mm'. The temperature variables were further converted to ^{0}C using the raster calculator in ArcGIS version 10.7.1 ESRI. All the four representative concentration pathways viz., RCP 2.6, RCP 4.5, RCP 6.0 and RCP 8.5 were chosen (Table.1). They followed the Hadley Global Environment Model 2-Earth System Model (HadGEM2-ES) with a spatial resolution of 30 arc-seconds (~1 km) as stated in the fifth assessment report (AR5) of the Intergovernmental Panel on Climate Change (IPCC, 2014).

Table 3.1 Different RCP scenarios used for the future projection of *Senna spectabilis* in the Wayanad district of Kerala

Sl No.	Representation concentration pathways	Radiative forcing	Temperature anomaly (^{0}C)	CO_2 concentration (ppm)
1	RCP 2.6	3.1 W/m2 then decline by 2100	0.3^0C – 1.7^OC	490
2	RCP 4.5	4.5 W/m2 after 2100	1.1^0C – 2.6^OC	650
3	RCP 6.0	6 W/m2 after 2100	1.4^0C – 3.1^OC	850
4	RCP 8.5	8.5 W/m2 by 2100	2.6^0C –4.8^OC	1370

In addition to climatic variables, land use, land cover, topographic variables (slope, aspect and elevation), soil, population density, normalised vegetation index, distance from water bodies and distance from the road were also considered for modelling. The non-climatic variables were selected after the literature survey to understand the importance of these variables to the invasive species.

The digital elevation model was directly procured from Global 30 arc second elevation (GTOPO30) from the U.S. Geological Survey (https://lta.cr.usgs.gov/GTOPO30) on 16/05/2021. The slope and aspect maps were derived from DEM using ArcMap ver.10.7.1 ESRI. The Landcover dataset was accessed from the Global 1-km Consensus Land Cover Earthenv database archive (http://www.earthenv.org//landcover) on 16/05/2021. These datasets provide one km resolution consensus data on the prevalence of 12 different land-cover classes by combining multiple global remote sensing-derived land-cover products. Normalised difference vegetation index layers were retrieved from the Land Processes Distributed Active Archive Center (LP DAAC, https://lpdaac.usgs.gov) maintained by NASA EOSDIS Land Processes Distributed Active Archive Center (LP DAAC) at the USGS

Earth Resources Observation and Science (EROS) Center on 16/05/2021. The datasets were obtained from the temporal monthly average of the Terra Moderate Resolution Imaging Spectroradiometer (MODIS) Vegetation Indices (MOD13A3) Version 6 data at one km spatial resolution. Vegetation indices are used for global monitoring of vegetation conditions and are continuous raster datasets. The soil type information was obtained from the Department of Soil Survey and Soil Conservation (http://www.keralasoils.gov.in/). The vector map was georeferenced and rasterised to one km spatial resolution.

The layers' distance from water bodies and distance from the road were derived using ArcMap ver.10.7.1 ESRI, the datasets for water bodies were obtained from Near-global freshwater-specific environmental variables for biodiversity analyses in one km resolution Earthenv database archive (Domisch *et al*., 2015, https://www.earthenv.org/) and datasets for road network were obtained from the NASA Socioeconomic Data and Applications Center (SEDAC) Global Roads Open Access Data Set, (gROADSv1, https://sedac.ciesin.columbia.edu/data/set/groads-global-roads-open-access-v1). Anthropogenic pressure is an important driver of bioinvasion (Liu *et al.*, 2005; Bhattarai *et al.*, 2014; Shrestha *et al*., 2015), and therefore, the inclusion of the population density layer is important. Population density layer was obtained from Gridded Population of the World (GPW, v4.11) from 33 the NASA Socioeconomic Data and Applications Center (SEDAC, https://sedac.ciesin.columbia.edu/data/set/gpw-v4-population-density-rev11) at 1 km spatial resolution. These datasets comprise estimates of human population density (number of persons per km^2). The geographic dimensions of all environmental layers for the study area and pixel size were made uniform using the resample tool in ArcGIS ver.10.7.1 ESRI and the environmental layer tiles were available at ~ one km^2 spatial resolution.

3.1.2.2. Model design

Selection of optimal environmental variables: The model's accuracy is influenced even when there remains a mild correlation between the explanatory variables (Veloz *et al.,* 2009). Therefore, to reduce the masking effect of a large number of collinear variables and to obtain an optimum predictive model result, the variables were tested for multicollinearity using Pearson correlation coefficient (r). One among the two

strongly cross-correlated variables (Pearson correlation coefficient r>0.70) was chosen for inclusion in the model, considering its biological significance to the species and ease of interpretation. For example, the precipitation of the driest quarter was kept if precipitation seasonality (BIO14) and precipitation of the driest quarter (BIO17) were correlated since it has higher significance to species than precipitation seasonality. The variability of the bioclimatic variables with the different RCP scenarios and currents was also critically analysed.

3.1.2.3. Model Development

Model selection: Maximum entropy modelling (MaxEnt), the most accepted species distribution model, was used for modelling presence-only data (Bosso *et al.*, 2018; Soucy *et al.,* 2018; Zhang *et al.,* 2018). MaxEnt version 3.4.4 was downloaded from 34 https://biodiversityinformatics.amnh.org/open_source/maxent/ on 24/11/2020. The model was used to predict the potential habitat suitability for invasion of *S. spectabilis* and generate a distribution map. It was also used to model the future suitability and distribution map under HadGEM2-ES climate change scenarios for 2050 and 2070. MaxEnt uses a machine-learning algorithm to estimate the relationship between species presence data and those sites' spatial/environmental characteristics (Franklin, 2009).

MaxEnt computes for each grid cell the predicted suitability of conditions for the species. The species distribution output is obtained when the georeferenced species occurrence records and environmental variables are fed into it. Species data were made into '.csv' (comma separated value) format and the bioclimatic layers as '.asc' (American Standard Code for Information Interchange) format when inputting into the MaxEnt model. All the selected fourteen variables except soil type were continuous. The default settings options in the software were programmed for the model training (Phillips *et al.*, 2004; 2006).

3.1.2.4. Model Training and Optimisation

Model optimisation was determined using the "ENMeval" package (Muscarella *et al.,* 2014) in R platform. The least delta AIC (Akaike Index Criterion) was selected for choosing the best fit model for the current species distribution modelling. Forty eight models with different regularisation multiplier values and different levels of complexity were developed. Regularisation multiplier features were employed to

prevent model overfitting (Philips and Dudik, 2008). The best replication run type was then determined from the literature review. Finally, the subsampling replication run type was determined, where random replicate sample sets were chosen for evaluation by removing random test percentages without replacement. All variables were analysed to determine each variable's contribution to the distribution modelling for the species. This was done for the current distribution (no projections for the future).

3.1.2.5. Variable Contribution to the Model

The contribution of each selected variable (static and dynamic variables) to modelling the distribution of *S. spectabilis* was identified by analysis. This was done to model the current distribution (no future projection). The model was run for *S. spectabilis* with 5000 iterations and ten replicates with a subsampling procedure, among which 75 percent was used for testing, and the remaining 25 percent of iterations were used for training. The output was made in logistic format to get the probability of occurrence in the range of 0-1. The increased regularised gain was added to the contribution of the corresponding variable in determining the percentage contribution or subtracted from it if the change in the absolute value of lambda was negative in each run of the training algorithm. The values of each environmental variable on training presence and background data were randomly permuted to determine permutation importance.

3.1.2.6. Model Evaluation

3.1.2.6.1. Accuracy assessment

Threshold independent ROC AUC: AUC is a threshold-independent metric that quantifies the model's ability to distinguish between random and background points (Raman *et al.,* 2020). AUC values above 0.90 indicate excellent model accuracy, suggesting that the model can distinguish between presence and absence records; values between 0.7 and 0.9 indicate good accuracy; values between 0.5 and 0.7 indicate low accuracy, and values below 0.5 are not better than a random chance. It equally weighs omission and commission errors (reliable when using the presence/absence model). AUC values are correlated by the prevalence of the occurrence point size of the study area and distribution of species and ignore the predicted probability and the model's goodness of fit (Philips *et al.,* 2006).

True skill statistics: A highly suggested measurement/index, a threshold-dependent measure that accounts for omission and commission errors, TSS is defined as "sensitivity + specificity – 1". The index range ranges from -1 (a random fit) to +1 (a perfect fit). Unlike AUC, TSS values are not affected by the study region's size and the occurrence records' prevalence (Allouche *et al.,* 2006).

3.1.2.7. Sensitivity Analysis

The jackknife technique was used to test the sensitivity of the model. The relative importance of the predictor variable was determined by jackknifing, and it calculates the training gain of each variable if the model is being run in isolation and compares it to the training gain with all the variables.

3.1.2.8. Thresholding of Model Outputs

The output was formatted in logistic format (binary maps) to obtain the probability of occurrence in the range of 0-1 (Phillips *et al.*, 2004) based on the selected logistic threshold value 'maximum test sensitivity plus specificity (max sss), regarded a recommended threshold selection method for presence/absence data (Liu *et al.,* 2005). Across models, sensitivity and specificity were not affected by prevalence because they were independent of each other (Allouche *et al.,* 2006).

Furthermore, sensitivity is defined as the proportion of correctly predicted presences, and specificity is the proportion of correctly predicted absences among all the absences. Therefore, it has been proved valid to use with presence-only data, and instead of true absences, random records are used (Liu *et al.,* 2013). The 'max SSS' selects a point in the receiver operating characteristic (ROC) curve that plots sensitivity and 1-specificity to maximise the TSS where the tangent slope equals 1 (Smeraldo *et al.,* 2017; Bosso *et al.,* 2018). Using SDM toolbox 2.4 in ArcMap ver.10.7.1, binary 37 rasters were utilised to analyse the predicted contraction, expansion, areas of no change and no occupancy.

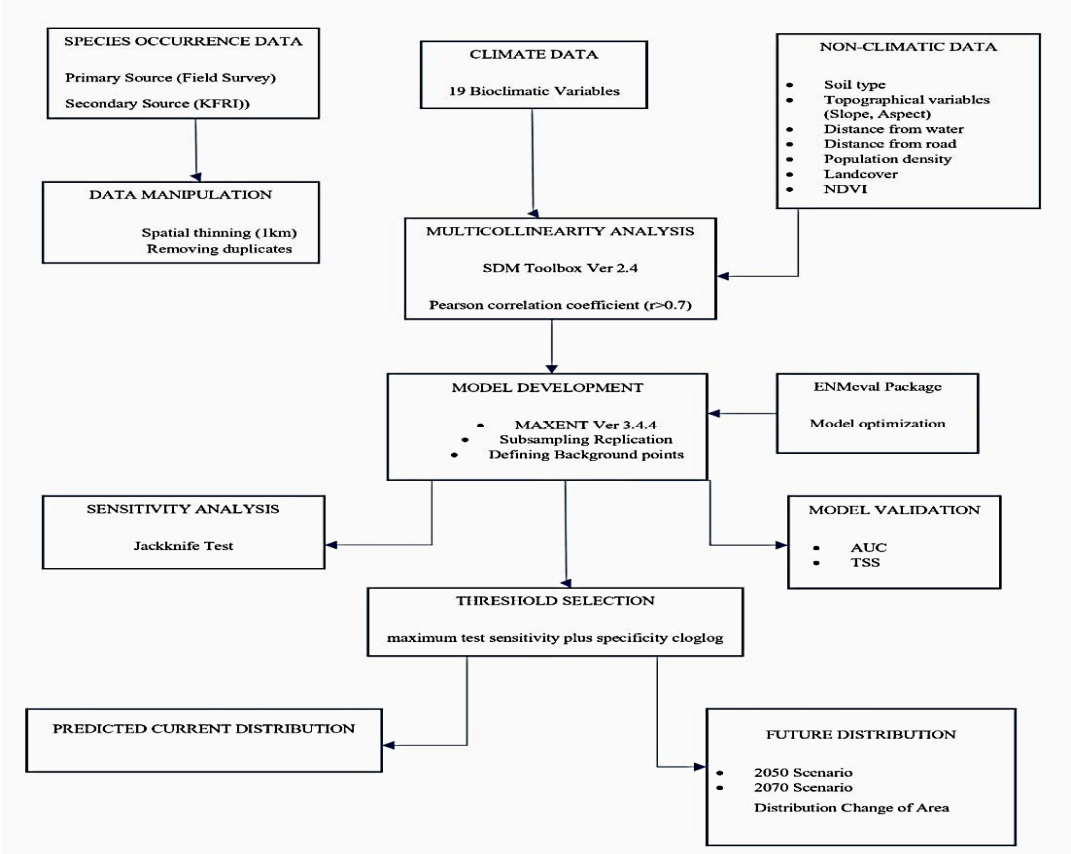
3.1.2.9. Potential Distribution under Future Scenarios

For the projected HadGEM2-ES climate change scenarios for 2050 and 2070 with 30 arc-second (~1 km) spatial resolution, as presented in the fifth assessment report (AR5) of the Intergovernmental Panel for Climate Change (IPCC, 2014), the impact of climate change on the potential distribution of the selected invasive species was

done using MaxEnT modelling. Environmental variables and species occurrence records were used to train the model by projecting a future environmental variable onto a set of current environmental variables.

Only the static non-climatic variables, for example, soil type, aspect and slope were kept. In contrast, dynamic and non-climatic variables and the variables with negligible permutation importance obtained from the training gain were removed. Also, the variables chosen after multicollinearity analysis were chosen. Models of different RCPs 2.6, 4.5, 6.0, 8.5 were done for the years 2050 (2040 – 2069) and 2070 (2060 – 2089) with ten replicates and 25 test percentage. The projection was done using a sub-sampling type of replication. Importantly, the layers should have the same name as the training data, and map projection and geographic dimensions must be the same. Maxent outputs were mapped using ArcMap ver.10.7.1, and the distribution area and the change in the distribution area were measured from the current and future binary species distribution maps (logistic threshold output, 0-1). The methodology flow diagram is shown in Figure 3.2.

Fig. 3.2. Methodology flow diagram



3.2. Study of pollination, seed dispersal, phenology and variability of *Senna spectabilis* populations in Kerala

3.2.1. Phenological Observations and Data Analysis

Phenological studies were carried out in Wayanad WLS and Anaikaty area of Attappady. Ten mature individuals growing in similar environmental conditions in different forest patches of Wayanad WLS and Anaikkaty were observed. Reproductive and healthy individuals between 10 to 12 m in height and diameter at breast height ≥30 cm were selected. The presence or absence of phenophases was recorded monthly for 12 months, from June 2018 to May 2021, and phenophases were classified as leaf phenology with leaf flushing, mature leaf and leaf abscission, reproductive phenophases classified as flower bud initiation and anthesis or open flower and immature fruit and mature fruits period for fruiting phenophases.

Flowering phenology was observed surface-based with a 10 x 50 binocular. Phenophases were analysed using the qualitative method, by the simple presence or absence, and the semi-quantitative method that ranks the phenophases in five intensity categories. This ranking scale ranges from 0 to 4, where 0 is absence, 1 represents 1 to 25%; 2 represents 26 to 50%, 3 represents 51 to 75% and 4 represents 76 to 100%. These semi-quantitative data were used to calculate the intensity index of the phenophases for each month by applying the formula proposed by Fournier (1974):

Fournier intensity index (FII)

$I = [(\Sigma i).(4n)-1] \times 100$

Where I = intensity index of a given phenophase;

Σi = sum of the intensity indexes of each sampled tree;

n = number of sampled trees.

Fournier intensity index (FII) shows the peaks of intensity, indicating a given phenophase occurrence more intensively in the population. It accentuates the calculated quantity of sprouting, senescence, flower buds, flowers and fruits produced and the number of individuals displaying a given phenophase.

3.2.1.1. Statistical Analysis

The statistical analyses were completed using Microsoft Office Excel 2010 and Spearman's correlation coefficients (r_s). The correlation between phenological events

and climatic variables, such as rainfall maximum and minimum temperature of the three-year study periods (June 2018 to May 2021) was examined with Spearman's rank correlation coefficient (Stalin and Swamy, 2018).

$$r_s = 1 - \frac{6 \sum d_i^2}{n(n^2 - 1)},$$

r_s = Spearman's rank correlation coefficient

n = number of data points of the two variables

d_i = difference in ranks of the "ith" element

Spearman's rank correlation coefficient, denoted by r_s is a numerical value such that $-1 \leq rs \leq 1$. It measures the likelihood of one variable increasing as the other increases (a direct association) or one variable decreasing as the further increases (an inverse association). Positive values indicate direct associations and inverse associations are shown by negative values. No association is indicated by a value of 0. The stronger the association, the closer r_s is to -1 or 1; the weaker the association, the closer it is to 0. Rank correlation coefficient values of 1 or -1 mean that the ranks agree entirely ($r_s = 1$) or are direct opposites ($r_s = -1$).

3.2.2. Reproductive Biology of *Senna spectabilis*

Reproductive studies were conducted at the model site established at Muthanga Forest, WWl Sanctuary and selected areas of Meppadi and Kalpetta part of the South Wayanad Territorial Forest Division of Kerala, India.

3.2.2.1. Data Collection

The plant species for the study was selected after carrying out a field study in Wayanad. Field investigations and experiments were conducted from September 2019 to January 2020 and from October 2020 to January 2022. Following a preliminary field study of the flowering seasons of the selected species, regular field studies were carried out to collect information and data on the reproductive aspects. The functional events of individual flowers, sexual status, floral rewards and their details, breeding system, flower visitors and their behaviour and pollination role, natural fruit and seed output rates, and fruit maturation duration were carefully examined, and seed dispersal aspects were examined. Floral structural and

functional aspects were studied, as per the methods of Raju and Reddi (1994), Raju and Rao (2004) and Dafni *et al.,* (2005).

3.2.2.1.1. Flower morphology: The details of flower morphology, such as flower sex, shape, size, colour, odour, sepals, petals, stamens and ovary, and the position of stamens, exposed or hidden, were described. The morphology and dimensions of the inflorescence were studied from the fresh inflorescence and those fixed in Formalin-Aceto- Alcohol under the microscope. The order of wilting or dropping off of floral parts was recorded. These details of the selected plant species were provided due to inadequate and confusing taxonomic descriptions. Inflorescence was observed from flower bud initiation up to fruit fall and recorded the seed production pattern (Reeder *et al.,* 2014).

3.2.2.1.2. Pollen-ovule ratio: The pollen-ovule ratio was determined by dividing the average number of pollen grains per flower by the number of ovules per flower. The value thus obtained was taken as the pollen-ovule ratio (Cruden, 1977).

3.2.2.1.3. Nectar characters: The presence of nectar was determined by observing the mature buds and open flowers. When the volume of nectar secreted was measurable, nectar from 10 flowers of each plant was determined. Then, the average nectar volume per flower was determined and expressed in µl (Dafni *et al.,* 2005).

The flowers used for this purpose were bagged at the mature bud stage, opened after anthesis, and the nectar was squeezed into micropipettes for measuring the volume of nectar. Nectar sugar concentration was determined with the help of a hand sugar refractometer.

3.2.2.1.4. Stigma receptivity: The stigma receptivity was observed visually and by the H_2O_2 test. In the visual method, the stigma's physical state (wet/dry) and the unfolding of its lobes were considered to record the commencement of receptivity, withering of the lobes was taken as loss of receptivity. The H_2O_2 test was followed to record the stigma receptivity period (Dafni *et al.*, 2005). This test is widely followed, although it does not indicate the exact location of the receptive area. In this study, the period of slow release of bubbles from the surface of the stigma following the application of hydrogen peroxide was taken as stigma receptivity.

3.2.2.1.5. Anther dehiscence: Anthesis was initially recorded by observing markedly mature buds in the field. Later, the observations were repeated three to four times on different days to provide an accurate anthesis schedule for this species. Similarly, the mature buds were followed to record the time of anther dehiscence. It was confirmed by observing the anthers under a 10x hand lens.

3.2.2.1.6. Breeding systems: In *S. spectabilis*, mature flower buds of some inflorescences on different individuals were tagged and enclosed in paper bags. A fixed number of flowers from different inflorescences was bagged or tagged and followed further to study whether the pollination is vector-dependent and to understand the flower abortion rate. Another set of flowers was used for experiments on apomixis, self-pollination, and cross-pollination, such as geitonogamy and xenogamy, to collect data for understanding the breeding behaviour. All these categories of flower pollination were followed for the fruit set. If the fruit set was present, the percentage of the fruit set was calculated for each mode.

3.2.2.1.7. Plant-pollinator interaction: Flower visitors were also observed concerning their mode of approach, landing, probing behaviour, forage collected, and contact with sex organs to effect pollination and inter-tree foraging activity. Foraging visits made by major pollinators were recorded on selected inflorescences.

3.2.2.1.8. Pollen viability: The viability of pollen at the time of dehiscence was tested using 1% acetocarmine, considering stained grains as viable and shrivelled grains as non-viable (Radford *et al.,* 1974, Koshy and Jee, 2001, Beena *et al.,* 2007). The viable pollen in the microscopic field was counted and expressed as a percentage of the total.

In-vitro germination of pollen was tested in five different germination media. Fresh mature anthers were collected from the field at anthesis, and pollen grains were carefully dusted on cavity slides containing germination media. One hour after inoculation, the number of pollen grains germinated, and the number of grains per field of view were recorded. Pollen grains were considered to have germinated when the pollen tube length was greater than the diameter of the pollen grain (Tuinstra and Wedel, 2000). Pollen diameter and tube length were observed under an image analyser (Leica Q 500 MC) at 40 x magnification (Figure 3a).

3.2.2.1.9. Seed viability: Seed viability is the measure of live seeds which can develop into plants. A germination test and cutting test were done to test viability. As per the International Seed Testing Association (ISTA), 800 (n=100 x 8) is the standard sample for germination tests of small sized seeds. However, because of high seed emptiness, 8000 (n=1000 x 8) seeds were used for the experiments (Ribeiro-Oliveira *et al.,* 2016). A manual cutting machine was used for seed cutting. The presence and absence of seeds within the fruits were recorded per standard methods. For germination experiments, seeds were sown in plastic trays (LBH= 30 cm x 40.5 cm x 7 cm) filled with vermiculite and maintained in the laboratory at room temperature (25^0C). Data were recorded from the start of germination to culmination as per standard methods.

3.2.3. Observations on Fruit and Seed Predation and Seed Dispersal

Observations on fruit and seed predation and seed dispersal were also studied in the established plot of Wayanad WLS. It recorded from June 2020 to May 2023. The areas were patches of tropical deciduous forest, and the plots are located at an altitude of 750 to 800 m.

Ten trees with pods were randomly chosen from the established plot. Four types of data were collected from these trees; direct observations having opportunistic data, seed dispersal, fruit and seed fate along transect for which we monitored fallen fruit and seeds and feeding signs along four transects located under the canopies of the ten trees and faecal analysis. Fresh faeces of herbivores were observed, identified and examined for the presence of seeds. The faeces were confirmed based on their characteristic size and shape (Jhala *et al.,* 2009). Herbivore faeces were collected near the *S. spectabilis* populations. The random faecal samples (pellets/dung) of herbivores were collected from the study sites. Opportunistic data were also incorporated for the dispersal study of seeds. Mega herbivores such as the *Elephas maximus* (Asian Elephant) and large herbivores like *Rusa unicolor* (Sambar Deer) and *Axis axis* (Chital) that consume dry pods of *S. spectabilis* and can disperse seeds were reasonably common in the studied plot of Wayanad WLS. Apart from them, faecal samples of the Indian hare (*Lepus nigricollis*) were also found.

3.2.4. Morphological Variation Study

The tree characters are associated with their heredity to a certain degree, while environmental factors highly change some. The growth of trees in a particular

geographical area depends on genetic and environmental factors. Several methods are available to estimate the variation between the populations of a tree species. For this study, variation between populations was assessed by analysing phenotypic characters. First it was done in seven populations of *Senna spectabilis* across Kerala. Eight characters of six-month and twelve-month seedlings from seven populations with ten replications were used to analyse the morphological variation among the populations. Collected seeds of seven populations such as Muthanga, Vythiri, Tholpetty, Azhinjilam, Meppadi, Anaikkaty and Thiruvananthapuram gave rise to seedlings in KFRI nursery. Morphological characters such as shoot height (SH), root length (RN), root collar diameter, leaf length (LL), leaf width (LW), root number (RN), twig number (TN) and chlorophyl content (CC) were measured by standard methods (Wight *et al.,* 2004; Diaz *et al.,* 2015; Perez-Harguindeguy *et al*., 2016; Xu *et al.,* 2016).

Sixteen phenotypic characters were selected based on the literature survey for estimating the variation between the eleven tree populations such as Ponkuzhy, Muthanga, Vythiri, Tholpetty, Begur, Azhinjilam, Meppadi, Anaikkatty, Kottathara, Periyar and Thiruvananthapuram (Al-Sagheer and Prasad, 2010; Capuzzo *et al.*, 2012; Dangi *et al.*, 2012; Diaz *et al.,* 2015; Guo *et al.,* 2017; Ashwath *et al.,* 2020). It included characteristics that can be easily observed and measured at any season of the year independent of different phenophases (girth at breast height (GBH), total tree height (HT), crown area (CA), number of primary branches (PB), number of secondary branches (SB), length of pods (LP) , diameter of pods (DP), seeds per pods (SP)) and characters that can be understood and measured only by tests (bark thickness (BT), sapwood moisture content (SM), leaf chlorophyll content (LC), leaf area (LA), leaf fresh mass (LF), leaf dry mass (LD), Germination percent (GM%), Viability percent (VP).

A minimum of ten individuals were selected from each population (n=11x10) for data collection. Standard protocols were followed for the selection of characters and measurements were carried out with the help of conventional instruments and tools (Wight *et al.,* 2004; Diaz *et al.*, 2015; Perez-Harguindeguy *et al.,* 2016; Xu *et al.,* 2016) (Tables.3-2,3-3, Plate.1.b-f). The measurement of each tree character was carried out as follows:

Table 3.2 Measurement details of the tree characteristics of *S. spectabilis* in the state of Kerala.

Sl. No	Name of character	Measurement
1	Girth at breast height (GBH)	The measurement of the circumference around the stem, taken in an upright position relative to the stem axis at breast height (1.37 m) was conducted in a region that was devoid of any damages, bumps, flutes, climbers, and close proximity to other trees. The measurement of GBH (girth at breast height) was taken in regions of the tree that were free from any defects.
2	Tree height (HT)	The distance between the upper boundaries of the primary photosynthetic tissues, excluding inflorescences, and the ground level, expressed in meters, is referred to as the shortest distance. To measure the height, one must move away from the tree until the tree tip is visible and then measure the horizontal distance from the base of the tree. The instrument's appropriate distance scale is established using the rotating rod. The instrument's pointer is released by pressing the side button, and the required point on the tree is sighted. After the pointer has settled, the trigger is pulled, and the height is directly read from the appropriate scale in meters. The base of the tree is then sighted and measured, and the height is determined by combining both measurements.
3	Crown area (CA)	The crown of a tree refers to the uppermost canopy, characterized by branches that extend from the main stem and support photosynthesis. The vertical length of a tree, excluding the clear bole, is commonly referred to as the crown length (CL). This measurement is obtained by subtracting the clear bole length from the total height of the tree. Another measure of the crown portion of a tree is the crown diameter (CD). To determine CD, the distance from the base of the tree to the tip of branches in four cardinal directions (south, north, east and west) is measured. The average of these measurements is then used in conjunction with the area formula to calculate the crown area.
4	Primary branches (PB) and Secondary branches (SB)	The number of branches includes both PB and SB. PB originates from the tree trunk and SB originates from PB. Both PB and SB were counted manually with the help of binoculars.
5	Bark thickness (BT)	The bark, which constitutes the outermost layer of the stem, is comprised of non-living tissues on the exterior and living tissues (vascular cambium) on the interior. The breast height of the tree was utilized to measure bark thickness (BT), whereby a square-shaped section of the bark was removed from the stem using mechanical tools, and its thickness was subsequently determined.
6	Sapwood moisture content	Sapwood, the outermost layer of the woody stem, is comprised of living cells and can be distinguished from heartwood based on its colouration. In this study, a square-shaped section of

	(SM)	sapwood was obtained by removing the bark with the aid of mechanical tools such as a chisel and hammer. The mass of both fresh and dry sapwood samples was determined using a weighing balance. To achieve the dry state, the samples were subjected to drying in a hot air oven. The sapwood moisture content (SM) was then calculated as the disparity between the fresh and oven-dried weights.
7	Leaf area (LA)	The measurement of leaf size is commonly assessed through the determination of leaf area, which refers to the one-sided or projected area of an individual leaf. In this study, the leaf area (LA) was quantified using a LI-3100C leaf area meter.
8	Leaf chlorophyll content (LC)	LC was measured using an instant SPAD (Soil-Plant Analysis Development) 502 plus chlorophyll meter. Before measuring the LC, the instrument was calibrated. Measurements were taken from the leaf lamina where leaf veins were not prominent. Six measurements were taken from a single leaf and the average value was taken as the LC.
9	Leaf fresh weight (LF) and Leaf dry weight (LD)	For estimating the leaf fresh (LF) and dried mass (LD), leaves were directly collected from plants without the loss of moisture content and brought to the laboratory. Fresh and oven dried mass of the leaf was estimated using weighting balance
10	Length of pods (LP), Diameter of pods (DP), Seeds per pod (SP)	Fruit characteristics, including the length of pods (LP), diameter of pods (DP), and seeds per pod (SP), are significant traits of a particular tree species. These parameters were assessed utilizing a Vernier caliper.
11	Germination percent (Gm%),Viability percent (VP)	Germination percent was calculated using pre-sowing treatments that indicate high germination rate with seeds collected from different locations and viability percent calculated by seed cutting test by selected seeds from different locations

Table 3.3 The instruments and tools utilized for the purpose of gathering and quantifying data

Sl. No.	Character	Code	Unit	Measuring instruments
1	Girth at breast height	GBH	m	Measuring tape (flexible)
2	Total tree height	HT	m	Hypsometer/rangefinder
3	Crown area	CA	m	Measuring tape (flexible) & Haga Altimeter
4	Primary branches	PB	-	-
5	Secondary branches	SB	-	-
6	Bark thickness	BT	mm	KEN CY 0- 150 mm digital caliper
7	Sapwood moisture content	SM	%	Sartorius weighing balance, Rotek hot air oven
8	Chlorophyll content	LC	-	SPAD 502 Plus chlorophyll meter
9	Leaf area	LA	cm^2	LI- 3100C area meter
10	Leaf fresh mass	LF	g	Sartorius weighing balance
11	Leaf dry mass	LD	g	Sartorius weighing balance, Rotek hot air oven
12	Length of pods	LP	cm	-
13	Diameter of pods	DP	cm	KEN CY 0- 150 mm digital caliper
14	Seeds per pod	SP	-	-
15	Germination percentage	GM%	-	-
16	Seed viability percentage	VP	-	-

3.2.4.1. Univariate Summary Statistics

Univariate summary statistics were analysed to ascertain fundamental statistical information pertaining to the data, including but not limited to minimum, maximum, mean, standard error, standard deviation, skewness, kurtosis, geometric mean and coefficient variance. Such basic statistical information facilitates comprehension of the degree of dispersion, the shape of the distribution, and an approximation of statistical dependence. The PAST software, as described by Hammer *et al.* (2001), was employed for the purpose of conducting univariate summary statistics.

Plate 3.1. a) Image analyser (Leica Q 500 MC) , **b)** KEN CY 0- 150 mm digital caliper **c)** Hypsometer/Rangefinder, **d)** Haga Altimeter,**e)** SPAD 502 Plus Chlorophyll Meter, **f)** LI- 3100C Area Meter

3.2.4.2. Analysis of Variance (ANOVA)

An ANOVA analysis was conducted utilizing IBM SPSS statistics software (version 26: 2018) (IBM, 2019) to examine morphological traits. The inclusion of data pertaining to morphological tree characteristics aids in validating the presence of variation among populations. The phenotypic traits of trees are subjected to ANOVA for further investigation.

3.2.4.3. Correlation Analysis

Correlation analysis was done using IBM SPSS statistics software (version 26: 2018) to estimate the relationship and strength between the characters used for variation studies.

3.2.4.4. Cluster Analysis

Cluster analysis is a method used to categorize a dataset into distinct clusters based on similarities within the data (Plotkin *et al.,* 2002; Suranto, 2002; Li-Hammed *et al.*, 2015; Abozeid *et al.,* 2017). In this particular study, all selected seedling populations were grouped into clusters based on their morphological characteristics at six and twelve months. The purpose of conducting cluster analysis on these populations was to determine their similarities and identify the most closely related populations. To achieve this, a dendrogram was constructed using the paired group UPGMA algorithm and Euclidean similarity index in the PAST software.

3.2.5. Karyotype Analysis

Variability studies were also done through karyotype analysis. The seed samples for the study from six populations were collected from different areas of Kerala during the summer of 2019 (Table 3.4), and seeds germinated in the nursery at Kerala Forest Research Institute, Thrissur.

Table 3.4 Populations of *Senna spectabilis* sampled for karyotype analysis

Locations	Latitude °N	Longitude °E	Altitude (m) (Msl)
Muthanga wildlife sanctuary	11.66	76.39	875
Vythiri	11.55	76.03	789
Anakkatty	11.11	76.74	527
Thiruvanthapuram	8.50	76.95	6.23
Azhinjilam	11.98	75.86	2.6
Munnar	10.14	77.17	1712

For the cytological observations, vigorously growing root tips were subjected to pre-treatment. Root tips were collected before 10 AM and then pre-treated in 0.2 M of 8-hydroxyquinine (0.29 g of hydroxyl quinoline dissolved in 100 ml distilled water) for three hours and fixed overnight in freshly prepared Carnoy's fluid (60% ethanol, 30% chloroform, and 10% glacial acetic acid).

The root tip was cut without any prior softening. The entire root cross-sections were excised by dabbing the surface with absolute alcohol with a fine brush before cutting since alcohol hardens the tissue immediately. Water-saturated and very soft samples were placed for 24 hrs in a solution of 30, 60, and 100 % ethylene glycol 4000 and kept in an oven.

The fixed sample was hydrolysed in 1N HCl for eight minutes at room temperature after washing with distilled water, stained with 1% aceto-orcein for 15 min, and squashed on a glass slide after slight warming.

The squashed cells were observed under a microscope (Olympus BX 61 TRF motorised microscope with cytovision 3.92), and photographs were taken using cytovision 3.2 software. The observations were repeated several times from different sets of slides. A minimum of five mitotic cells was used to determine chromosome number. Characters such as somatic chromosome number, genomic chromosome length, total form percentage, total chromosome length, and genomic chromosome volume were estimated by standard procedure (Mohanty et *al.*, 2006 and Li *et al.*, 1985). Analysis of variance (ANOVA) was carried out to test the significance of variations between the various characters among the populations using IBM SPSS statistics software (version 26: 2018) (IBM, 2019).

3.2.6. Evaluation of the Genetic Variability of *Senna spectabilis* in different Locations of Kerala

Evaluation of the genetic variability of *Senna spectabilis* populations was done with very young and tender leaves of eleven accessions collected from various locations in Kerala. Fresh and young leaf materials are the primary preference for acquiring DNA of high quality. Nevertheless, mature leaves encompass elevated amounts of polyphenols and polysaccharides, thereby posing a significant challenge in isolating DNA of good quality (Porebski *et al.,* 1997).

3.2.6.1. Sample Collection

Eleven accessions of *Senna spectabilis* were collected from various locations in Kerala (Table 3.5). Young and tender leaves were collected separately from each accession and placed in a zip lock cover containing silica gel crystal. The samples were transported to the laboratory in a box filled with ice. Leaf materials were stored in a freezer until DNA extraction.

Table 3.5 Selected accessions of *Senna spectabilis* in Kerala, India, with GPS coordinates

Sl. No.	Code	Location of samples	Leaf type	GPS coordinates	
1	Ss-1	Ponkuzhy	Young	11.693183	76.395081
2	Ss-2	Vythiri	Young	11.560220	76.040610
3	Ss-3	Muthanga	Young	11.674143	76.371295
4	Ss-4	Tholpetty	Young	11.944860	76.065040
5	Ss-5	Meppadi	Young	11.532544	76.038971
6	Ss-6	Azhinjilam	Young	11.98457672	75.867614
7	Ss-7	Begur	Young	11.874822	76.072633
8	Ss-8	Anaikatty	Young	11.114385	76.746002
9	Ss-9	Kottathara	Young	11.135161	76.700009
10	Ss-10	Periyar	Young	9.525922	77.217583
11	Ss-11	Thiruvananthapuram	Young	8.722080	77.028380

3.2.6.2. DNA Extraction

DNA extraction was performed using Himedia's HipurA plant DNA isolation kit, by modified CTAB method (Southern, 1975; Tai *et al.*, 1990; Sanghai *et al.*, 1984; Williams *et al.*, 1990). According to the method, 400 mg of the sample was crushed in 9 ml CTAB extraction buffer and incubated at 65^0C for 90 minutes, with occasional inversion after adding 90 µl of 2- mercaptoethanol. After incubation, samples were allowed to cool at room temperature for 5 minutes. Five ml of chloroform: Isoamyl alcohol (24:1) was added to the slurry and mixed gently by inverting the tubes for five minutes. It was then centrifuged at 3000 rpm for two minutes at room temperature, and the supernatant was transferred to a fresh tube and incubated at room temperature

for 30 minutes, followed by the addition of 25µl RNase A. DNA was precipitated by adding 6 ml of isopropanol. It was then centrifuged at 3000 rpm for five minutes. The supernatant was discarded, and the precipitate was re-suspended in eight ml of cold diluted CTAB wash buffer and incubated at room temperature for 20 minutes. It was again centrifuged (3000 rpm; 5 minutes) at room temperature and the supernatant was discarded. The residual pellet was re-suspended by adding 8 ml of cold 70% ethanol. It was centrifuged at 3000 rpm for five minutes at room temperature, and the supernatant was discarded. The pellet was air-dried to remove the traces of ethanol. The air dried DNA pellet was dissolved in elution buffer and stored at -20^{0}C until further use.

3.2.6.3. Quantification and Quality Check of DNA

Isolated genomic DNA was subjected to separation on ethidium bromide stained agarose gel (0.8% w/v) in 1x TAE under a constant voltage of 50V for one hour and 30 minutes to check the purity of the sample. Purified samples were quantified using a Nanodrop 1000 spectrophotometer.

3.2.6.4. Standardisation of Amplification

Temperature and time were optimised for standardisation by varying the annealing temperature from 55^{o}C to 60^{o}C for ISSR. Also, the time was by varying time from 45 sec to 55 sec. The temperature at which maximum amplification was observed was selected for the amplification.

3.2.6.5. ISSR Analysis

PCR were performed using different ISSR primers – (5’ CAACAACAACAACAA 3’), (5’ GATAGATAGATAGATA 3’), (5’ ACAGACAGACAGACAG 3’), (5’ CAGCAGCAGCAGCAG 3’), (5’ GACAGACAGACAGACA 3’), (5’ GAGGAGGAGGAG 3’), (5’ ACTGACTGACTGACTG 3’), (5’ GAGAGAGAGAGAGA 3’) (Mohanty *et al*., 2010) for amplification of genomic DNA. The PCR amplification was carried out in a 25 µl reaction volume containing 200 nM primer, 1x-2x PCR master mix, 20 ng of genomic DNA and the required amount of molecular biology grade water. PCR master mix contains all the reagents required to perform a standard PCR. The reaction conditions were: initial denaturation step at 94^{o}C for 5 min; 30 cycles comprising denaturation at 94^{0}C for 30 sec, annealing conducted for 30 sec (annealing temperature was according to Tm), and extension at 72^{o}C for 1 min and final extension step at 72^{0}C for 5 min.

The amplicons were subjected to electrophoresis on a 2% agarose gel in 1x TAE at a constant voltage of 50 V for a duration of one hour and thirty minutes. Subsequently, the gel was stained with ethidium bromide, visualised using a gel doc and photographed. The molecular weight of individual ISSR bands was determined using a 100 bp ladder. Furthermore, to ascertain any indication of variation among populations, the amplicons were separated using polyacrylamide gel electrophoresis (PAGE) and the bands were examined in detail.

Steps involved in polyacrylamide gel electrophoresis: An acrylamide-based polyacrylamide gel (8%) was prepared in accordance with the following protocol. Firstly, the glass plates were assembled, and the separating gel solution was prepared by combining all the reagents listed in Table 3.6. APS and TEMED were added to the monomer solution and mixed thoroughly by gently swirling. The gel solution was then poured into the glass plates, ensuring that no air bubbles were present. A comb was placed in the gel solution to create wells (lanes) for loading samples, again ensuring that no air bubbles were present. The gel was allowed to polymerize for 10 minutes.

Table 3.6 Composition of polyacrylamide gel

Sl. No.	Reagents	Volume
1	TAE / TBE Buffer	60 ml
2	Acrylamide	20 ml
3	TEMED	120 µl
4	APS (Ammonium persulfate)	300 µl

To run the gel, the binder clips and comb were removed, and the gel was fixed in the electrophoresis apparatus using binder clips. The apparatus was then placed in the gel running tank, and the inner chamber of the tank was filled with buffer. The loading tip was inserted a few millimeters from the well bottom, and samples (4 µl) were delivered into the well. The loading tip was rinsed with distilled water after loading a few times. To compare DNA size and quantity, 2 µl of 100 bp ladders were added. The power supply was attached by putting the lid on and setting the voltage to 160 V and the gel was run for 2 hours and 45 minutes. Staining solutions were prepared in accordance with the following protocol. Fixer was prepared by combining 60 ml of ethanol and 1ml of acetic acid and making up the volume to 500 ml (to be used three

times). Stainer was prepared by dissolving 0.5 g of silver nitrate in 500 ml of distilled water (to be used three times), and developer was prepared by dissolving 9 g of NaOH pellets and 500 µl of formaldehyde in 300 ml of distilled water. To stain the gel, after running, the power supply was switched off, and the gel plates were removed. The gel was then removed from the glass plates using a spatula and placed in the staining solutions (fixer for 15 minutes, stainer for 15 minutes, and developer for 7-8 minutes). The gel was destained until the bands were clearly visible. The samples were then subjected to analysis to determine the approximate molecular weight of the visualized DNA bands by comparing them with the 100 bp ladders (markers).

3.2.6.6. Data analysis

Gel doc system pictures of gel were used to score the data. After data scoring and analysis phylogenetic tree or a genetic grouping was done using UPGMA (Unweighted pair-group method using arithmetic averages) method in the software based on the binary matrix data by the program NTSYSpc.2.02 software. This cluster analysis/genetic grouping was carried out to assess the extent of similarity using simqual method and dissimilarity by simgend method between the populations. Plate 3.2.a-h depicts the procedures involved in the DNA analysis.

Plate 3.2.a) Spectrophotometer (Nanodrop, ND-1000), b) Gradient thermal cycler c) Preparation of acrylamide gel & pouring into the glass plates d) Glass plates fixed with the comb e) Centrifuge machine f) Eelectrophoresis apparatus g) Loading the DNA samples into the wells h) Gel doc system.

3.3. Development of Management Protocol for Controlling *Senna spectabilis* and Restoration Protocol using native species.

3.3.1. Development of a protocol for controlling *Senna spectabilis*

Several approaches have been used for managing *Senna spectabilis*, but to date, no single method alone is effective. Many of the management challenges involve killing this tree. Thus, an integrated management approach is more effective for *S. spectabilis* invasion and involves several possible and available methods.

According to the age of trees, a three-tire protocol has been standardised to manage *Senna spectabilis*. Both physical and chemical methods were adopted to manage this species. Experimental setups were designed for managing *Senna spectabilis* invasion, including mechanical and chemical methods (Lukosi, 1997; Wakibara *et al.*, 2002).

The study involved conducting experiments in designated sites located in the Muthanga and Tholpetty regions within the Wayanad Wildlife Sanctuary. The objective of these experiments was to effectively manage the invasive tree species, *S. spectabilis*. A total of 33 experiments were implemented as a pilot observation in an area highly invaded by *S. spectabilis* in Muthanga and Tholpetty as indicated in Table 3.7. From these experiments, a subset of treatments was selected based on their high percentage of mortality. These selected treatments were then applied in the established model site to assess their efficacy in controlling *S. spectabilis* growth across different growth forms, including seedlings and saplings, medium-sized and multi-branched trees, and large trees with a girth at breast height greater than 30 centimeters. Tools used for the eradication experiments are mentioned in Table 3.8.

The observations were recorded at regular intervals. The efficacy and success of the treatments were confirmed through the dryness/death of trees after the monitoring period. The trial and error treatments for pilot observation were performed twice a year, pre-monsoon and post-monsoon time. Since post-monsoon treatments were better,the final experiments were conducted in the post monsoon time.

3.3.1.1. Treatment plot layout

One hectare experimental plot at Muthanga was divided into twenty five subplots (20 m X 20 m) with more than 100 trees in various stages as follows

1. M1-T1 to M1-T5 (Mechanical treatment-1 to 5, for medium-sized and multi-branched trees)
2. M2-T1 to M2-T5 (Mechanical treatment-1 to 5, for large trees)

3. C1-T1 to C1-T5 (Chemical treatment-1 to 5, for medium-sized and multi-branched trees)
4. C2-T1 to C2-T5 (Chemical treatment-1 to 5, for large trees)

The treatments were applied to 50 trees (n) in the experimental block. Saplings and seedlings were considered a single category, and separate randomised blocks with 50 x 50 meter size were allocated, each with five replicate blocks for experiments S-T1 to S-T3 (treatments for seedlings and saplings). After the application of each treatment, regular monitoring was done for three weeks and later weekly monitoring for one year, again monthly evaluation for three years. Taken care for observing the treated patch of tree invaded area mainly for any re-sprouting that would come up from the treated trees, whether it was dried or it's leaves were yellowing and showed any symptoms of mortality. All sproutings and seedlings in the invasion area were removed immediately.

Table 3.7 Different treatments applied on *Senna spectabilis* invaded area for pilot observation

Growth form	Mechanical methods	Chemical methods
Seedlings/Saplings	1: Cut from ground level 2: Hand pulling/Uprooting	1. Foliar spraying of Herbicide 2. Slash weeding and apply chemicals
Medium sized (<30cm gbh) &Multi-branched trees	3. Uprooting the whole tree using excavator 4. Uprooting the whole tree by manually. 5. Cut from ground level 6. Cut from 30 cm above ground 7. Ring barking 9. Cut from ground level at every 2 months 10. Cut from the base and cover with soil 11. Cut the tree and bark stripping 12. Cut the tree from base and split the scion portion	3. Hatch and squirt -1 ft. wide above the ground and apply Herbicide 4. Root feeding (Petrol+Diesel+ Neem) 5. Cut from ground level & Apply Glyphosate 6. Cut from ground level & Apply $Cuso_4$ 7. Application of rock salt 8. Cut the tree and bark stripping –apply chemicals 9. Cut from the base and cover with soil and apply chemicals.

Large trees ($\geq$ 30 cm gbh)	13. Debarking at 1m around the tree (including collar region &cut the buttresses around the tree trunk) 14. Completely cut the tree at 50cm above the ground level and strip the bark including collar region 15. Uprooting the whole tree using JCB 16. Uprooting the whole tree by manually 17. Complete cut and fill soil 18. Cut the tree 1m above ground	10. Drill fill herbicide 11. Cut from ground level & Apply Glyphosate 12. Debarking & Apply $Cuso_4$ 13. Completely cut the tree at 50cm above the ground level and strip the bark including collar region and apply Glyphosate 14. Cut the root and apply chemicals 15. Cut stump treatment with herbicide

Table 3.8 List of tools used for eradication treatments.

Sl. No.	List of tools/instruments used for eradication treatments
1	Digging fork
2	Crow bar
3	Weed puller
4	Wood cutting Knife
5	Splitting ax
6	Chisel
7	Hammer
8	Digging hoe
9	Knapsack Sprayer
10	Squirt bottle
11	Syringe
12	Hire an excavator for uprooting trees

3.3.1.2. Statistical Data Analysis

All results were statistically analysed using ANOVA technique to calculate the effects of the treatment. All statistical analyses were conducted with SPSS Statistics (v. 21, SPSS). Statistical significance was tested at $p<0.05$.

3.3.2. Development of a Restoration Protocol for locations where the species is eradicated.

A field site of one hectare was selected for restoration in Muthanga Wildlife Sanctuary, where *Senna spectabilis* was eradicated in 2019. The area was subdivided into 20 m x 20 m plots with five planting modes to determine the restoration success. Active restoration methods were applied. The experiment was initiated in June 2019.

A total of 13 native species were identified as suitable for this area (*Terminalia crenulata, Garcinia gummi-gutta, Terminalia arjuna, Aegle marmelos, Mangifera indica, Shorea roxburgii, Bambusa bamboo, Ficus racemosa, Syzygium cuminii, Artocarpus heterophyllus, Ficus relegiosa, Phyllanthus emblica, Bohinia verigata, Limonia acidissima*) and seedlings with approximately six months of age and a mean height of 30 cm were produced in the KFRI nursery and Tholpetty forest office nursery. The experiment was laid out as an RBD, with five treatments by fast-growing species, a mixture of fast and slow growing species, and seeds broadcasted combined with three planting densities (2000 plants/ha, 3000 plants/ha, 4000 plants/ha).

Three evaluations were conducted in the third, sixth and ninth months after planting, as stated by Carvalho *et al.,* (2019). The assessment of survival was determined by considering the individuals that remained alive throughout all three evaluations.

Chapter-4

Results and Discussion

Chapter-4

Results and Discussion

4.1. Estimation of the Current Distribution and Abundance of *Senna spectabilis* in Kerala

4.1.1. Population Structure of *Senna spectabilis* in Kerala

In a survey conducted across 14 districts of Kerala, the presence of *Senna spectabilis* was observed in six districts (Fig. 4.1), namely Wayanad, Palakkad, Idukki, Kozhikode, Kannur and Thiruvananthapuram (Fig. 4.2 - 4.7). Notably, Wayanad, Idukki and Palakkad exhibited the invasive characteristics of this species, with dense populations and significant regenerative abilities.

The density of plants in various growth stages of *S. spectabilis* in Kerala (except Wayanad Wildlife Sanctuary) is given in Table 4.1. This particular sanctuary is home to the aggressive growth habitat of this invasive tree species, which is exclusively discussed in Chapter 4.1.2. The density classification of this species reveals that the highest tree density was observed in populations such as Begur, with a recorded density of 308 individuals per hectare, and in Meppadi, with a recorded density of 163 individuals per hectare, specifically along the border areas of the forest regions. These areas also exhibited high densities of regeneration in the form of seedlings. Invasive populations are depicted in Plate 4.1, and avenue populations in Plate 4.2.

The Vythiri region, among other areas, exhibited a notable concentration of seedling regeneration within the avenue populations. In Palakkad, specifically in the Attapady regions such as Anaikkatty, Sholayur and Kottathara, the avenue populations of *S. spectabilis* were predominantly found. These populations tended to establish new colonies close to the existing ones, resulting in a high growth density.

Close to Anaikkatty, populations of *S. spectabilis* were observed along the banks of a tributary of the Bhavani river. The dispersal of pods of this species through the river has contributed to a high regeneration density in the surrounding areas.

Another invasive species, *Senna siamea*, is commonly associated with *S. spectabilis*. It is recognized as a widely planted street tree and falls under invasive alien species. It has been observed that along the Anaiikkatty - Sholayur road, *S. siamea* has spread alongside *S. spectabilis*, particularly in roadside areas and degraded lands.

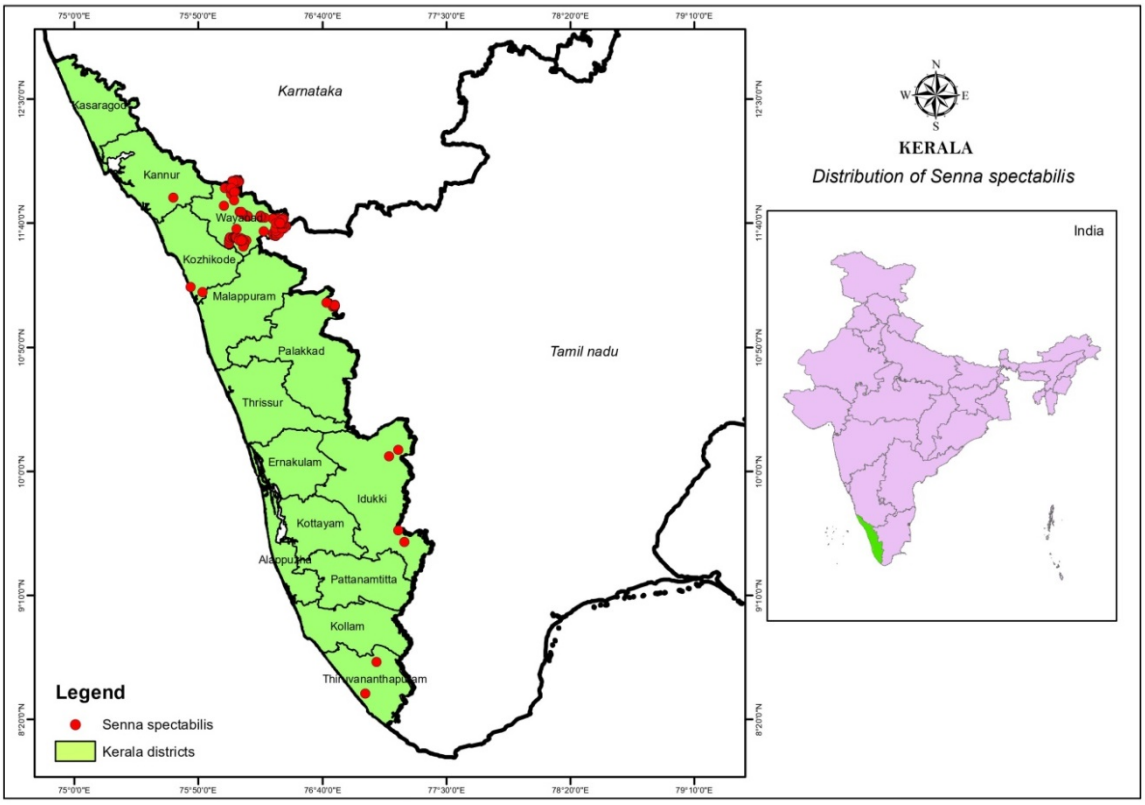


Fig. 4.1 Distribution of *Senna spectabilis* in Kerala

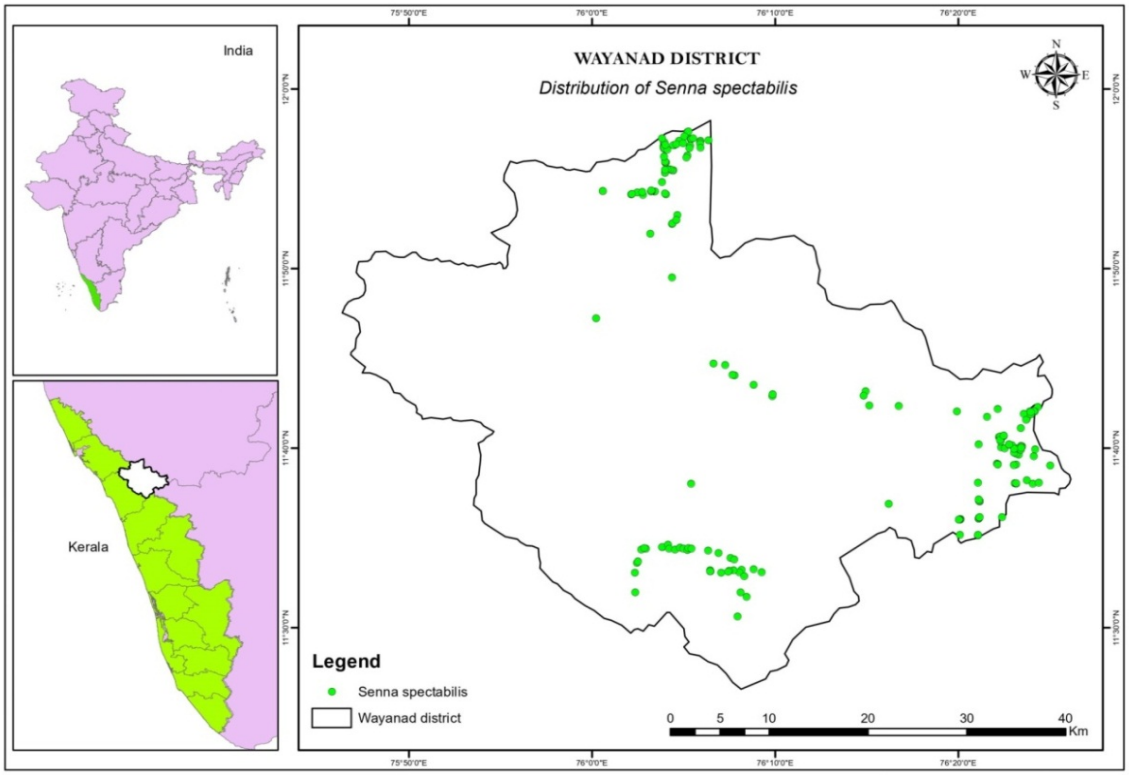


Fig. 4.2 Distribution of *Senna spectabilis* in Wayanad District

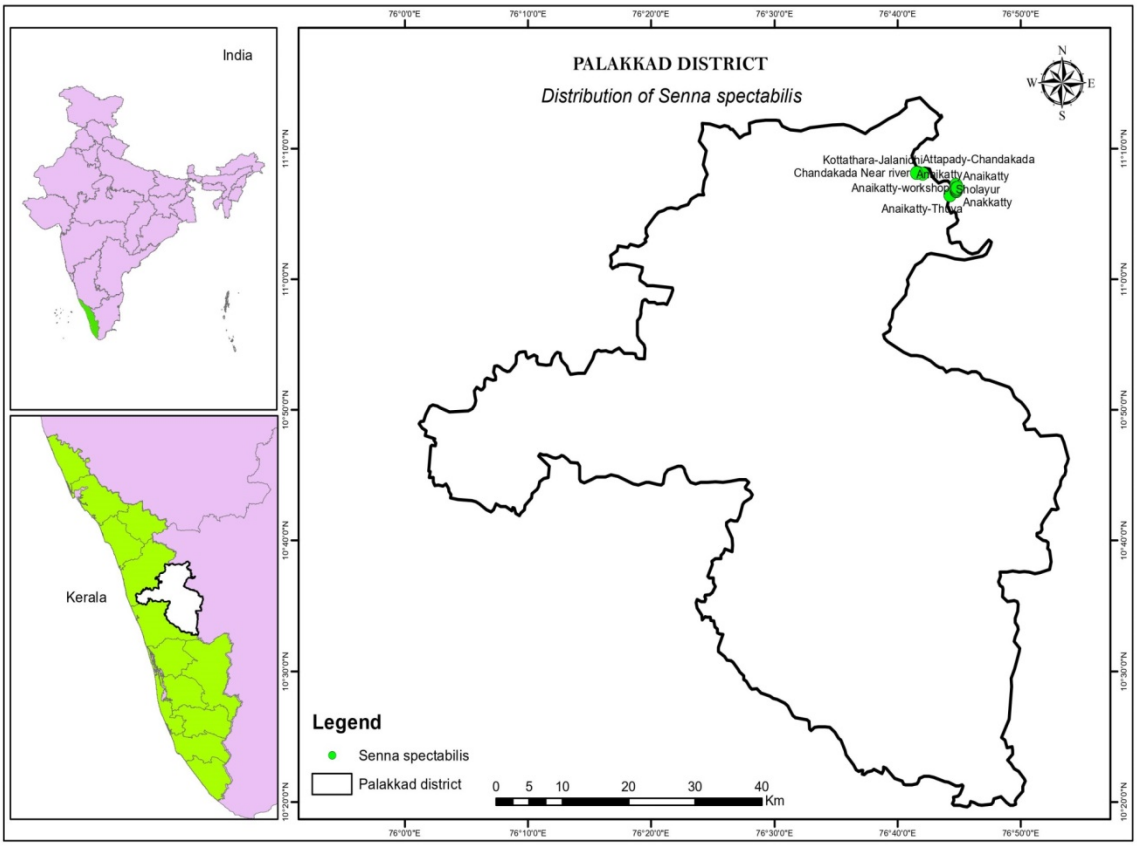


Fig. 4.3 Distribution of *Senna spectabilis* in Palakkad

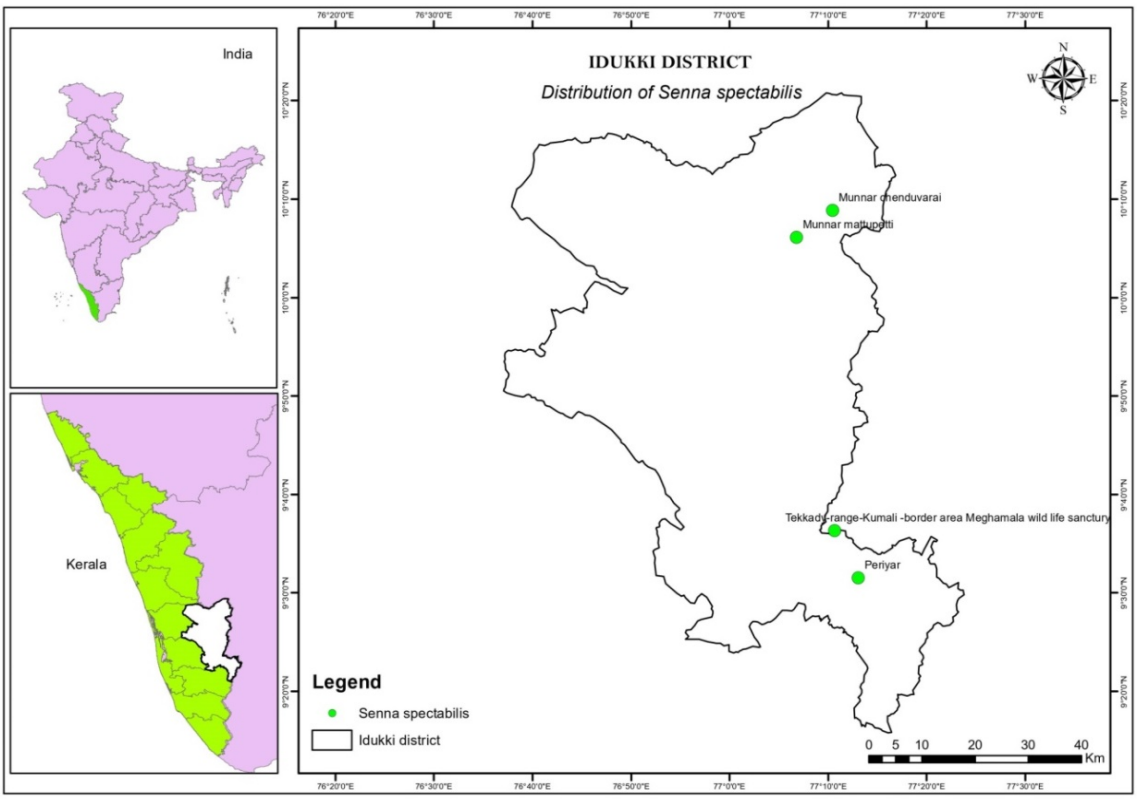


Fig. 4.4 Distribution of *Senna spectabilis* in Idukki

In the district of Idukki, it has been observed that in some regions of Munnar, such as Mattuppety and Chenduvarai, there is a significant presence of large avenue trees, while the natural regeneration of vegetation is minimal. Furthermore, higher concentrations of *S.spectabilis* have been noted in the periphery of the Periyar tiger reserve, particularly in the Kokkara and Thekkady areas. Similarly, in Kozhikode, there is a notable population of avenue trees along the road sides of the Ramanatukara-Azhinjilam area, with signs of regeneration near these trees.

Table 4.1 Density, Abundance and Frequency of various growth stages of *Senna spectabilis* in different locations of Kerala

Sl. No	Range/ Location	Habitat	Trees			saplings			seedlings		
			Density/ ha	Abundance	Frequency	Density/ha	Abundance	Frequency	Density/ha	Abundance	Frequency
1	Begur	MDF	325	13	1	308	12.33	1	733	29.33	1
2	Meppadi	MDF	163	6.55	1	186	7.44	1	413	16.55	1
3	Munnar	Avenue	25	1	1	0	0	0	0	0	0
4	Kottathara	Avenue	25	1	1	0	0	0	2	8	1
5	Anakatty	DDF	10	4.18	1	79	3.176	1	186	7.471	1
6	Sholayur	MDF	62	2.5	1	0	0	0	13	5.3	1
7	Vythiri	Avenue	58	2.33	1	75	3	1	1266	50.66	1
8	PTR	MDF	72	3.1	1	82	5.1	1	350	14	1

(PTR-Periyar Tiger Reserve)

Fig. 4.5 Distribution of *Senna spectabilis* in Kannur

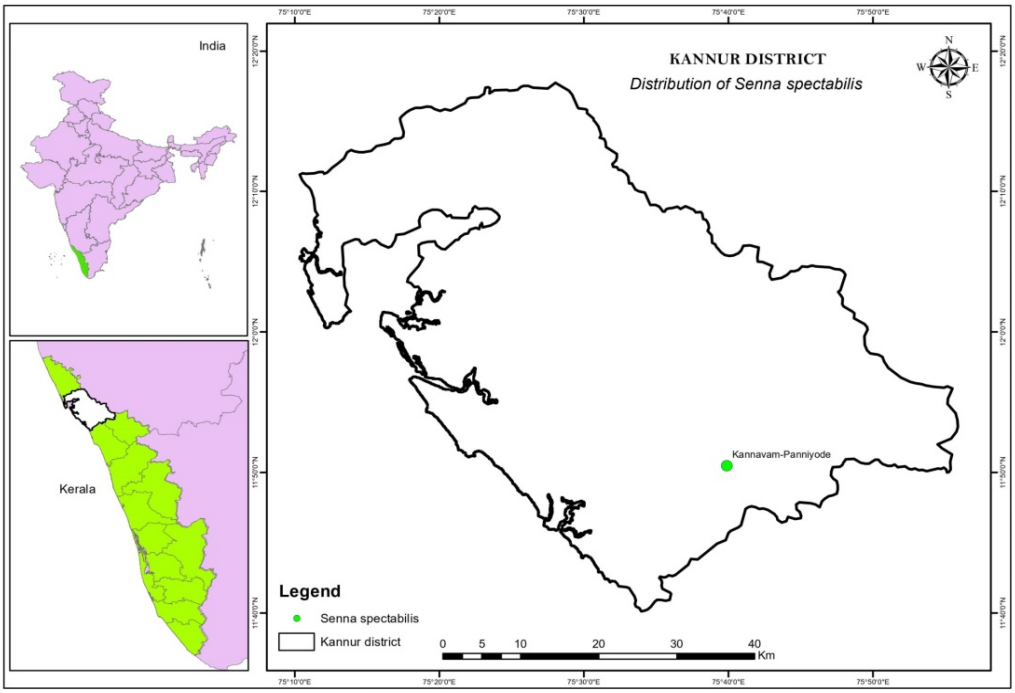


Fig. 4.6 Distribution of *Senna spectabilis* in Kozhikode

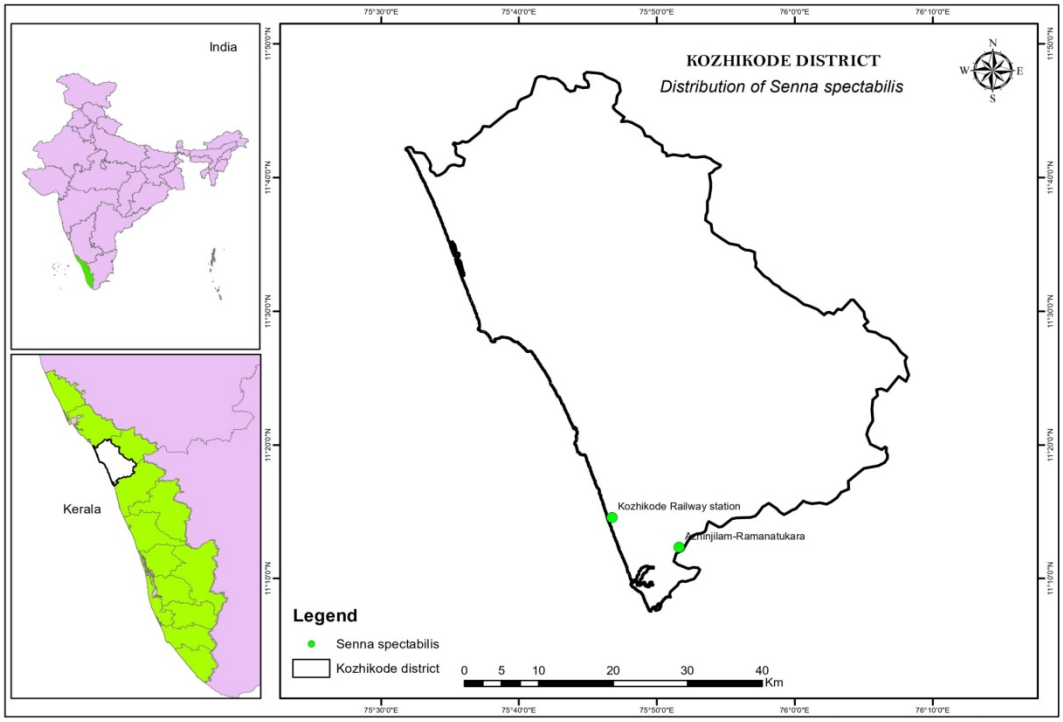


Fig. 4.7 Distribution of *Senna spectabilis* in Thiruvananthapuram

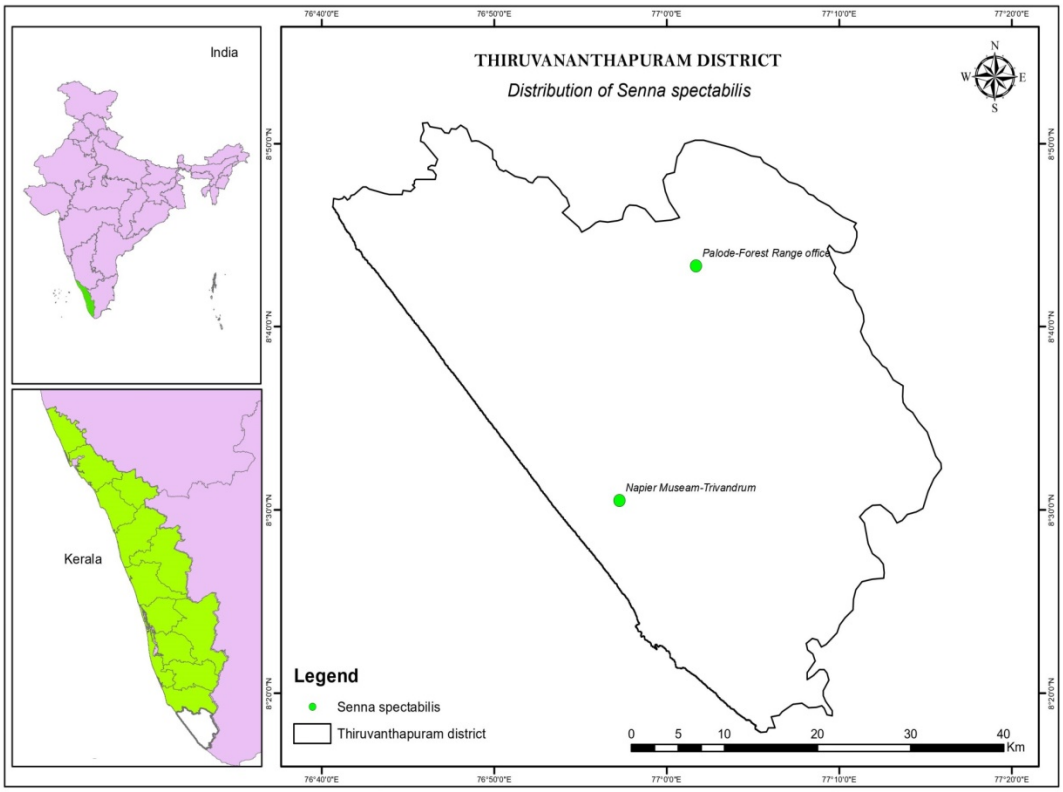


Plate 4.1(a-d). Invasive populations of *Senna spectabilis* in different locations of Kerala. a)Tholpetty, b)Muthanga, c)Meppadi, d)Anaikkaty

Plate 4.2 (a-h) Avenue trees of *Senna spectabilis* in different locations of Kerala. a)Sholayur, b)Vythiri, c)Chelode, d)Begur, e)Thiruvananthapuram, f)Ponkuzhy, g)Azhinjilam, h)Munnar

4.1.2. Assessment of vegetation composition and distribution pattern of *S.spectabilis* in Wayanad Wildlife Sanctuary

The vegetation analysis indicates that invasive alien plants, such as *S.spectabilis*, have had a significant detrimental impact on the ecosystem and the distribution patterns of plants within the Wayanad wildlife sanctuary. Quantitative data depict the status of the floristic composition and current distribution and dominance status of this tree invasive species.

4.1.2.1. Vegetation composition of different growth forms in Southern ranges

Distribution analysis of tree layer: 80 tree species were recorded in the tree layer. *Senna spectabilis* (DC.) H.S. Irwin & Barneby is the most dominant tree in terms of highest IVI value (40.02) (Table 4.3; Figure 4.8), followed by *Terminalia eliptica* Willd., *Pterocarpus marsupium* Roxb., *Tectona grandis* L.f. and *Terminalia anogeissiana* Gere & Boatwr. Eighty-five per cent of the tree species showed regular distribution and contagious or aggregated distribution was lowest (Table 4.2). Regarding frequency class, 68.75% of tree species are in the lower frequency class (A). The most frequent species found in the tree layer is *Cassia fistula* L., followed by *T. anogeissiana* and the invasive tree *S. spectabilis*. The observed Frequency class is almost similar to Raunkier's frequency class (Figure 4.10). *S. spectabilis* is the most densely recruited tree.

Distribution analysis of shrub layer: 51 shrub species were recorded in the shrub layer. *Chromolena odorata* (L.) R.M. King & H. Rob. showed the highest IVI value (Figure 4.8; Table 4.4), followed by *Senna spectabilis* (DC.) H.S. Irwin & Barneby. Because saplings of this species dominate over other species and it has a more basal area. *Lantana camara* L. possess third position. As per the abundance frequency ratio, 100% of shrub species in this study area showed contagious distribution, or it is in a discrete distribution that exhibits clustering (Neyman 1939) (Table 4.2). Regarding frequency class, 74.50% of the shrub species are in the lower frequency class (A). The most frequent species found in the shrub layer are saplings of *S. spectabilis* followed by *C. odorata* and *L. camara*. The observed Frequency class seems to vary with Raunkier's frequency class (Figure 4.11).

Distribution analysis of herb layer: 103 species were recorded in the herb layer. Species such as *Ageratum conyzoides* L., *Mimosa pudica* L., *Axonopus compressus* (Sw.) P.Beauv. and *Themeda triandra* Forssk. are the species showing

high IVI value and dominant over other species (Table.4.5; Fig.4.8). In these herb layer seedlings of *S. spectabilis* is also dominant. Under the storey of every *S. spectabilis* tree clumped distribution of many seedlings was observed. In frequency class *A. conyzoides, Tridax procumbens* L., and *A. compressus* are in the higher frequency class (Figure 4.12), and 97% of the herb species are in clumped distribution.

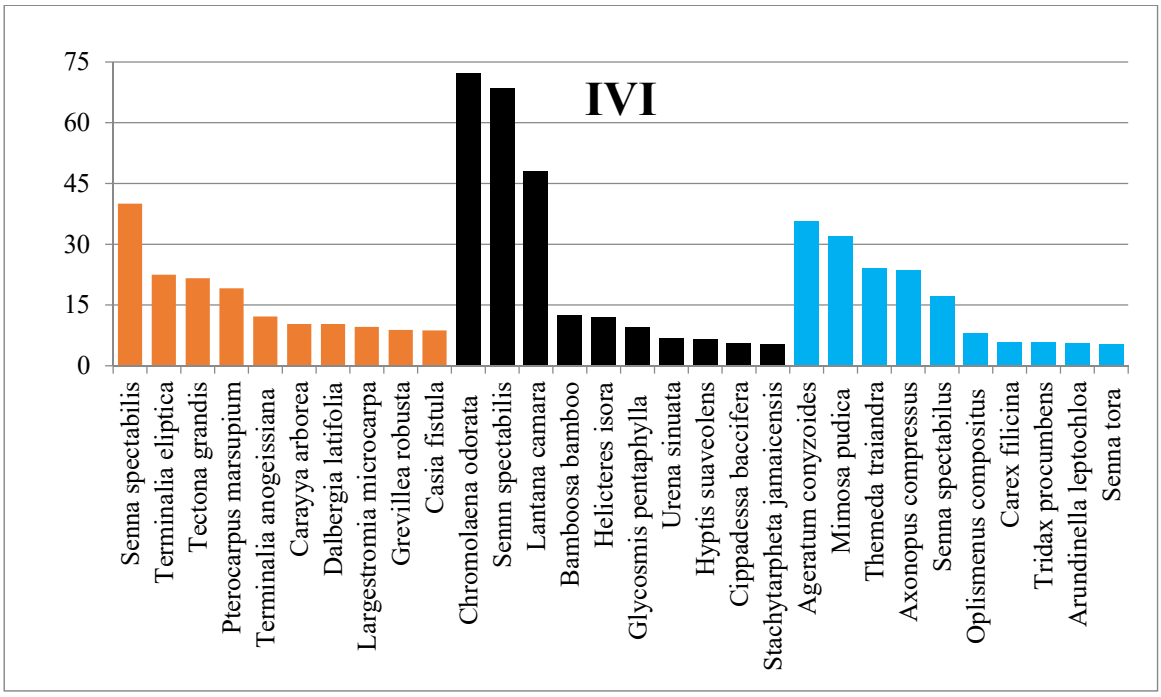


Fig. 4.8 Dominant species in Southern Ranges of Wayanad WLS, India (Red bar: Trees; Black bar: Shrubs; Blue bar: Herbs)

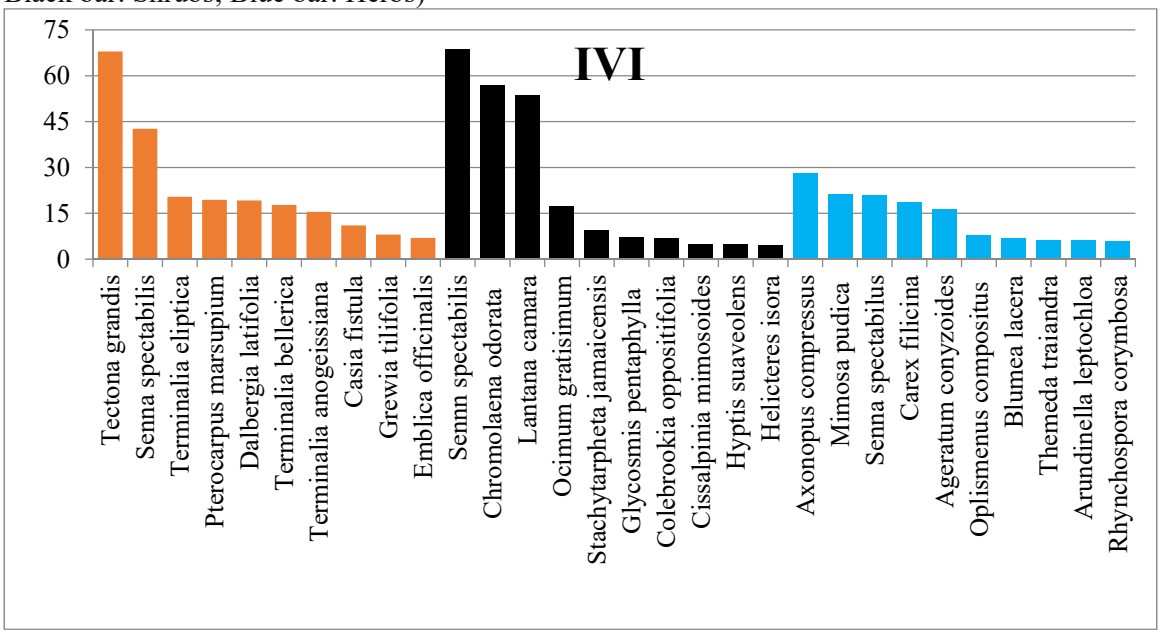


Fig. 4.9 Dominant species in the North-West Range of Wayanad WLS, India (Red bar: Trees; Black bar: Shrubs; Blue bar: Herbs)

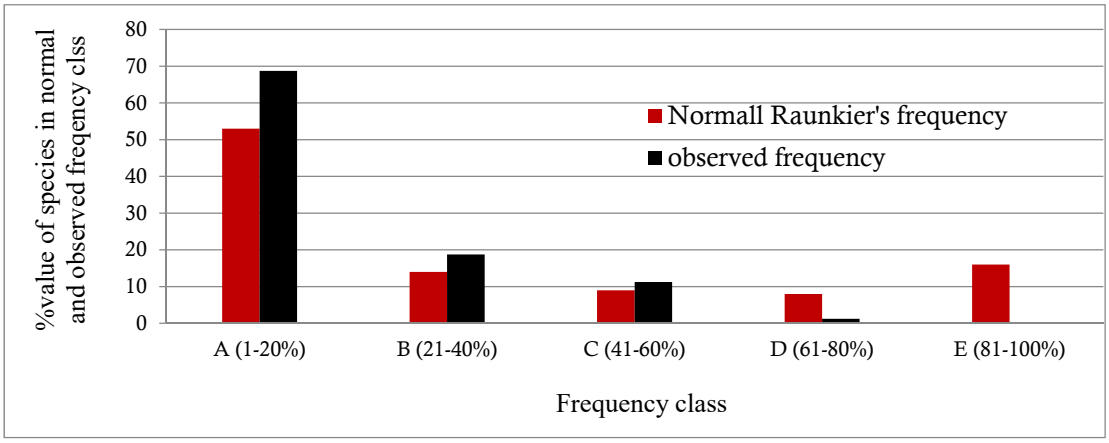


Fig. 4.10 Frequency class of tree layer in Southern Ranges

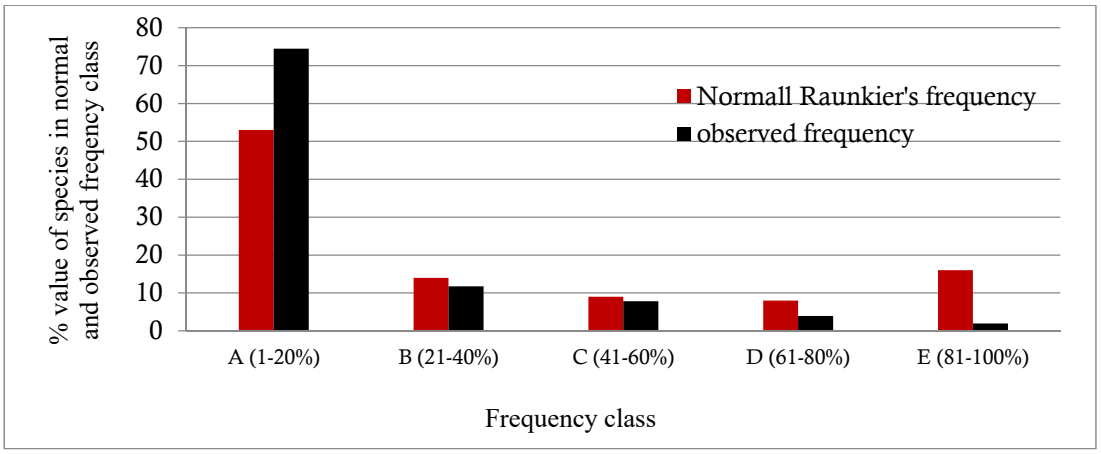


Fig. 4.11 Frequency class of shrub layer in Southern Ranges

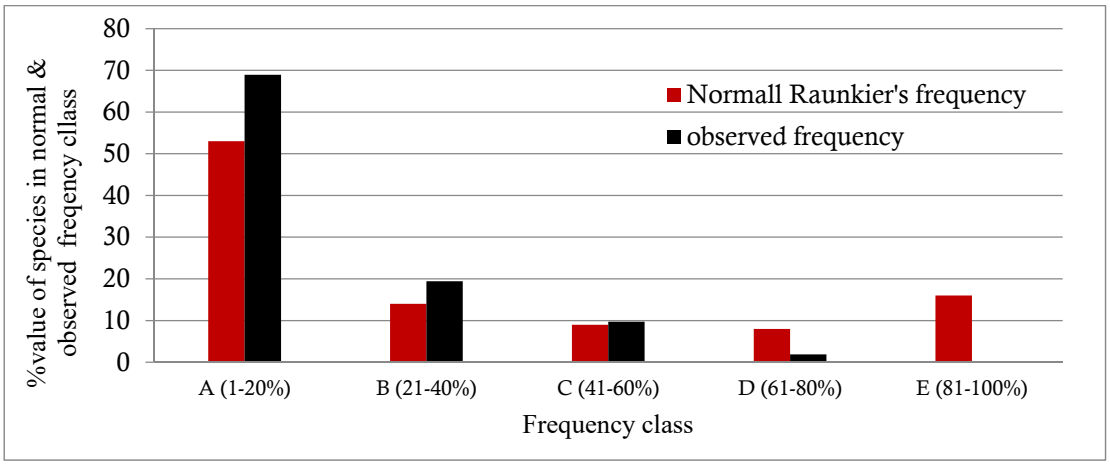


Fig. 4.12 Frequency class of herb layer in Southern Ranges

4.1.2.2. Vegetation composition of different growth forms in the North-West Range

Distribution analysis of tree layer: 47 tree species were recorded in the tree layer. *Tectona grandis* is the most dominant tree in terms of the highest IVI value (67.95) (Table 4.6; Figure 4.9), followed by *Senna spectabilis* (DC.) H.S. Irwin & Barneby, *Terminalia eliptica* Willd., *Pterocarpus marsupium* Roxb. and *Dalbergia latifolia* Roxb. 93% of the tree species showed contagious distribution (Table 4.2). Regarding frequency class, 80.85% of the tree species are in the lower frequency class (A) (Figure 4.13). The most frequent species found in the tree layer is the invasive tree *S. spectabilis.*

Distribution analysis of shrub layer: 43 shrub species were recorded in the shrub layer. Saplings of *Senna spectabilis* (DC.) H.S. Irwin & Barneby, *Chromolaena odorata* (L.) R.M. King & H.Rob., *Lantana camara* L. and *Osmium gratissimum* L. are the species that showed the highest IVI value (Table 4.6; Figure.4.9). Abundance frequency ratio indicates that 100% of the shrub species in this study area showed contagious distribution (Table 4.2). In terms of frequency class, 79.06% of the shrub species in the lower frequency class (A) and 11.62% in B class (Figure 4.14). *C. odorata* and *L. camara* are the most frequent and dense species in the shrub layer.

Distribution analysis of herb layer: 87 species were recorded in the herb layer. The species include *Ageratum conyzoides* L., *Mimosa pudica* L. and grass species such as *Carex stramentitia* Boott ex Boeckeler, *Axonopus compressus* (Sw.) P. Beauv. and seedlings of *Senna spectabilis* (DC.) H.S. Irwin & Barneby are the species showing high IVI value and dominant over other species (Table 4.7; Figure 4.9). Regarding frequency class, *Elephantopus scaber* L., *M. pudica* and *A. compressus* are in the higher frequency class (Figure 4.15). 96.45% of the herb species are in contagious distribution.

Table 4.2 Distribution pattern of vegetation in the study area

Stratum		Regular (%)	Random (%)	Contagious (%)
Southern Ranges	Tree layer	85	2.50	12.50
	Shrub layer	100		
	Herb layer	97		1.13
North-West Range	Tree layer	93		6.38
	Shrub layer	100		
	Herb layer	96.45		5.80

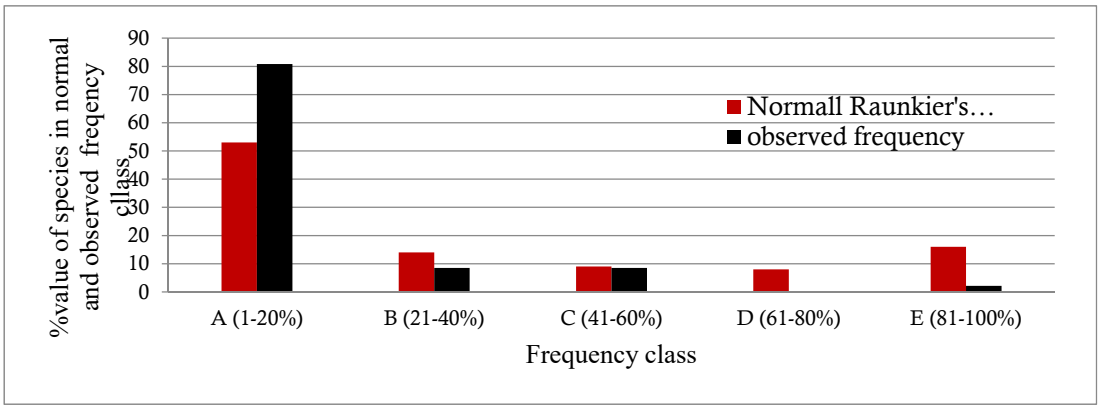


Fig. 4.13 Frequency class of tree layer in the North-west Range

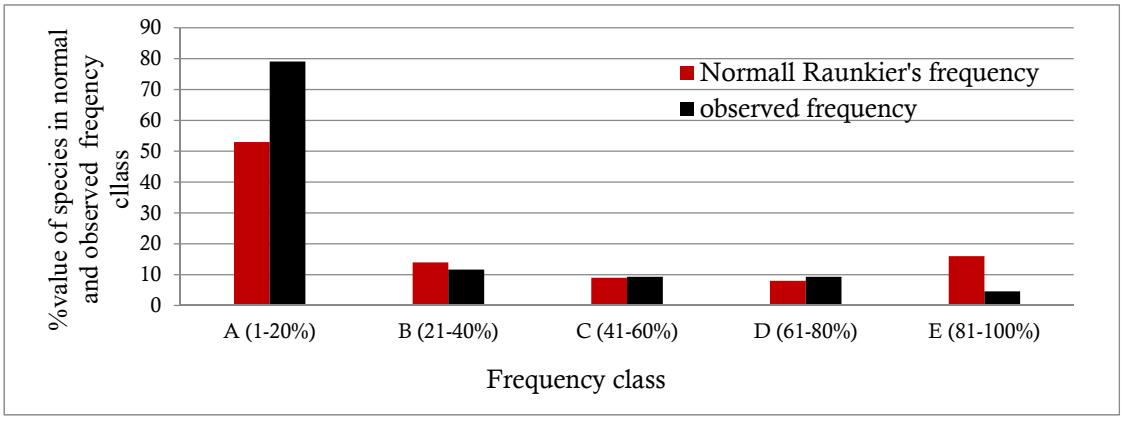


Fig. 4.14 Frequency class of shrub layer in the North- west Range

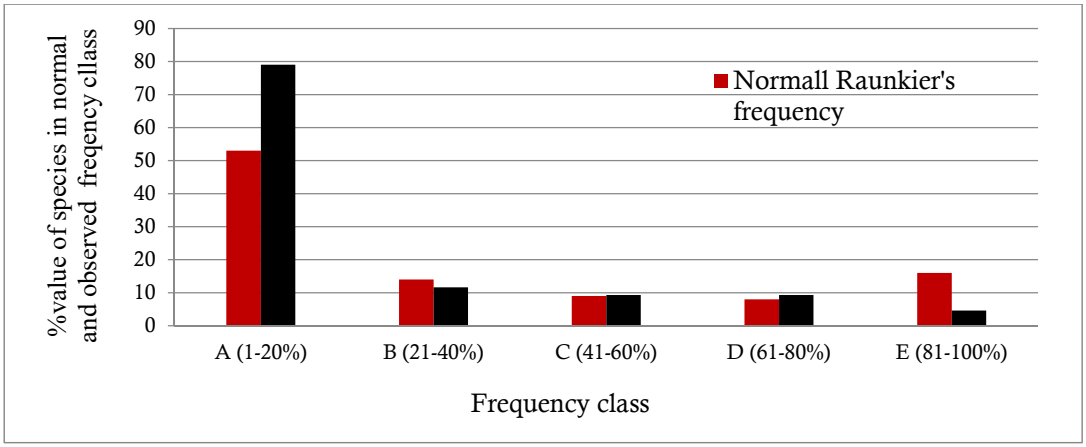


Fig. 4.15 Frequency class of herb layer in the North-west Range

In the two study sites, most of the plant populations show an aggregated or clumped pattern of species distribution. The structure of a population depends on the patterns of dispersal of the members of populations. The dispersion pattern refers to the spatial distribution of individuals within a population (Tiwari, 2004). Typically, individuals exhibit a clumped distribution, the most common pattern observed. However, if individuals tend to form groups of a specific size, the distribution of these groups may approach randomness. To understand a population's nature and accurate measure of its density, it is necessary to determine the type of distribution, the degree of clumping, and the size and permanence of groups. Sample methods and statistical analysis appropriate for random or uniform distributions may be inadequate or misleading when applied to strongly clumped distributions (Odum, 1996). In a given community, a limited number of species or species groups frequently exercise substantial influence over the population owing to their numerical abundance, size, productivity, or other activities despite the coexistence of numerous other organisms. In the present context, such dominant organisms are represented by invasive alien plants.

This study unequivocally demonstrates that the most dominant species observed in the shrub and herb layers are Invasive Alien Plants. Furthermore, we have included *S. spectabilis* within this category in the tree layer, as it exhibits a contagious distribution pattern and possesses a higher frequency class.

The tree layer in the studied sites is dominated by *Senna spectabilis* (DC.) H.S. Irwin & Barneby, while the shrub layer is dominated by *Lantana camara* L. and

Chromolaena odorata (L.) R.M. King & H. Rob. This has led to the replacement of native species in the region, with the native species now being relegated to a secondary status. The spreading foliage of *L. camara* creates impenetrable thickets, covering the ground thoroughly and promoting shade conditions that favor shade or moisture loving species. Our observations indicate that certain plants, such as *L. camara* and *C. odorata*, can occupy and multiply rapidly in dry, eroded, and disturbed localities, as well as in the understory of teak and eucalyptus plantations. The saplings of *S. spectabilis* are also very dense in these areas.

In the late 2000s, a notable gregarious flowering of bamboo occurred in Wayanad Sanctuary and its surrounding regions, which has exacerbated the situation by creating ample canopy openings and open lands for the spread of invasive species on that vacant space occupied by this tree-invasive *S. spectabilis*. Above the fallen and dried bamboo breaks, large thickets of *Mikania micrantha* Kunth, a climber invasive were spread. That species also has high density and more IVI value. These species are reducing the native vegetation. In the Muthanga range, Ambukuthi vayal areas, where *S. spectabilis* was present in low densities, the open lands were covered by the highly invasive *Lantana camara*.

4.1.2.3. Evaluation of Invasive Plant Species in Wayanad WLS

In this floristic study, 79 invasive plant species were identified. These invasive species that encroached Wayanad WLS were native to different geographic regions; the majority of the species, about 51 species, originated in American regions, mainly South America and tropical America, followed by Tropical Africa, Australia, and Asia, mainly southeast Asia. Some species came from Madagascar and the Mediterranean regions. Among them are six trees, 15 climbers, 20 shrubs, five aquatic, and 33 herbaceous invasive species. Asteraceae (17 species) was the dominant family, followed by Amaranthaceae, Convolvulaceae, Fabaceae, and Malvaceae. Asteraceae species produces many seeds and it has an excellent dispersal mechanism like wind dispersal and sticks on the animal fur. *Chromolaena odorata* (L.) R.M. King & H. Rob. and *Lantana camara* L. occupied the majority area of the sanctuary (Plate.4.3), mainly in abandoned plantations and open areas of Wayanad WLS, plants belonging to 36 families were reported**.** This study shows that *C. odorata* and *L. camara* are in the shrub layer, *Ageratum conyzoides* L. in the

herb layer and *Senna spectabilis* (DC.) H.S. Irwin & Barneby in the tree layer dominated the study area. *Maesopsis eminii* Engl. is a tree invader seen in the Kurichyad range in southern ranges that aggressively spreading species shows higher density and contagious distribution pattern. *Ocimum gratissimum* L., *Blumea lacera* (Burm. f.) DC. and *Hypoestes phyllostachya* Baker pose a significant threat in the study area. *B. lacera* has been observed to emerge between February and May and has since spread throughout the understorey and open vayal ecosystem. This invasive species has resulted in a reduction of forage for grazing herbivores.

According to a study conducted by Prajitha and Sudhabai (2022), it is apparent that the invasion of *S. spectabilis* is having a significant impact on the natural vegetation of WWS. Compared to the non-invaded forest areas, the ground cover in the invaded area was notably scarce. Additionally, their study found that other well-known invasive species, such as *Lantana camara* and *Chromolena odorata* were diminishing in the regions invaded by *S. spectabilis.* Consequently, it is imperative to prioritize the management of *S. spectabilis* over other documented invasive species in WWS.

Plate 4.3 a-e. Major Invasive Plants in WWL a) *Lantana camara*, *b) Chromolaena odorata, c) Bidens pilosa* var *minor,* d) *Mikania micrantha, e) Hypoestes sanguinolenta*

4.1.2.4. Population structure of *S. spectabilis* in Wayanad Wildlife sanctuary.

The invasion of *Senna spectabilis* in the Wayanad Wildlife Sanctuary was identified as a significant predicament. By examining phytosociological characteristics within the sanctuary, the populations of *Senna spectabilis* were categorized into areas of high density, medium density, and low density. The Muthanga region exhibited the highest recorded density and abundance, followed by Tholpetty.

The occurrence of *Senna spectabilis* in various forests in the Wayanad Wildlife Sanctuary ranges is depicted in distribution maps (Figures 4.16 to 4.18), while further details regarding density and distribution can be found in Tables 4.9 and 4.10.

Kakkapadam, Thakarappady, Mudumalakallu, located in the Muthanga forest ranges, followed by the state border and Ponkuzhy in S. Bathery, Tholpetty F.S, Kaimarm, and Kattapallam are recognized as high-density areas for *S. spectabilis*, as well Veetikutty, Pacahay, Thathoor regions in Kurichyad range possess higher densities of this tree population.

Fig. 4.16 *S. spectabilis* distribution in Muthanga-Wayanad Wildlife sanctuary.

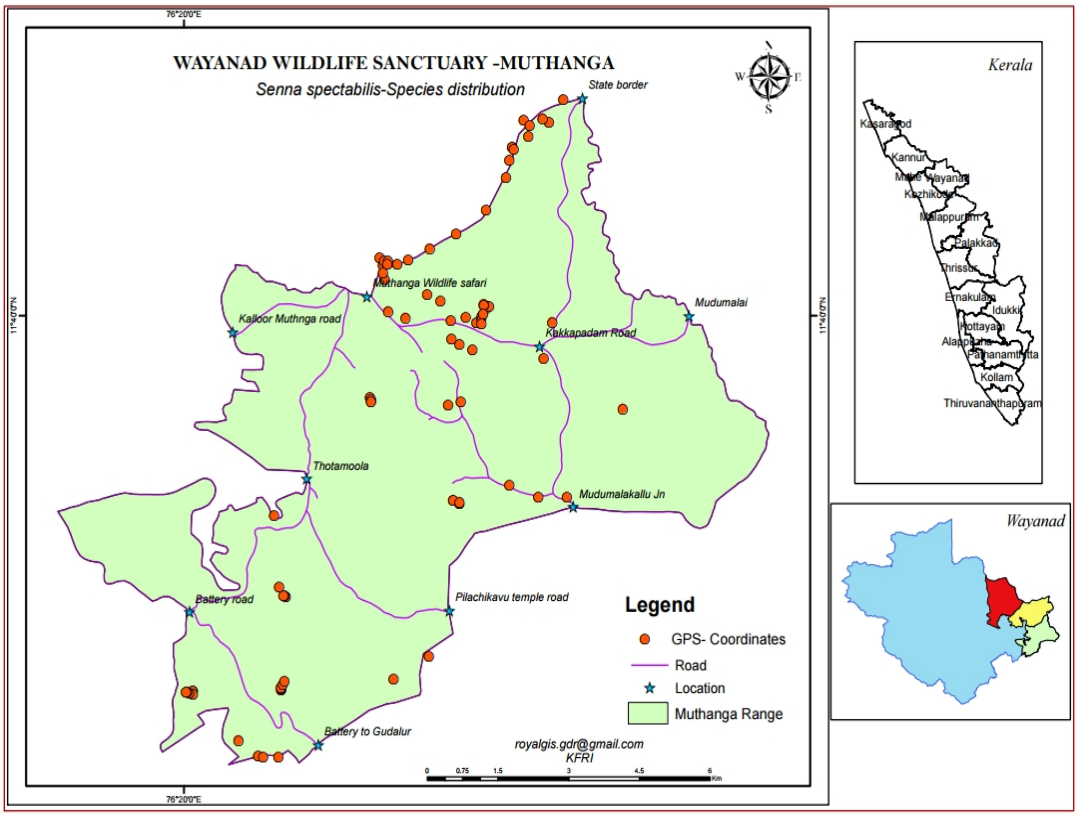


Fig. 4.17 *S. spectabilis* distribution in Sulthan bathery-Wayanad Wildlife sanctuary.

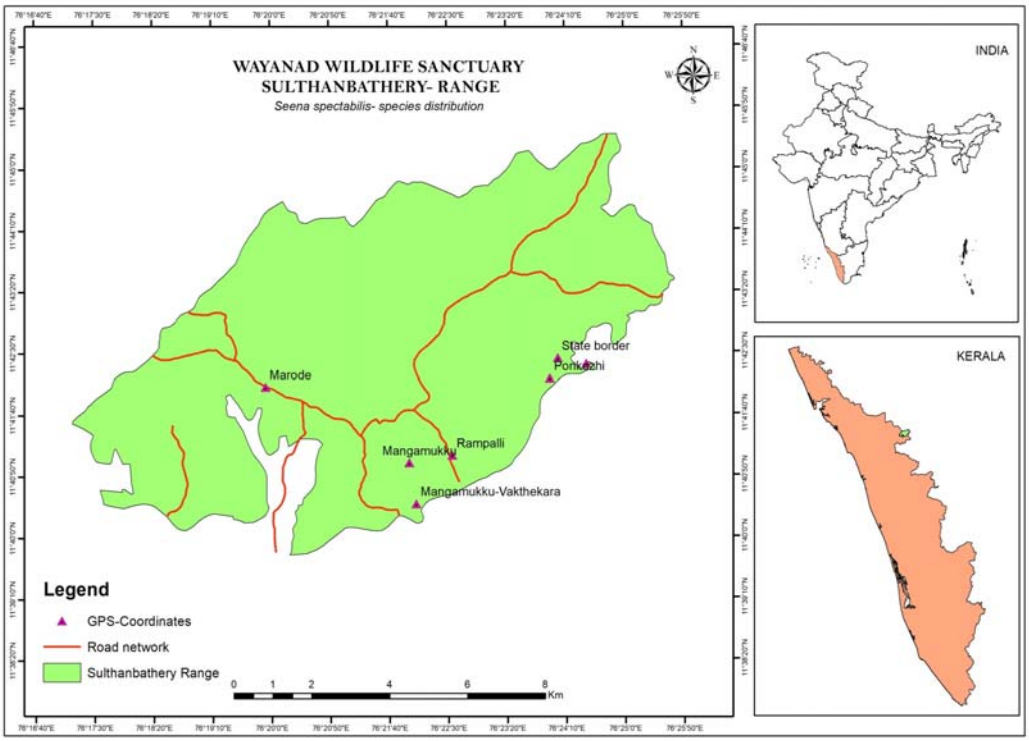
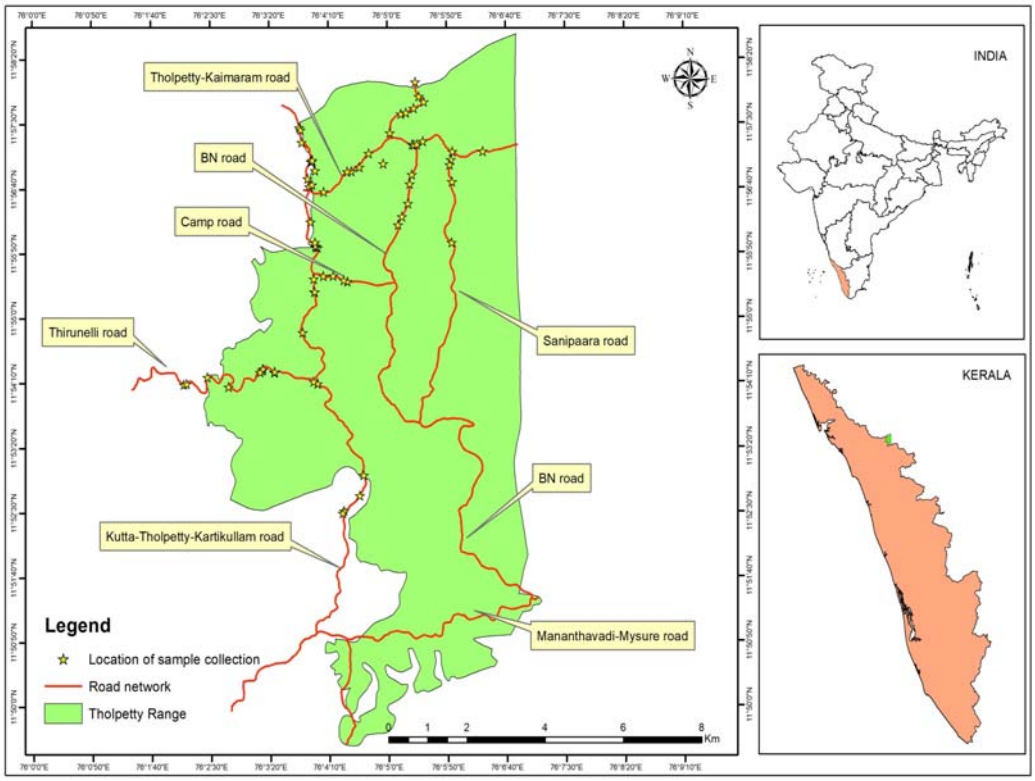


Fig. 4.18 *S. spectabilis* distribution in Tholpetty-Wayanad Wildlife sanctuary.



The species has demonstrated a continuous expansion in these areas of elevated population density. Medium-density areas are Maragadha, Ambukuthi vayal in Muthanga and Mangamukku, Rampalli in Sulthan bathery, Bavaly, Thirunelly-thet road and Narimanthikolli in Thopetty range. In other regions, the presence of this particular tree was identified and classified as areas of low density.

In the southern ranges of the sanctuary, *S. spectabilis* was predominantly distributed along the Kerala –Karnataka state border areas of Sulthan bathery and Muthanga ranges (Fig. 4.16-4.17), predominantly in Ponkuzhy area (402 tree individual ha^{-1} with high density of seedlings and saplings). A dry deciduous forest patch lies along this track of border areas of the Sulthan bathery range, and the occurrence of *S. spectabilis* is absent. Regions, namely Thakarapady near Muthanga forest station, onwards to Kakkapadam, a prominent vayal area, the *S. spectabilis* has aggressively invaded. The stretch is nearly 5.5 km in Kakkapadam. The density of tree individuals ranges from 702 $ha^{-1}$and in the case of seedlings, around 9500 individuals ha^{-1}.

According to the Kerala Forests and Wildlife department, it has been reported that as a component of the Social Forestry's shade tree planting programme in 1986, seedlings of *S. spectabilis* were initially cultivated in the Ponkuzhy area within the Muthanga range. The seedlings were planted close to the Muthanga forest station and along the periphery of the range office. Fifteen seedlings were planted in the Muthanga region, while additional seedlings were planted in the Wayanad territorial division's Meppadi and Aanappara regions. After seven years since its initial planting, the *S. spectabilis* plant finally reached its first flowering stage. The blossoming of these vibrant yellow flowers proved to be highly captivating, successfully drawing the attention of numerous tourists. After 15 years of planting, these trees attained a GBH of 270 cm, proving the fast-growing character of *S. spectabilis* (Harilal, 2019).

The observed high-density areas of this invasive are situated in the Vayal systems of WWL Sanctuary. Vayals (Swamps /low-lying grasslands) in the sanctuary are the edaphic climax. The main features of the Vayal system are that they have deep clayey soils and will be waterlogged in the rainy season. An area of 715.79 ha is under Vayals inside the sanctuary. They are called 'Vayal' as they sustain grasses yearly. Because of waterlogged conditions, there are only a few trees. These open grasslands are the main sites of herbivores foraging. Muthanga extends up to 189.792 hectares,

Sulthan Bathery- 147.003 ha, Kurichyad- 217.004 ha and Tholpetty 162.00 ha (Source: GOK, 2012).

The lamentable circumstance is that areas along the banks of the Vayal system, particularly Kakkapadam near Muthanga, are being invaded by the highly dense *S. spectabilis*. This invasive tree species is encroaching upon the Vayals and poses a significant threat to preserving these pristine ecosystems, potentially leading to their complete eradication in the future.

This study observed Kaimaram, in the Tholpetty range, under a higher density infestation area with tree individuals of 470.6 ha^{-1} (Fig. 4.18, Table 4.9). The research conducted by Harilal (2019) investigated the impact of invasive alien plants on understorey vegetation in the Tholpetty range of the Wayanad Wildlife Sanctuary. The study revealed that the Kaimaram section exhibited the highest density level, while the Bavali section demonstrated significantly lower infestation of *S. spectabilis* compared to the overall sampled area. Their results stated that *S. spectabilis* invasion accounted for 0.37% of the sampled area in the Vayal ecosystem, with an observed density of 589 individuals per hectare in the plantation areas of the study sites.

A study conducted by Prajitha and Sudhabai (2022) revealed that the Wayanad Wildlife Sanctuary has been invaded by *S. spectabilis*, covering approximately 25% of the sanctuary. The peripheral 500m area of both Muthanga and Tholpetty forest ranges show a significant absence of native and endemic flora, including ground cover herbaceous vegetation and shrubs. The study also found a positive correlation between the loss of herbivore habitat and the abundance of *S. spectabilis* in heavily invaded sites, which include monoculture plantation sites and natural forest regions. The present study also concurs with these results.

Wakibara and Mnaya (2002) reported that approximately 225 hectares, which account for 10% of the entire Mahale Mountain National Park in Tanzania, had been invaded by *S. spectabilis*. The density of the *S. spectabilis* trees in the Mahale Mountains N.P. was 586 per hectare. Similarly, in the current study conducted in the WWL sanctuary, the density of *S. spectabilis* individuals ranged from 400 to 700 ha^{-1}.

Based on these observations, the impact of *S. spectabilis* in the forests of Wayanad Wildlife Sanctuary was deemed highly significant, resulting in the rapid spread and distribution followed by the decline of native species.

Table 4.3 Vegetation composition of Tree layer in Southern ranges

(D: density, F: frequency; A: abundance; TBA: total basal area; RD: relative density; RF: relative frequency; RDo: relative dominance; IVI: important value index; DBN: distribution; RFC: Raunkiers frequency class)

Sl No.	Species Name	D	F	A	TBA	RD	RF	RDo	IVI	DBN (A/F)	RFC
1.	*Alstonia scholaris*	0.43	41.80	1.04	0.42	1.31	3.33	0.08	4.72	0.02	C
2.	*Terminalia anogeissiana*	3.90	58.60	6.64	15.00	5.85	3.33	2.99	12.16	0.11	C
3.	*Bauhinia racemosa*	0.06	6.70	1.00	0.09	0.20	0.54	0.02	0.76	0.15	A
4.	*Bischofia javanica*	0.04	3.01	1.50	0.06	0.14	0.24	0.01	0.39	0.50	A
5.	*Bombax ceiba*	0.13	12.00	1.12	0.22	0.41	0.96	0.04	1.41	0.09	A
6.	*Bombax insigne*	0.03	3.30	1.00	0.05	0.10	0.27	0.01	0.38	0.30	A
7.	*Bridelia retusa*	0.27	26.70	1.04	0.32	0.84	2.13	0.06	3.03	0.04	B
8.	*Buchanania lanzan*	0.08	5.20	1.57	0.82	0.25	0.42	0.16	0.83	0.30	A
9.	*Bucharia axilaris*	0.04	4.50	1.00	0.04	0.14	0.36	0.01	0.50	0.22	A
10.	*Buhinia malabarica*	0.06	5.60	1.06	0.07	0.18	0.45	0.01	0.64	0.19	A
11.	*Butea monosperma*	0.16	13.50	1.20	0.08	0.50	1.08	0.02	1.59	0.09	B
12.	*Carayya arborea*	1.01	42.30	2.36	19.60	3.03	3.36	3.91	10.30	0.06	C
13.	*Caseria tomentosa*	0.03	2.64	1.28	0.05	0.10	0.21	0.01	0.32	0.48	A
14.	*Casia fistula*	1.22	63.40	1.92	0.06	3.66	5.03	0.01	8.71	0.03	D
15.	*Cleistanthus collinus*	0.06	5.20	1.28	0.06	0.20	0.42	0.01	0.64	0.25	A
16.	*Cretava magna*	0.03	2.26	1.50	0.06	0.10	0.18	0.01	0.29	0.66	A
17.	*Dalbergia lanceolaria*	0.28	23.30	1.22	0.21	0.86	1.86	0.04	2.76	0.05	B
18.	*Dalbergia latifolia*	0.71	36.20	1.96	26.50	2.14	2.88	5.29	10.30	0.05	B
19.	*Dalbergia sissoides*	0.14	12.80	1.11	2.31	0.43	1.02	0.46	1.91	0.09	A
20.	*Diospyros cordifolia*	0.07	4.50	1.50	0.54	0.20	0.36	0.11	0.67	0.33	A
21.	*Dolichandrone arcuata*	0.08	7.10	1.15	0.05	0.25	0.57	0.01	0.83	0.16	A
22.	*Eliocarpus tuberculatus*	0.02	1.13	1.16	0.01	0.06	0.09	0.00	0.15	1.03	A
23.	*Emblica officinalis*	0.52	23.80	2.19	2.36	1.56	1.89	0.47	3.92	0.09	B
24.	*Eucalyptus grandis*	1.20	9.05	11.00	18.10	3.62	0.72	3.61	7.94	1.22	A
25.	*Ficus mysoorensis*	0.02	1.50	1.00	0.04	0.05	0.12	0.01	0.17	0.67	A
26.	*Gemilina arborea*	0.26	20.40	1.31	0.86	0.80	1.62	0.17	2.59	0.06	B
27.	*Grevillea robusta*	0.91	15.80	5.76	24.40	2.74	1.26	4.87	8.86	0.36	A
28.	*Grewia tilifolia*	0.31	18.10	1.75	11.30	0.95	0.72	2.25	3.92	0.10	A
29.	*Haldinia cordifolia*	0.14	9.05	1.58	5.42	0.43	0.72	1.08	2.23	0.17	A
30.	*Holigarna grahamii*	0.04	3.01	1.65	0.01	0.15	0.54	0.00	0.69	0.55	A
31.	*Hopea parviflora*	0.03	2.20	1.33	0.04	0.09	0.18	0.01	0.28	0.60	A
32.	*Hopea ponga*	0.14	6.03	1.18	1.63	0.43	0.48	0.33	1.23	0.20	A
33.	*Hydnocarpus pentandra*	0.08	6.79	1.27	0.08	0.26	0.54	0.02	0.82	0.19	A
34.	*Kydia calcyna*	0.12	8.31	1.45	1.82	0.36	0.66	0.36	1.38	0.17	A
35.	*Lannea coramandalica*	0.04	3.01	1.30	0.03	0.12	0.24	0.01	0.37	0.43	A
36.	*Largestromia microcarpa*	0.46	31.70	1.41	28.60	1.38	2.52	5.71	9.60	0.04	B
37.	*Largestromia parviflora*	0.36	32.40	0.79	0.68	1.11	2.58	0.14	3.82	0.02	B

38.	*Litsea floribunda*	0.04	3.30	1.30	0.07	0.14	0.27	0.01	0.42	0.39	A
39.	*Lucena lucocephala*	0.11	6.03	1.93	0.39	0.35	0.48	0.08	0.91	0.32	A
40.	*Macaranga peltata*	0.55	41.50	1.37	0.05	1.65	3.30	0.01	4.96	0.03	C
41.	*Mallotus tetracoccus*	0.09	8.41	1.18	0.02	0.29	0.66	0.00	0.96	0.14	A
42.	*Mangifera indica*	0.10	6.79	1.50	0.08	0.32	0.54	0.02	0.87	0.22	A
43.	*Melia azedararach*	0.09	8.30	1.18	0.09	0.29	0.66	0.02	0.97	0.14	A
44.	*Melia dubia*	0.14	13.20	1.08	0.25	0.43	1.05	0.05	1.53	0.08	A
45.	*Mesopsis eminii*	0.98	5.60	17.50	0.81	2.96	0.45	0.16	3.57	3.12	A
46.	*Milusa tomentosa*	0.02	1.80	1.20	0.04	0.07	0.15	0.01	0.23	0.67	A
47.	*Mimosops elengi*	0.07	6.70	1.16	0.07	0.24	0.54	0.01	0.79	0.17	A
48.	*Narigi crenulata*	0.30	29.40	1.03	0.08	0.92	2.34	0.02	3.27	0.04	B
49.	*Neolamarckia cadamba*	0.05	4.90	1.15	0.78	0.17	0.39	0.16	0.71	0.23	A
50.	*Olea dioica*	0.58	37.30	1.57	0.34	1.76	2.97	0.07	4.80	0.04	B
51.	*Persia macrantha*	0.13	9.81	1.34	0.61	0.40	0.78	0.12	1.30	0.14	A
52.	*Pongamia pinnata*	0.09	7.10	1.60	0.08	0.29	0.57	0.02	0.88	0.23	A
53.	*Pterocarpus marsupium*	0.97	48.10	2.00	61.90	2.92	3.86	12.35	19.13	0.04	B
54.	*Radermarchea xylocarpa*	0.06	3.39	1.66	0.38	0.17	0.27	0.08	0.52	0.49	A
55.	*Randia gardeneri*	0.07	4.50	1.60	0.31	0.23	0.36	0.06	0.65	0.36	A
56.	*Salix tetrasperma*	0.06	4.50	1.30	0.09	0.18	0.36	0.02	0.56	0.29	A
57.	*Schleichea oleosa*	0.33	17.40	1.91	23.60	1.00	1.38	4.71	7.08	0.11	A
58.	*Schrebera swietenioides*	0.14	6.79	2.11	2.36	0.21	0.27	0.47	0.96	0.31	A
59.	*Semecarpus anacardium*	0.29	27.90	1.05	0.16	0.88	2.22	0.03	3.13	0.04	B
60.	*Senna siamea*	0.60	45.60	1.30	3.21	1.82	3.65	0.64	6.12	0.03	C
61.	*Senna spectabilis*	7.63	49.10	15.60	32.40	29.66	3.89	6.46	40.02	0.32	C
62.	*Shorea roxburghii*	0.21	6.79	3.16	5.01	0.64	0.54	1.00	2.18	0.47	A
63.	*Spathodium companulatum*	0.13	4.10	3.18	0.06	0.40	0.33	0.01	0.74	0.78	A
64.	*Stereospermum colias*	0.30	6.02	1.45	14.20	0.93	1.32	2.83	5.08	0.24	A
65.	*Symplocos racemosa*	0.04	3.70	1.30	0.08	0.15	0.30	0.02	0.46	0.35	A
66.	*Syzigium cumini*	0.27	12.80	2.11	5.61	0.81	1.02	1.12	2.95	0.16	A
67.	*Syzigium travancuricum*	0.01	0.37	4.00	0.01	0.05	0.03	0.00	0.08	10.81	A
68.	*Tabernaemontana alternifolia*	0.55	45.70	1.22	0.14	1.67	3.62	0.03	5.33	0.03	C
69.	*Tamilnadia uliginosa*	0.81	13.60	1.30	0.42	0.44	0.60	0.08	1.12	0.10	A
70.	*Tectona grandis*	1.01	41.80	2.40	81.40	3.02	2.34	16.24	21.60	0.06	C
71.	*Terminalia bellerica*	0.27	23.00	1.18	0.22	0.81	1.83	0.04	2.69	0.05	A
72.	*Terminalia eliptica*	1.47	34.30	4.29	76.90	4.42	2.73	15.34	22.49	0.12	B
73.	*Terminalia paniculata*	0.09	6.79	1.33	2.25	0.54	1.08	0.45	2.07	0.20	A
74.	*Trema orientalis*	0.41	40.70	1.02	0.41	1.26	3.24	0.08	4.57	0.03	C

75.	*Trivia nudiflora*	0.09	7.90	1.14	0.06	0.27	0.63	0.01	0.91	0.14	A
76.	*Triwia nudiflora*	0.03	1.80	1.60	0.08	0.09	0.15	0.02	0.26	0.89	A
77.	*Vateria indica*	0.14	10.90	1.06	0.57	0.35	0.87	0.11	1.33	0.10	A
78.	*Vitex altisima*	0.24	21.80	1.13	0.06	0.75	1.74	0.01	2.50	0.05	B
79.	*WrightiaWrightiatea arborea*	0.04	2.26	2.00	0.48	0.14	0.18	0.10	0.41	0.88	A
80.	*Ziziphus glabarata*	0.23	22.60	1.06	0.08	0.71	1.77	0.02	2.50	0.05	B
81.	*Ziziphus mauritiana*	0.06	6.03	1.12	0.06	0.20	0.48	0.01	0.70	0.19	A

Table 4.4 Vegetation composition of Shrub layer in the Southern range

(D: density, F: frequency; A: abundance; TBA: total basal area; RD: relative density; RF: relative frequency; RDo: relative dominance; IVI: important value index; DBN: distribution; RFC: Raunkiers frequency class)

Sl No.	Species Name	D	F	A	TBA	RD	RF	RDo	IVI	DBN (A/F)	RFC
1.	*Ageratina adenophora*	1.06	5.60	18.70	0.06	1.08	0.71	0.10	1.89	3.34	A
2.	*Bamboosa bamboo*	0.06	5.60	1.06	7.40	0.06	0.71	11.70	12.47	0.19	A
3.	*Barleria cristata*	0.29	6.70	4.20	0.08	0.30	0.85	0.13	1.27	0.63	A
4.	*Boehmeria glomerulifera*	0.06	5.20	1.14	0.01	0.06	0.66	0.02	0.74	0.22	A
5.	*Calotropis gigantea*	0.17	6.03	2.87	0.02	0.18	0.75	0.03	0.96	0.48	A
6.	*Capparis brevispina*	0.06	1.50	4.50	0.01	0.07	0.19	0.02	0.28	3.00	A
7.	*Catunaregam spinosa*	0.15	15.40	1.51	0.08	0.24	1.93	0.13	2.29	0.10	A
8.	*Chassalia curviflora*	0.11	10.60	1.10	0.01	0.12	1.32	0.02	1.46	0.10	A
9.	*Chromolaena odorata*	45.4	71.30	63.7	10.60	46.48	8.89	16.81	72.17	0.89	D
10.	*Cippadessa baccifera*	1.23	34.30	3.60	0.02	1.27	4.28	0.03	5.58	0.10	B
11.	*Clerodendron infortunatum*	0.34	20.40	1.68	0.06	0.35	2.54	0.09	2.98	0.08	B
12.	*Colebrookia oppositifolia*	1.28	23.00	5.57	0.07	1.31	2.87	0.11	4.29	0.24	B
13.	*Crotalaria spectabilis*	0.26	4.10	6.40	0.01	0.27	0.52	0.01	0.80	1.56	A
14.	*Datura innoxia*	0.14	3.01	4.80	0.03	0.15	0.38	0.05	0.58	1.59	A
15.	*Decaschita crotoniflora*	0.05	3.39	1.50	0.01	0.05	0.42	0.01	0.49	0.44	A
16.	*Flemingia grahamiana*	0.23	20.30	1.12	0.02	0.24	2.54	0.03	2.81	0.06	B
17.	*Glycosmis pentaphylla*	1.94	55.80	3.47	0.41	1.99	6.96	0.65	9.59	0.06	C
18.	*Grewia hirsuta*	0.07	3.01	2.37	0.01	0.07	0.38	0.01	0.46	0.79	A
19.	*Helicteres isora*	0.86	43.00	2.00	3.60	0.88	5.36	5.69	11.94	0.05	C
20.	*Helixanthera obtusata*	0.15	9.81	1.61	0.08	0.16	1.22	0.13	1.51	0.16	A
21.	*Hibiscus furcatus,*	0.11	2.26	2.25	0.02	0.12	0.28	0.03	0.43	1.00	A
22.	*Hibiscus lobatus*	0.09	6.70	1.33	0.03	0.09	0.85	0.05	0.99	0.20	A
23.	*Hyptis capitata*	0.07	3.39	2.30	0.01	0.08	0.42	0.01	0.52	0.68	A
24.	*Hyptis suaveolens*	2.23	32.80	6.80	0.14	2.29	4.09	0.22	6.60	0.21	B
25.	*Indigofera constricta*	1.50	0.75	2.00	0.01	0.02	0.09	0.01	0.12	2.67	A
26.	*Lantana camara*	18.6	62.60	29.8	13.40	19.08	7.80	21.23	48.12	0.48	D
27.	*Leea indica*	0.25	22.20	1.15	0.03	0.26	2.77	0.05	3.09	0.05	B
28.	*Lepianthus peltata*	0.53	2.64	20.10	0.05	0.54	0.33	0.08	0.95	7.63	A

29.	*Ludwigia peruviana*	0.10	3.01	3.50	0.01	0.11	0.38	0.01	0.49	1.16	A
30.	*Melostoma malabatrum*	0.09	6.41	1.50	0.06	0.10	0.80	0.09	0.99	0.23	A
31.	*Mesopsis eminii*	0.81	2.70	35.8	0.09	0.83	0.28	0.14	1.26	13.26	A
32.	*Milusa indica*	0.22	0.75	3.00	0.01	0.02	0.09	0.01	0.13	4.00	A
33.	*Ocimum americanum*	0.18	7.92	2.28	0.01	0.19	0.99	0.01	1.19	0.29	A
34.	*Ocimum gratisimum*	0.61	19.60	3.11	0.01	0.63	2.44	0.01	3.08	0.16	A
35.	*Osbekia aspera*	0.08	5.20	1.64	0.01	0.09	0.66	0.01	0.76	0.32	A
36.	*Randia dumetorum*	0.06	5.20	1.14	0.03	0.06	0.66	0.05	0.77	0.22	A
37.	*Rauvolfia serpentina*	0.15	11.30	1.36	0.01	0.16	1.41	0.01	1.58	0.12	A
38.	*Sarcandra chloranthoides*	0.07	6.03	1.18	0.04	0.07	0.75	0.06	0.89	0.20	A
39.	*Senna occidentalis*	0.23	14.30	1.65	0.04	0.24	1.79	0.06	2.09	0.12	A
40.	*Sennn spectabilis*	12.5	86.70	14.5	28.30	12.84	10.81	44.77	68.42	0.17	E
41.	*Sida acuta*	0.32	12.10	2.68	0.05	0.33	1.50	0.08	1.92	0.22	A
42.	*Solanum torvum*	0.63	14.30	4.42	0.05	0.65	1.79	0.08	2.52	0.31	A
43.	*Stachytarpheta cayennensis*	0.53	9.81	5.46	0.09	0.55	1.22	0.14	1.91	0.56	A
44.	*Stachytarpheta jamaicensis*	2.68	19.60	13.7	0.09	2.74	2.44	0.14	5.33	0.70	A
45.	*Synadenium grantii*	0.12	1.10	10.7	0.01	0.12	0.14	0.02	0.29	9.69	A
46.	*Thespesia lampas*	0.26	8.30	3.22	0.02	0.27	1.03	0.04	1.35	0.39	A
47.	*Tithonia diversifolia*	0.11	4.50	2.58	0.02	0.12	0.56	0.03	0.71	0.57	A
48.	*Triumfetta rhomboidea*	0.29	11.70	2.38	0.08	0.29	1.46	0.13	1.87	0.20	A
49.	*Urena lobata*	0.64	20.30	3.16	0.02	0.66	2.54	0.03	3.23	0.16	C
50.	*Urena sinuata*	1.03	44.90	2.30	0.04	1.06	5.59	0.06	6.71	0.05	C
51.	*Ziziphus xylocarpus*	0.06	5.28	1.14	0.01	0.06	0.66	0.01	0.73	0.22	A

Table 4.5 Vegetation composition of herb layer in the Southern range

(D: density, F: frequency; A: abundance; TBA: total basal area; RD: relative density; RF: relative frequency; RDo: relative dominance; IVI: important value index; DBN: distribution; RFC: Raunkiers frequency class)

Sl. No.	Species Name	D	F	A	TBA	RD	RF	RDo	IVI	DBN (A/F)	RFC
1.	*Acanthospermum hispidum*	3.90	28.30	13.80	0.08	0.83	1.51	0.46	2.80	0.49	B
2.	*Achyranthes aspera*	2.35	55.10	4.20	0.06	0.50	2.94	0.38	3.81	0.08	C
3.	*Aeschynomene indica*	1.55	8.67	11.50	0.05	0.33	0.46	0.30	1.10	1.33	A
4.	*Ageratum conyzoides*	63.80	67.20	95.00	3.10	13.54	3.58	18.47	35.59	1.41	D
5.	*Alternanthera betzikiana*	12.90	16.60	65.60	0.08	2.73	1.05	0.48	4.25	3.95	A
6.	*Alternanthera brassiliana*	6.19	6.03	103.0	0.03	1.31	0.32	0.18	1.82	17.02	A
7.	*Amaranthus spinosus*	0.64	10.60	6.10	0.01	0.14	0.56	0.04	0.74	0.58	A
8.	*Anisochilus carnosus*	0.30	11.70	2.61	0.01	0.06	0.62	0.04	0.72	0.22	A
9.	*Argemone mexicana*	0.15	3.39	4.60	0.00	0.03	0.18	0.02	0.23	1.36	A
10.	*Arundinella leptochloa*	11.80	32.10	36.60	0.23	2.49	1.71	1.37	5.57	1.14	B

11.	*Asclepias curassavica*	1.78	47.90	3.71	0.18	0.38	2.94	1.07	4.39	0.08	C
12.	*Axonopus compressus*	52.90	50.50	105.0	1.60	11.23	2.70	9.53	23.46	2.07	C
13.	*Bidens pilosa* var *minor*	2.68	43.30	6.11	0.06	0.57	2.31	0.37	3.25	0.14	C
14.	*Biophytum rainwardii*	1.30	8.60	15.00	0.01	0.28	0.46	0.04	0.78	1.75	A
15.	*Blumea lacera*	1.28	6.03	21.30	0.00	0.27	0.32	0.02	0.62	3.52	A
16.	*Calpogonium mucunoides*	0.12	4.10	2.90	0.02	0.03	0.22	0.09	0.34	0.71	A
17.	*Canscora decusata*	0.23	3.39	6.70	0.01	0.05	0.18	0.03	0.26	1.98	A
18.	*Canscora perviflora*	0.15	19.50	5.10	0.01	0.03	0.16	0.04	0.24	0.26	A
19.	*Carex filicina*	12.10	26.70	45.20	0.31	2.57	1.43	1.85	5.84	1.69	B
20.	*Carya arborea*	0.27	9.40	2.96	0.03	0.06	0.50	0.18	0.75	0.31	A
21.	*Centella asiatica*	2.31	11.60	19.80	0.01	0.49	0.62	0.05	1.17	1.70	A
22.	*Centranthera indica*	0.34	7.90	4.30	0.01	0.07	0.42	0.04	0.53	0.54	A
23.	*Centrosema molle*	1.19	46.70	2.50	0.09	0.25	2.50	0.51	3.26	0.05	C
24.	*Chloris barbata*	1.60	13.20	12.30	0.07	0.35	0.70	0.42	1.47	0.93	A
25.	*Conyza bonariensis*	1.40	17.00	8.24	0.01	0.30	0.91	0.05	1.25	0.49	A
26.	*Conyza canadensis*	0.66	9.40	7.04	0.00	0.14	0.50	0.02	0.66	0.75	A
27.	*Costus speciosus*	0.58	7.10	12.30	0.08	0.12	0.38	0.49	1.00	1.74	A
28.	*Crassocephalum crepidioides*	7.81	15.50	50.50	0.08	1.66	0.83	0.48	2.96	3.27	A
29.	*Croton blonpladianaum*	1.21	9.40	12.80	0.01	0.26	0.50	0.03	0.79	1.37	A
30.	*Curculigo orchioides*	1.17	19.20	6.11	0.01	0.25	1.03	0.04	1.31	0.32	A
31.	*Curcuma olignatha*	0.43	10.10	4.29	0.00	0.09	0.54	0.02	0.66	0.42	A
32.	*Cuscuta reflexa*	1.17	18.10	6.50	0.00	0.25	0.97	0.01	1.23	0.36	A
33.	*Cyclea peltata*	0.27	23.00	1.18	0.01	0.06	1.23	0.05	1.33	0.05	B
34.	*Cymbopogan flexosus*	1.93	11.60	16.50	0.13	0.41	0.62	0.77	1.81	1.42	A
35.	*Cynodon dactylon*	3.14	19.20	17.90	0.03	0.73	1.03	0.18	1.94	0.93	A
36.	*Cynoglossum zeylanicum*	0.26	4.10	6.45	0.01	0.06	0.22	0.05	0.33	1.57	A
37.	*cyperus rotundas*	4.58	24.50	18.70	0.06	0.97	1.31	0.38	2.66	0.76	B
38.	*Cyprus cyperinus*	1.70	19.20	9.01	0.18	0.37	1.03	1.07	2.47	0.47	A
39.	*Dalbergia latifolia*	0.34	33.20	1.03	0.02	0.07	1.77	0.13	1.97	0.03	B
40.	*Desmodium gangeticum*	0.23	15.80	1.45	0.07	0.05	0.85	0.43	1.32	0.09	A
41.	*Desmodium heterocarpon*	0.12	10.60	1.14	0.04	0.03	0.56	0.24	0.83	0.11	A
42.	*Desmodium triflorim*	0.03	2.26	1.33	0.00	0.01	0.12	0.01	0.14	0.59	A
43.	*Digitaria ciliaris*	4.66	23.30	20.00	0.07	0.99	1.25	0.42	2.66	0.86	B
44.	*Elephantopus scaber*	1.56	52.40	2.99	0.10	0.33	2.80	0.57	3.70	0.06	C

45.	*Elueutherantherr rudarallis*	1.17	19.20	1.17	0.06	0.25	1.03	0.36	1.64	0.06	A
46.	*Eregrostis tenella*	6.22	27.10	22.90	0.08	1.32	1.45	0.49	3.26	0.85	B
47.	*euphorbia heterophylla*	0.79	4.90	16.20	0.01	0.17	0.26	0.05	0.48	3.31	A
48.	*Euphorbia hirta*	1.56	7.10	21.80	0.03	0.33	0.38	0.18	0.90	3.07	A
49.	*Globba marantina*	1.50	10.90	14.20	0.08	0.33	0.58	0.49	1.40	1.30	A
50.	*Gomphorena serreta*	1.21	7.10	16.90	0.03	0.26	0.38	0.20	0.84	2.39	A
51.	*Gomphostemma heyneana*	0.28	6.70	2.60	0.05	0.06	0.36	0.30	0.73	0.39	A
52.	*Hedyotis neesiana*	0.14	9.81	1.50	0.01	0.03	0.52	0.04	0.59	0.15	A
53.	*Heliotropium indicum*	0.83	4.10	22.10	0.00	0.18	0.40	0.02	0.60	5.39	A
54.	*Hemidesmus indicus*	0.85	34.30	2.48	0.02	0.18	1.83	0.14	2.16	0.07	B
55.	*Hypoestes sanguinolenta*	5.32	12.10	44.10	0.11	1.13	0.64	0.66	2.43	3.66	A
56.	*Ipomea carnea*	0.42	7.90	5.33	0.04	0.09	0.42	0.21	0.73	0.67	A
57.	*Ipomoea hederifolia*	0.30	18.50	1.65	0.04	0.06	0.99	0.24	1.30	0.09	A
58.	*Ipomoea obscura*	0.40	6.00	6.68	0.02	0.09	0.32	0.13	0.53	1.11	A
59.	*Kyllinga nemoralis*	6.84	20.00	34.20	0.07	1.45	1.07	0.42	2.94	1.71	A
60.	*Leucas biflora*	0.12	6.40	2.00	0.01	0.03	0.34	0.05	0.42	0.31	A
61.	*Leucas urticifolia*	0.13	4.10	3.18	0.01	0.03	0.22	0.05	0.30	0.78	A
62.	*Lindernia crustata*	0.26	4.50	5.91	0.01	0.06	0.24	0.04	0.34	1.31	A
63.	*Merremia vitifolia*	0.51	23.00	2.29	0.01	0.11	1.23	0.04	1.38	0.10	B
64.	*Mikania micrantha*	4.84	13.60	35.60	0.57	1.03	0.72	3.40	5.15	2.62	A
65.	*Mimosa diplotricha*	1.15	12.10	9.56	0.09	0.24	0.64	0.54	1.43	0.79	A
66.	*Mimosa pudica*	72.10	54.70	132.0	2.31	15.29	2.92	13.76	31.97	2.41	C
67.	*Mucuna bracciata*	0.81	4.10	19.50	0.01	0.17	0.22	0.08	0.48	4.77	A
68.	*Naravelia zeylanica*	0.15	13.20	1.83	0.01	0.03	0.70	0.05	0.79	0.14	A
69.	*Oplismenus compositus*	8.72	24.90	35.00	0.81	1.85	1.33	4.82	8.00	1.41	B
70.	*Oxalis corniculata*	3.06	28.70	10.70	0.06	0.65	1.53	0.38	2.56	0.37	B
71.	*Panicum typheron*	0.43	7.90	5.52	0.03	0.09	0.42	0.15	0.67	0.70	A
72.	*Parthenium hysterophorus*	2.73	40.00	6.83	0.09	0.58	2.13	0.54	3.26	0.17	C
73.	*passiflora foetida*	0.77	18.40	4.20	0.01	0.16	0.99	0.07	1.22	0.23	A
74.	*Peninsetum hohenakeri*	1.29	12.10	10.60	0.10	0.27	0.64	0.57	1.49	0.88	A
75.	*Pennisetum pedicellatum*	1.18	5.20	22.50	0.07	0.25	0.28	0.43	0.96	4.33	A
76.	*Pennisetum polystachyon*	0.78	7.10	10.90	0.06	0.17	0.38	0.36	0.91	1.53	A
77.	*Phaulopsis imbricata*	0.23	15.80	1.45	0.01	0.05	0.85	0.05	0.95	0.09	A

78.	*Phyllanthus amarus*	2.32	27.50	8.42	0.07	0.49	1.47	0.43	2.39	0.31	B
79.	*Pimpinella wallichiana*	0.06	3.01	2.00	0.01	0.01	0.16	0.04	0.21	0.66	A
80.	*Plectranthus mollis*	0.23	1.50	15.50	0.04	0.05	0.08	0.25	0.38	10.33	A
81.	*Polygala telephioides*	0.34	10.90	3.19	0.01	0.07	0.58	0.06	0.71	0.29	A
82.	*Pteridium aquilinum*	1.17	2.20	51.80	0.01	0.25	0.12	0.05	0.42	23.55	A
83.	*Rhynchospora corymbosa*	12.20	17.30	70.20	0.16	2.59	0.93	0.95	4.47	4.06	A
84.	*Rorippa indica*	0.31	13.50	2.30	0.01	0.07	0.72	0.04	0.83	0.17	A
85.	*Rungia parviflora*	2.69	43.70	6.10	0.08	0.57	2.33	0.48	3.38	0.14	C
86.	*Sacciolepis indica*	6.10	12.00	50.90	0.07	1.30	0.04	0.40	1.74	4.24	A
87.	*Scleria lithosperma*	0.41	28.60	1.43	0.01	0.09	1.53	0.04	1.65	0.05	B
88.	*Scoparia dulcis*	0.49	7.90	6.28	0.01	0.11	0.42	0.04	0.56	0.79	A
89.	*Senna spectabilus*	49.80	54.70	91.00	0.61	10.57	2.92	3.63	17.12	1.66	C
90.	*Senna tora*	15.20	26.80	56.60	0.11	3.22	1.43	0.66	5.30	2.11	B
91.	*Sida alnifolia*	2.20	29.80	7.39	0.07	0.47	1.59	0.42	2.48	0.25	B
92.	*Sida cordifolia*	1.41	13.60	10.40	0.21	0.30	0.72	1.25	2.27	0.76	A
93.	*Sida rhombifolia*	0.81	4.10	19.50	0.04	0.17	0.22	0.25	0.64	4.77	A
94.	*Sphagneticola trilobata*	0.84	4.52	18.80	0.07	0.18	0.24	0.42	0.84	4.15	A
95.	*Spilanthes ciliata*	1.46	17.30	8.40	0.07	0.31	0.93	0.43	1.67	0.49	A
96.	*Spilanthes radicans*	0.79	13.20	6.00	0.04	0.17	0.70	0.24	1.12	0.45	A
97.	*Themeda triandra*	19.30	27.90	69.10	3.10	4.10	1.49	18.47	24.05	2.48	B
98.	*Tiliacora acuminata*	0.17	7.10	2.42	0.01	0.04	0.38	0.04	0.46	0.34	A
99.	*Tribulus terrestris*	0.41	30.50	1.34	0.01	0.09	1.63	0.04	1.75	0.04	B
100.	*Tridax procumbens*	8.83	64.50	13.70	0.08	1.87	3.44	0.49	5.81	0.21	D
101.	*Tylophora indica*	0.14	7.90	1.80	0.01	0.03	0.42	0.04	0.49	0.23	A
102.	*Vernonia cinerea*	0.53	24.50	2.18	0.01	0.11	1.31	0.04	1.46	0.09	B
103.	*Zingiber nimmonii*	0.19	27.50	1.18	0.01	0.04	0.87	0.05	0.96	0.04	B

Table 4.6 Vegetation composition of Tree layer in the North-West range

(D: density, F: frequency; A: abundance; TBA: total basal area; RD: relative density; RF: relative frequency; RDo: relative dominance; IVI: important value index; DBN: distribution; RFC: Raunkiers frequency class)

Sl No.	Species Name	D	F	A	TBA	RD	RF	RDo	IVI	DBN (A/F)	RFC
1.	*Ailanthes triphysa*	0.17	9.41	1.87	4.03	0.65	1.30	0.98	2.93	0.20	A
2.	*Albizia odorotisima*	0.08	5.88	1.75	0.48	0.30	0.65	0.11	1.06	0.30	A
3.	*Alstonia scholaris*	0.17	12.90	1.36	0.18	0.65	1.80	0.04	2.49	0.11	A
4.	*Terminalia anogeissiana*	0.43	22.10	3.77	12.8	9.50	2.90	3.10	15.50	0.17	B
5.	*Azadirachta indica*	0.10	5.88	1.80	1.25	0.39	0.81	0.30	1.50	0.31	A
6.	*Bauhinia malabarica*	0.24	20.00	1.23	0.06	0.92	2.70	0.01	3.63	0.06	A
7.	*Bauhinia racemosa*	0.04	3.50	1.00	0.04	0.08	0.32	0.01	0.41	0.29	A
8.	*Bridelia retusa*	0.10	8.23	1.28	0.09	0.39	1.14	0.02	1.55	0.16	A
9.	*Bucharia axilaris*	0.05	4.70	1.00	0.86	0.17	0.65	0.21	1.03	0.21	A
10.	*Butea monosperma*	0.21	12.90	1.63	0.53	0.78	1.80	0.44	3.02	0.13	A
11.	*Carayya arborea*	0.05	3.50	1.33	0.67	0.17	0.49	0.16	0.82	0.38	A
12.	*Caseria tomentosa*	0.04	2.35	1.50	0.08	0.13	0.32	0.02	0.47	0.64	A
13.	*Casia fistula*	1.54	37.60	4.09	0.54	5.70	5.20	0.13	11.03	0.11	B
14.	*Cordia obliqa*	0.08	5.80	1.40	0.10	0.30	0.81	0.02	1.13	0.24	A
15.	*Dalbergia lanceolaria*	0.07	4.70	1.50	0.28	0.70	0.65	0.06	1.41	0.32	A
16.	*Dalbergia latifolia*	0.74	44.70	1.65	41.8	2.76	6.22	10.24	19.22	0.04	C
17.	*Dalbergia sissoides*	0.09	7.05	1.33	0.34	0.35	0.98	0.08	1.41	0.19	A
18.	*Dillenia pentagyna*	0.05	3.52	1.33	0.05	0.17	0.49	0.01	0.67	0.38	A
19.	*Dolichandrone arcuata*	0.02	1.17	2.00	0.06	0.08	0.16	0.01	0.25	1.71	A
20.	*Emblica officinalis*	0.54	24.70	2.19	6.12	2.01	3.44	1.49	6.94	0.09	B
21.	*Erythrina indica*	0.07	4.70	1.50	0.94	0.26	0.65	0.23	1.14	0.32	A
22.	*Gmelina arborea*	0.10	8.20	1.28	5.20	0.39	1.14	1.27	2.80	0.16	A
23.	*Grewia tilifolia*	0.49	28.20	1.75	9.43	1.80	3.93	2.31	8.04	0.06	B
24.	*Haldinia cordifolia*	0.22	14.10	1.58	13.4	0.83	1.90	3.28	6.01	0.11	A
25.	*Kydia calcyna*	0.08	6.20	1.45	3.01	0.70	1.80	0.73	3.23	0.23	A
26.	*Lagerstroemia lanceolata*	0.18	12.90	1.45	11.3	0.70	1.80	2.70	5.20	0.11	A
27.	*Naringi crenulata*	0.15	9.41	1.62	0.08	0.57	1.31	0.02	1.90	0.17	A
28.	*Olea dioica*	0.22	15.30	1.46	0.49	0.83	2.10	0.12	3.05	0.10	A
29.	*Pavetta indica*	0.03	2.06	1.66	0.03	0.21	0.49	0.01	0.71	0.81	A
30.	*Pterocarpus marsupium*	0.67	45.90	1.46	42.9	2.50	6.39	10.51	19.40	0.03	C
31.	*Radermarchea xylocarpa*	0.05	3.52	1.66	0.32	0.21	0.49	0.08	0.78	0.47	A
32.	*Randia gardeneri*	0.05	3.50	1.66	0.97	0.21	0.49	1.08	1.78	0.47	A
33.	*Salix tetrasperma*	0.05	3.50	1.33	0.42	0.17	0.49	0.10	0.76	0.38	A

34.	*Schleichera oleosa*	0.36	12.90	2.81	12.10	1.36	1.80	2.94	6.10	0.22	A
35.	*Senna siamea*	0.30	17.60	1.73	0.12	1.14	2.45	0.03	3.62	0.10	A
36.	*Senna spectabilis*	7.27	94.10	7.72	10.10	27.11	13.11	2.47	42.69	0.08	E
37.	*Stereospermum chelonoides*	0.16	7.05	2.33	0.04	0.61	0.98	0.01	1.60	0.33	A
38.	*Syzigium cumini*	0.42	20.00	2.11	5.53	1.50	2.70	1.35	5.55	0.11	A
39.	*Tamilnadia uliginosa*	0.15	5.80	2.60	0.08	0.57	0.81	0.02	1.40	0.45	A
40.	*Tectona grandis*	6.35	60.00	6.35	147.00	23.69	8.36	35.90	67.95	0.11	C
41.	*Terminalia bellerica*	0.21	10.60	2.00	6.48	0.78	1.14	15.80	17.72	0.19	A
42.	*Terminalia eliptica*	1.01	56.00	1.79	36.50	3.77	7.80	8.82	20.39	0.03	C
43.	*Terminalia paniculata*	0.14	0.10	0.63	9.70	2.10	1.40	2.37	5.87	6.30	A
44.	*Trema orientalis*	0.07	4.70	1.50	0.04	0.26	0.65	0.01	0.92	0.32	A
45.	*Trivia nudiflora*	0.07	4.70	1.50	0.08	0.26	0.65	0.02	0.93	0.32	A
46.	*Wrighiea tinctoria*	0.12	5.88	2.20	0.97	0.48	0.81	0.23	1.52	0.37	A
47.	*Ziziphus mauritiana*	0.10	5.80	1.80	0.07	0.39	0.81	0.02	1.22	0.31	A

Table 4.7 Vegetation composition of Shrub layer in North-West Ranges

(D: density, F: frequency; A: abundance; TBA: total basal area; RD: relative density; RF: relative frequency; RDo: relative dominance; IVI: important value index; DBN: distribution; RFC: Raunkiers frequency class)

Sl No.	Species Name	D	F	A	TBA	RD	RF	RDo	IVI	DBN (A/F)	RFC
1.	*Ageratina adenophora*	0.22	4.70	44.80	0.05	1.74	0.65	0.21	2.60	9.52	A
2.	*Barleria cristata*	0.17	3.50	5.10	0.05	0.50	1.14	0.22	1.86	1.46	A
3.	*Boehmeria glomerulifera*	0.09	5.88	1.60	0.01	0.08	0.82	0.04	0.93	0.27	A
4.	*Calotropis gigantea*	0.24	4.70	5.25	0.03	0.20	2.12	0.16	2.49	1.12	A
5.	*Capparis brevispina*	0.10	4.70	2.20	0.02	0.16	0.65	0.11	0.92	0.47	A
6.	*Catunaregam spinosa*	0.32	12.9	1.90	0.08	0.20	1.80	0.37	2.38	0.15	A
7.	*Chassalia curviflora*	0.10	8.23	1.20	0.01	0.09	1.14	0.04	1.27	0.15	A
8.	*Chromolaena odorata*	36.6	95.3	38.40	2.89	30.30	13.24	13.32	56.86	0.40	E
9.	*Caesalpinia mimosoides*	0.43	8.30	5.27	0.07	1.13	3.59	0.33	5.05	0.63	A
10.	*Colebrookea oppositifolia*	3.67	21.2	17.30	0.21	3.04	2.94	0.97	6.95	0.82	B
11.	*Crotalaria spectabilis*	0.16	5.00	2.80	0.07	0.69	1.80	0.34	2.83	0.56	A
12.	*Datura innoxia*	0.16	5.00	2.80	0.04	0.14	0.82	0.19	1.15	0.56	A
13.	*Decaschita crotoniflora*	0.05	3.50	1.66	0.03	0.05	0.49	0.14	0.68	0.47	A
14.	*Flemingia grahamiana*	0.36	25.9	1.40	0.06	0.30	3.59	0.27	4.16	0.05	B
15.	*Glycosmis*	3.60	28.2	13.00	0.01	3.04	3.92	0.06	7.02	0.46	B

	pentaphylla										
16.	*Grewia hirsuta*	0.10	5.88	1.80	0.01	0.09	0.82	0.06	0.97	0.31	A
17.	*Helicteres isora*	1.40	14.1	9.91	0.33	1.16	1.96	1.52	4.64	0.70	A
18.	*Helixanthera obtusata*	0.16	10.6	1.550	0.06	0.14	1.47	0.28	1.89	0.15	A
19.	*Hibiscus furcatus*	0.17	3.50	5.00	0.02	0.15	0.49	0.07	0.71	1.43	A
20.	*Hibiscus lobatus*	0.12	5.80	2.20	0.03	0.11	0.82	0.14	1.07	0.38	A
21.	*Hyptis capitata*	0.08	4.70	1.75	0.03	0.07	0.65	0.16	0.88	0.37	A
22.	*Hyptis suaveolens*	3.48	10.5	32.9	0.14	2.88	1.47	0.65	5.00	3.13	A
23.	*Lantana camara*	28.4	91.8	30.9	3.78	23.5	12.75	17.43	53.67	0.34	E
24.	*Leea indica*	0.15	8.20	2.28	0.02	0.16	1.14	0.09	1.39	0.28	A
25.	*Ludwigia peruviana*	0.16	4.70	3.50	0.03	0.14	0.65	0.14	0.93	0.74	A
26.	*Miliusa indica*	0.02	1.17	2.00	0.01	0.02	0.16	0.02	0.21	1.71	A
27.	*Ocimum americanum*	0.40	8.20	4.80	0.01	0.33	1.14	0.03	1.51	0.59	A
28.	*Ocimum gratisimum*	7.20	57.6	12.5	0.75	5.96	8.01	3.46	17.42	0.22	C
29.	*Osbekia aspera*	0.09	3.50	1.60	0.02	0.13	0.65	0.11	0.89	0.46	A
30.	*Pseudarthria viscida*	0.20	7.00	2.83	0.02	0.17	0.98	0.08	1.22	0.40	A
31.	*Randia dumetorum*	0.09	3.50	2.66	0.01	0.08	0.49	0.06	0.62	0.76	A
32.	*Sarcandra chloranthoides*	0.08	5.80	1.60	0.03	0.07	0.82	0.14	1.02	0.28	A
33.	*Senna occidentalis*	0.24	22.4	1.10	0.06	0.20	3.10	0.28	3.59	0.05	B
34.	*Sennn spectabilis*	13.6	80.0	20.80	9.50	13.78	11.11	43.80	68.69	0.26	D
35.	*Sida acuta*	0.76	20.0	3.80	0.03	0.63	2.78	0.15	3.56	0.19	A
36.	*Solanum torvum*	0.72	9.40	7.75	0.03	0.60	1.31	0.14	2.05	0.82	A
37.	*Stachytarpheta cayennensis*	1.31	11.8	11.00	0.04	1.09	1.63	0.19	2.91	0.94	A
38.	*Stachytarpheta jamaicensis*	7.32	22.4	32.80	0.08	6.07	3.10	0.37	9.54	1.47	B
39.	*Thespesia lampas*	0.34	9.40	3.22	0.01	0.28	1.31	0.06	1.65	0.34	A
40.	*Triumfetta rhomboidea*	0.10	4.70	2.25	0.02	0.50	2.94	0.11	3.55	0.48	A
41.	*Urena lobata*	0.63	5.88	10.8	0.01	0.53	0.82	0.06	1.41	1.84	A
42.	*Urena sinuata*	0.71	10.6	6.77	0.02	0.59	1.47	0.11	2.17	0.64	A
43.	*Ziziphus xylocarpus*	0.09	5.80	1.60	0.01	0.08	0.82	0.04	0.94	0.28	A

Table 4.8 Vegetation composition of herb layer in North-West Ranges

(D: density, F: frequency; A: abundance; TBA: total basal area; RD: relative density; RF: relative frequency; RDo: relative dominance; IVI: important value index; DBN: distribution; RFC: Raunkiers frequency class)

Sl No.	Species Name	D	F	A	TBA	RD	RF	RDo	IVI	DBN (A/F)	RFC
1.	*Acanthospermum hispidum*	4.80	18.80	25.90	0.06	0.92	0.86	0.93	2.71	1.38	A
2.	*Achyranthes aspera*	2.51	15.20	16.50	0.05	0.47	0.70	0.74	1.91	1.08	A
3.	*Aeschynomene indica*	2.49	8.23	30.30	0.04	0.47	0.38	0.58	1.43	3.68	A
4.	*Ageratum conyzoides*	24.9	64.70	38.40	0.59	4.68	2.95	8.58	16.21	0.59	D
5.	*Amaranthus spinosus*	1.81	22.40	8.10	0.01	0.34	1.02	0.13	1.49	0.36	B
6.	*Anisochilus carnosus*	0.48	9.40	5.12	0.01	0.09	0.43	0.07	0.59	0.54	A
7.	*Argemon mexicana*	0.36	4.70	7.75	0.01	0.07	0.21	0.08	0.36	1.65	A
8.	*Arundinella leptochloa*	13.80	36.50	37.90	0.14	2.60	1.66	2.04	6.30	1.04	B
9.	*Asclepias curassavica*	2.83	25.80	11.00	0.04	0.53	1.18	0.60	2.31	0.42	B
10.	*Axonopus compressus*	73.10	60.00	122.00	0.80	13.75	2.74	11.63	28.12	2.03	C
11.	*Bidens pilosa* var. *minor*	3.71	36.4	10.20	0.07	0.70	1.66	1.03	3.40	0.28	B
12.	*Biophytum rainwardi*	2.04	11.8	17.40	0.01	0.39	0.54	0.12	1.04	1.48	A
13.	*Blumea axillaris*	1.54	10.6	14.60	0.05	0.29	0.48	0.74	1.51	1.38	A
14.	*Blumea lacera*	22.50	20.00	113.00	0.12	4.24	0.91	1.75	6.90	5.63	A
15.	*Canna indica*	0.89	2.35	38.00	0.04	0.17	0.11	0.58	0.86	16.17	A
16.	*Canscora decusata*	0.24	3.50	7.00	0.00	0.05	0.16	0.06	0.27	2.00	A
17.	*Canscora perviflora*	0.30	3.50	8.60	0.01	0.06	0.16	0.07	0.29	2.46	A
18.	*Carex filicina*	22.70	42.40	53.60	0.86	4.27	1.93	12.51	18.71	1.27	C
19.	*Centella asiatica*	3.69	18.80	19.60	0.01	0.69	0.86	0.09	1.64	1.04	A
20.	*Centranthera indica*	0.60	10.60	5.60	0.05	0.11	0.48	0.73	1.32	0.53	A
21.	*Centrosema molle*	3.34	72.90	4.50	0.03	0.63	3.33	0.44	4.39	0.06	D
22.	*Chloris barbata*	2.57	16.40	15.60	0.06	0.48	0.75	0.87	2.11	0.95	A
23.	*Conyza bonariensis*	2.76	34.10	8.10	0.04	0.52	1.56	0.58	2.66	0.24	B
24.	*Conyza canadensis*	0.95	12.90	7.36	0.04	0.18	0.59	0.58	1.35	0.57	A
25.	*Costus speciosus*	0.87	7.00	12.30	0.07	0.16	0.32	1.03	1.52	1.76	A
26.	*Crassocephalum crepidioides*	7.22	29.40	24.60	0.07	1.36	1.34	1.03	3.73	0.84	B
27.	*Curculigo orchioides*	1.34	7.05	19.00	0.01	0.25	0.32	0.13	0.71	2.70	A
28.	*Curcuma ecalcarata*	0.61	3.50	17.30	0.02	0.12	0.16	0.31	0.58	4.95	A
29.	*Curcuma olignatha*	0.72	12.90	5.63	0.01	0.14	0.59	0.13	0.86	0.44	A
30.	*Cuscuta reflexa*	2.28	36.40	6.25	0.00	0.43	1.66	0.02	2.11	0.17	B
31.	*Cyclea peltata*	0.65	36.40	1.80	0.01	0.12	1.66	0.12	1.91	0.05	B
32.	*Cynoglossum zeylanicum*	0.49	4.70	10.50	0.03	0.09	0.21	0.44	0.74	2.23	A
33.	*Cyperus rotundus*	7.17	30.60	23.50	0.05	1.35	1.40	0.79	3.53	0.77	B
34.	*Cyperus cyperinus*	4.37	16.50	26.60	0.07	0.82	0.75	1.08	2.65	1.61	A
35.	*Desmodium*	0.49	18.80	2.62	0.04	0.09	0.86	0.58	1.53	0.14	A

	gangeticum										
36.	*Desmodium heterocarpon*	0.24	12.90	1.90	0.04	0.05	0.59	0.55	1.19	0.15	A
37.	*Desmodium triflorim*	0.07	4.70	1.50	0.01	0.01	0.21	0.12	0.34	0.32	A
38.	*Digitaria ciliaris*	6.04	30.50	19.8	0.07	1.14	1.40	0.95	3.48	0.65	B
39.	*Elephantopus scaber*	3.08	76.50	4.03	0.05	0.58	3.49	0.79	4.86	0.05	D
40.	*Eregrostis tenella*	10.8	22.40	51.10	0.09	2.15	1.02	1.31	4.48	2.29	B
41.	*Globba marantina*	4.29	25.90	16.60	0.07	0.81	1.18	1.06	3.05	0.64	B
42.	*Gomphostemma heyneana*	0.56	10.60	5.33	0.04	0.11	0.48	0.60	1.19	0.5	A
43.	*Hedyotis neesiana*	0.28	20.00	1.41	0.02	0.05	0.91	0.32	1.29	0.07	A
44.	*Heliotropium indicum*	6.00	7.05	85.00	0.01	1.13	0.32	0.10	1.55	12.06	A
45.	*Hemidesmus indicus*	1.71	61.10	2.80	0.01	0.32	2.79	0.19	3.30	0.05	D
46.	*Hypoestes sanguinolenta*	6.36	12.90	49.20	0.08	1.20	0.59	1.16	2.95	3.80	A
47.	*Ipomoea hederifolia*	0.75	44.70	1.68	0.01	0.14	2.04	0.19	2.37	0.04	C
48.	*Kyllinga nemoralis*	8.37	24.70	33.90	0.06	1.58	1.13	0.87	3.58	1.37	B
49.	*Leucas biflora*	0.16	12.90	1.27	0.01	0.03	0.59	0.13	0.75	0.10	A
50.	*Leucas urticifolia*	0.24	9.41	2.62	0.01	0.05	0.43	0.09	0.56	0.28	A
51.	*Merremia vitifolia*	1.07	57.60	1.85	0.03	0.30	2.63	0.44	3.37	0.03	C
52.	*Mikania micrantha*	2.25	20.00	11.30	0.03	0.42	0.91	0.44	1.77	0.56	A
53.	*Mimosa diplotricha*	2.09	12.90	16.20	0.08	0.39	0.59	1.16	2.15	1.25	A
54.	*Mimosa pudica*	56.50	71.80	78.70	0.51	10.62	3.28	7.42	21.31	1.10	D
55.	*Naravelia zeylanica*	0.12	7.05	1.83	0.01	0.02	0.32	0.07	0.42	0.26	A
56.	*Oplismenus compositus*	20.00	44.70	44.70	0.13	3.76	2.04	1.89	7.69	1.00	C
57.	*Oxalis corniculata*	1.42	58.80	2.42	0.03	0.27	2.69	0.44	3.39	0.04	C
58.	*Parthenium hysterophorus*	3.89	16.50	23.60	0.07	0.73	0.75	1.02	2.50	1.44	A
59.	*Peninsetum hohencakeri*	1.65	18.80	8.81	0.15	0.31	0.86	2.18	3.35	0.47	A
60.	*Pennisetum pedicellatum*	2.51	10.60	23.70	0.09	0.47	0.48	1.31	2.27	2.24	A
61.	*Pennisetum polystachyon*	1.89	8.23	23.00	0.08	0.36	0.38	1.18	1.91	2.79	A
62.	*Persicaria glabra*	0.63	2.30	27.00	0.06	0.12	0.11	0.87	1.10	11.74	A
63.	*Phaulopsis imbricata*	0.37	16.50	2.28	0.01	0.07	0.75	0.12	0.94	0.14	A
64.	*Phyllanthus amarus*	3.09	48.20	6.41	0.06	0.58	2.20	0.87	3.66	0.13	C
65.	*Pimpinella wallichiana*	0.07	4.70	1.50	0.01	0.01	0.21	0.09	0.32	0.32	A
66.	*Plectranthus mollis*	1.07	9.41	11.40	0.04	0.20	0.43	0.61	1.24	1.21	A
67.	*Polygala telephioides*	0.65	24.70	2.66	0.01	0.12	1.13	0.13	1.38	0.11	B
68.	*Rhynchospora corymbosa*	14.30	25.80	55.20	0.14	2.69	1.18	2.04	5.90	2.14	B
69.	*Rorippa indica*	0.71	40.00	1.79	0.01	0.13	1.83	0.12	2.08	0.04	B
70.	*Rungia parviflora*	3.60	60.00	6.09	0.07	0.69	2.74	1.02	4.45	0.10	C
71.	*Scleria lithosperma*	0.87	36.50	2.38	0.01	0.16	1.66	0.12	1.94	0.07	B
72.	*Scoparia dulcis*	1.07	16.40	6.50	0.05	0.20	0.75	0.76	1.71	0.40	A

73.	*Senna spectabilis*	50.60	82.00	61.40	0.53	9.51	3.76	7.71	20.98	0.75	E
74.	*Senna tora*	8.71	28.20	29.60	0.07	1.57	1.29	0.95	3.81	1.05	B
75.	*Sida alnifolia*	7.39	54.10	7.39	0.05	0.75	2.47	0.76	3.98	0.14	C
76.	*Sida cordifolia*	1.89	14.10	13.40	0.08	0.36	0.64	1.18	2.18	0.95	A
77.	*Sida rhombifolia*	1.27	7.05	18.00	0.03	0.24	0.32	0.45	1.01	2.55	A
78.	*Sphagneticola trilobata*	1.42	4.70	30.30	0.06	0.27	0.21	0.87	1.36	6.44	A
79.	*Spilanthes ciliata*	2.10	9.41	23.50	0.06	0.41	0.43	0.92	1.76	2.50	A
80.	*Spilanthes radicans*	1.77	7.05	25.20	0.03	0.33	0.32	0.45	1.11	3.57	A
81.	*Themeda triandra*	15.30	24.70	61.90	0.16	2.88	1.13	2.33	6.33	2.51	B
82.	*Tiliacora acuminata*	0.24	16.50	1.50	0.01	0.05	0.75	0.09	0.89	0.09	A
83.	*Tribulus terrestris*	0.83	49.40	1.69	0.02	0.16	2.26	0.29	2.70	0.03	C
84.	*Tridax procumbens*	3.77	18.80	20.10	0.07	0.71	0.86	1.03	2.60	1.07	A
85.	*Tylophora indica*	0.16	10.60	1.50	0.01	0.03	0.48	0.07	0.59	0.14	A
86.	*Vernonia cinerea*	1.12	48.20	2.34	0.01	0.21	3.81	0.09	4.11	0.05	C
87.	*Zingiber nimmonii*	0.24	14.10	1.75	0.01	0.05	0.64	0.12	0.81	0.12	A

Table 4.9 Density, Abundance & Frequency of Various growth stages of *Senna spectabilis* in North-West ranges

Sl. No	**Range/ Location**	**Habitat**	**Trees**			**saplings**			**seedlings**		
			Density /ha	**Abundance**	**Frequency**	**Density/ha**	**Abundance**	**Frequency**	**Density/ha**	**Abundance**	**Frequency**
I	**Tholpetty**										
1	Kattapallam	MDF	447.5	23.7	0.8	550	31.42	0.7	7375	491.6	0.6
2	Kaimaram	MDF	470.6	22	0.6	230	18	0.6	4260	221.6	0.7
3	Narimanthikolli	MDF	87.5	35	1	155	7.75	0.8	5250	300	0.7
4	Doddady	MDF	65	3.71	0.7	230	11.5	0.8	2425	121.25	0.8
5	Ayyapanpara	MDF	150	10.28	0.7	277.5	15.85	0.7	4405	251.71	0.7
6	Dasanghatta	MDF	120	4.8	1	355	14.2	1	3100	124	1
7	Tholpetty F.S	MDF	470	18.8	1	530	23.5	0.9	7230	361.5	0.8
8	Thirunelly-thetroad	MDF	205	8.2	1	1170	6.8	1	560	28	0.8
9	Thirulkunnu	MDF	15	1.2	0.5	55	3.66	0.6	155	12.3	0.6
10	Undichiravayal	MDF	12.5	1.25	0.4	0	0	0	0	0	0
11	Bavali	MDF	105	4.2	1	60	2.4	1	560	22.4	1

Table 4.10 Density, Abundance & Frequency of Various growth stages of *Senna spectabilis* in Southern ranges

Sl. No	Range/ Location	Habitat	Trees			saplings			seedlings		
			Density/ ha	Abundance	Frequency	Density/ ha	Abunda-nce	Frequency	Density/ha	Abundance	Frequency
II	**Muthanga**										
1	Kakkapadam	MDF	780	31.2	1	1537.5	61.5	1	9500	380	1
2	Maragadha	MDF	350	14	1	475	23.75	0.5	3640	145.6	1
3	Mudumalakallu	MDF	475	23.75	0.8	300	12	1	777.5	31.1	1
4	Cheeranadakolly	MDF	77.5	3.44	0.9	100	4	1	875	35	1
5	Ambukuthy vayal	MDF	120	4.8	1	355	14.2	1	3100	124	1
6	Pilachikavu (Thotamoola)	MDF	470	18.8	1	530	23.5	0.9	7230	361.5	0.8
7	Thakarappady	MDF	525	21	1	1030	412	1	7125	285	1
8	Vattavayal	MDF	280	12.44	0.9	550	24.44	0.9	4000	177.77	0.9
III	**Sulthan Bathery**										
1	Mangamukku	MDF	340	13.6	1	560	22.4	1	8500	340	1
2	Rampalli	DDF	155	6.2	1	580	23.2	1	5500	220	1
3	State border	MDF	470	18.8	1	900	36	1	9900	396	1
4	Ponkuzhi	MDF	402.5	20.12	0.8	930	37.2	1	2225	84	1
iv	**Kurichyad**										
1	Pachady	MDF	475	19	1	410	19	1	4300	130	1
2	Veetikutti	MDF	470	18.8	1	520	14	1	5200	240	1
3	Thathoor	MDF	410	16.4	1	645	21	1	6125	290	1

4.1.3. Species distribution model; the potential habitat suitability of *S. spectabilis* in Wayanad

Climate change has exacerbated the threat of biological invasions, mainly because its range of climatically suitable regions for invasive alien species has increased. Many native and invasive species' distributions were anticipated to change as the future climate changes. Species distribution models may be instrumental in risk analysis of recently arrived, dangerous invasive species. Preventive management is required in areas where invasion is a danger. In this study, the potential habitat suitability of invasive species *S. spectabilis* was modelled using species distribution modelling under current and future climate conditions in Wayanad. Projected climate change is expected to significantly influence the spread and distribution of the *S. spectabilis* in Wayanad District, Kerala.

4.1.3.1. Model accuracy: The model output obtained was then assessed. Area Under Curve (AUC), True Skill Statistics (TSS), Sensitivity and Specificity were used for

measuring the model performance of the current potential distribution in the study area and are shown in Table 4.11. below.

Table 4.11 Model Performance of current potential distribution of *S. spectabilis* in Wayanad using independent and dependent thresholds (AUC, TSS)

ACCURACY METRICS	VALUES
Training AUC	0.96
Test AUC	0.94
TSS value	0.83
AUC Standard Deviation	0.02
Overall accuracy	0.96
Sensitivity	0.86
Specificity	0.96

The accuracy metrics in Table 4.11 showed that the MaxEnt model has a good performance with a test AUC value of 0.94 and a TSS value of 0.83. Furthermore, an overall accuracy of 0.96 was shown, similar to the specificity value. The model's sensitivity is 0.86 and showed a standard deviation of 0.02. These values explained that the model has a good fit.

4.1.3.2. Variable optimization: The statistical analysis using a multicollinearity test was conducted using Pearson correlation coefficient in ArcGIS ver.10.7.1 ESRI using SDM toolbox maintained 15 variables for modelling the potential habitat suitability of *S. spectabilis* invasive alien species. The Pearson correlation coefficient between variables is given in Table 4.12, and the highlighted ones are the highly correlated variables ($r > 0.7$) and were excluded from the model to avoid the effect of multi-collinearity thereby improving the accuracy of the model by reducing the masking effect and over prediction of the model.

The selected environmental variables for the study were Annual mean temperature (BIO1), Isothermality (BIO3), Temperature Seasonality (BIO4), Precipitation Seasonality (BIO15), Precipitation of Driest Quarter (BIO17), Precipitation of Warmest Quarter (BIO18), Aspect, Slope, Distance from water bodies, Distance from Road, Landcover, NDVI (Normalised Difference Vegetation Index), Soil type, Population Density were used as inputs in the study. Only bioclimatic variables correlated with each other compared to the non-climatic variables. The variables that had the most significant number of correlations between

other variables were BIO1 (Annual mean Temperature), BIO6 (Min Temperature of Coldest Month), BIO7 (Temperature Annual Range), BIO8 (Mean Temperature of Wettest Quarter), BIO9 (Mean Temperature of Driest Quarter), BIO11 (Mean Temperature of Coldest Quarter), BIO15 (Precipitation Seasonality) (six correlations under |r|). In |r|>0.7 criteria variables BIO3 (Isothermality) and BIO18 (Precipitation of Warmest Quarter), both the variables were chosen due to their essential contribution on the distribution of *S. spectabilis*.

Precipitation of Driest Quarter (BIO17) correlates with Precipitation of Driest Month (BIO14), from which Precipitation of Driest Quarter was selected. Precipitation Seasonality (BIO15) was selected among the correlated variables of Annual precipitation (BIO12), Precipitation of Wettest Quarter (BIO16), Precipitation of Wettest Month (BIO13), and Precipitation of Coldest Quarter (BIO19). Max Temperature of the Warmest Month (BIO5) was excluded, and Temperature Seasonality (BIO4) was chosen among the collinear variables. Among the bioclimatic variables, BIO1 (Annual mean temperature), BIO3 (Isothermality), BIO4 (Temperature Seasonality), BIO15 (Precipitation Seasonality), BIO17 (Precipitation of Driest Quarter), BIO18 (Precipitation of Warmest Quarter) were selected and all the non-correlated non-climatic variables were used as inputs in the model for the distribution of *S. spectabilis.*

4.1.3.3. Important environmental variables: Out of the 15 chosen environmental variables, the significant variable affecting the spatial distribution of S. spectabilis was isothermality (37.4% of variation) followed by elevation (20.8%), annual mean temperature (7.8%), slope (6.4%) and land cover (6%). The cumulative contribution of these variables was 78.4%. Aspect, soil type, NDVI and population density were negligible contributors to the model building. The temperature variables (BIO3, BIO1) contributed more than the precipitation variables (BIO18, BIO15, BIO17) in the distribution of S. spectabilis.

Table 4.12 Cross-correlations (Pearson correlation coefficient, r) among 28 bioclimatic and topographic variables. Correlation values depicted with bold highlighted text indicate high correlation.

Layer	BIO 1	BIO 2	BIO 3	BIO 4	BIO 5	BIO 6	BIO 7	BIO 8	BIO 9	BIO 10	BIO 11	BIO 12	BIO 13	BIO 14	BIO 15	BIO 16	BIO 17	BIO 18	BIO 19	elev	aspect	slope	soil	NDVI	landcover	water	road	population
BIO1	1	-0.474	0.442	-0.01	-0.534	0.984	0.986	0.9616	0.993	0.994	0.99584	0.2203	0.1026	-0.3699	0.128	0.1642	-0.2207	0.55373	0.3779	-0.9896	0.0519	-0.089	0.1848	-0.296	-0.2045	0.4471	-0.376	-0.1941
BIO2	-0.474	1	-0.489	0.535	0.9105	-0.52	-0.428	-0.26023	-0.5504	-0.409	-0.5037	-0.027	0.17165	-0.1667	0.2453	0.1017	-0.2679	-0.6215	-0.0889	0.48629	-0.0223	0.1486	-0.323	0.1386	0.0564	-0.5457	0.0067	-0.0429
BIO3	0.4419	-0.489	1	-0.77	-0.804	0.548	0.353	0.23707	0.4921	0.351	0.49241	-0.261	-0.5052	0.44281	-0.63	-0.439	0.6066	0.74553	-0.2663	-0.4055	0.0362	-0.035	0.1053	-0.065	0.0902	0.22505	-0.128	0.0139
BIO4	-0.014	0.5353	-0.774	1	0.7204	-0.16	0.102	0.23982	-0.0662	0.095	-0.0877	0.2837	0.50159	-0.714	0.6991	0.458	-0.781	-0.5673	0.4899	0.007	0.0013	0.0308	-0.126	-0.102	-0.2429	-0.0983	-0.091	-0.1082
BIO5	-0.534	0.9105	-0.804	0.72	1	-0.61	-0.462	-0.29247	-0.61	-0.447	-0.576	0.0983	0.3499	-0.3214	0.4605	0.2699	-0.4663	-0.7712	0.0503	0.52373	-0.0344	0.1162	-0.266	0.1247	-0.004	-0.473	0.0588	-0.0345
BIO6	0.9837	-0.52	0.548	-0.16	-0.613	1	0.95	0.9117	0.9808	0.961	0.9889	0.112	-0.0238	-0.2228	-0.024	0.0378	-0.0746	0.59581	0.2545	-0.9685	0.0521	-0.089	0.1873	-0.272	-0.1545	0.43607	-0.352	-0.1944
BIO7	0.9864	-0.428	0.353	0.102	-0.462	0.95	1	0.97643	0.9788	0.993	0.97376	0.259	0.17218	-0.4703	0.2264	0.2268	-0.3309	0.47076	0.4521	-0.9782	0.0559	-0.087	0.1702	-0.31	-0.2454	0.46372	-0.382	-0.1861
BIO8	0.9616	-0.26	0.237	0.24	-0.292	0.912	0.976	1	0.9361	0.985	0.94179	0.2728	0.22346	-0.5411	0.2998	0.267	-0.4106	0.37283	0.459	-0.9519	0.0489	-0.067	0.1236	-0.306	-0.2502	0.36977	-0.397	-0.2214
BIO9	0.993	-0.55	0.492	-0.07	-0.61	0.981	0.979	0.93614	1	0.98	0.99217	0.1899	0.0565	-0.3302	0.079	0.122	-0.1688	0.59251	0.3618	-0.9813	0.0532	-0.098	0.2004	-0.3	-0.2059	0.4803	-0.351	-0.1708
BIO10	0.9937	-0.409	0.351	0.095	-0.447	0.961	0.993	0.98466	0.9803	1	0.98251	0.2589	0.1671	-0.4539	0.2145	0.2228	-0.3124	0.48771	0.4303	-0.9852	0.051	-0.085	0.1693	-0.304	-0.2291	0.43282	-0.387	-0.205
BIO11	0.9958	-0.504	0.492	-0.09	-0.576	0.989	0.974	0.94179	0.9922	0.983	1	0.2097	0.07872	-0.3322	0.0925	0.1413	-0.1738	0.59638	0.336	-0.9867	0.0493	-0.091	0.1935	-0.285	-0.1864	0.45035	-0.372	-0.1848
BIO12	0.2203	-0.027	-0.261	0.284	0.0983	0.112	0.259	0.27277	0.1899	0.259	0.20972	1	0.94149	-0.5916	0.8008	0.9722	-0.5836	0.22254	0.7714	-0.268	-0.0206	-0.002	0.0432	-0.006	-0.0725	0.11741	-0.121	0.0512
BIO13	0.1026	0.1717	-0.505	0.502	0.3499	-0.02	0.172	0.22346	0.0565	0.167	0.07872	0.9415	1	-0.6957	0.9223	0.9903	-0.7407	-0.0868	0.7674	-0.1538	-0.016	0.0277	-0.054	0.0412	-0.0618	0.03608	-0.093	0.0235
BIO14	-0.37	-0.167	0.443	-0.71	-0.321	-0.22	-0.47	-0.54108	-0.3302	-0.454	-0.3322	-0.592	-0.6957	1	-0.872	-0.694	0.9577	0.12557	-0.6678	0.40555	-0.0151	0.0326	-0.03	0.2094	0.3323	-0.2579	0.3168	0.0826
BIO15	0.128	0.2453	-0.63	0.699	0.4605	-0.02	0.226	0.29975	0.079	0.215	0.09245	0.8008	0.92233	-0.8719	1	0.9054	-0.9119	-0.2442	0.7241	-0.179	-0.0089	0.0072	-0.038	-0.047	-0.1907	0.08587	-0.166	-0.0189
BIO16	0.1642	0.1017	-0.439	0.458	0.2699	0.038	0.227	0.26697	0.122	0.223	0.14131	0.9722	0.99034	-0.6945	0.9054	1	-0.7224	0.00906	0.7932	-0.2146	-0.0143	0.0118	-0.013	0.002	-0.0945	0.07719	-0.117	0.0284
BIO17	-0.221	-0.268	0.607	-0.78	-0.466	-0.07	-0.331	-0.41058	-0.1688	-0.312	-0.1738	-0.584	-0.7407	0.95769	-0.912	-0.722	1	0.32825	-0.6651	0.25975	-0.0191	0.0201	0.0071	0.1572	0.31	-0.2039	0.2704	0.0734
BIO18	0.5537	-0.621	0.746	-0.57	-0.771	0.596	0.471	0.37283	0.5925	0.488	0.59638	0.2225	-0.0868	0.12557	-0.244	0.0091	0.3283	1	0.092	-0.5468	-0.0113	-0.096	0.2962	-0.162	-0.0411	0.34497	-0.194	0.0106
BIO19	0.3779	-0.089	-0.266	0.49	0.0503	0.255	0.452	0.45898	0.3618	0.43	0.33601	0.7714	0.76743	-0.6678	0.7241	0.7932	-0.6651	0.09203	1	-0.4022	0.021	-0.029	0.0599	-0.112	-0.193	0.23935	-0.139	-0.0059
elev	-0.99	0.4863	-0.406	0.007	0.5237	-0.97	-0.978	-0.95185	-0.9813	-0.985	-0.9867	-0.268	-0.1538	0.40555	-0.179	-0.215	0.2598	-0.5468	-0.4022	1	-0.0441	0.091	-0.189	0.281	0.1916	-0.4536	0.3726	0.1972
aspect	0.0519	-0.022	0.036	0.001	-0.034	0.052	0.056	0.0489	0.0532	0.051	0.04932	-0.021	-0.016	-0.0151	-0.009	-0.014	-0.0191	-0.0113	0.021	-0.0441	1	0.2107	-0.061	-0.005	0.0073	0.02017	0.0033	-0.0188
slope	-0.089	0.1486	-0.035	0.031	0.1162	-0.09	-0.087	-0.06685	-0.0981	-0.085	-0.091	-0.002	0.02771	0.03255	0.0072	0.0118	0.0201	-0.0963	-0.0285	0.09104	0.2107	1	-0.249	0.1222	0.1524	-0.1651	0.0585	-0.0157
soil	0.1848	-0.323	0.105	-0.13	-0.266	0.187	0.17	0.12364	0.2004	0.169	0.19351	0.0432	-0.0539	-0.0301	-0.038	-0.013	0.0071	0.29621	0.0599	-0.1891	-0.0606	-0.249	1	-0.243	-0.3103	0.30633	-0.085	-0.0241
NDVI	-0.296	0.1386	-0.065	-0.1	0.1247	-0.27	-0.31	-0.30623	-0.2996	-0.304	-0.2855	-0.006	0.04124	0.20942	-0.047	0.002	0.1572	-0.1615	-0.1125	0.28102	-0.0054	0.1222	-0.243	1	0.4988	-0.2896	0.2078	0.0974
LC	-0.205	0.0564	0.09	-0.24	-0.004	-0.15	-0.245	-0.25022	-0.2059	-0.229	-0.1864	-0.073	-0.0618	0.3323	-0.191	-0.095	0.31	-0.0411	-0.193	0.19157	0.0073	0.1524	-0.31	0.4988	1	-0.3532	0.2679	0.08
water	0.4471	-0.546	0.225	-0.1	-0.473	0.436	0.464	0.36977	0.4803	0.433	0.45035	0.1174	0.03608	-0.2579	0.0859	0.0772	-0.2039	0.34497	0.2394	-0.4536	0.0202	-0.165	0.3063	-0.29	-0.3532	1	-0.28	0.0416
road	-0.376	0.0067	-0.128	-0.09	0.0588	-0.35	-0.382	-0.39748	-0.351	-0.387	-0.3721	-0.121	-0.0926	0.31675	-0.166	-0.117	0.2704	-0.194	-0.1395	0.37257	0.0033	0.0585	-0.085	0.2078	0.2679	-0.2802	1	0.0353
population	-0.194	-0.043	0.014	-0.11	-0.034	-0.19	-0.186	-0.22142	-0.1708	-0.205	-0.1848	0.0512	0.02349	0.08257	-0.019	0.0284	0.0734	0.01056	-0.0059	0.19723	-0.0188	-0.016	-0.024	0.0974	0.08	0.04156	0.0353	1

Distance from the road contributes 4.3% to the distance of water bodies, which is only 2.1% to the distribution of *S. spectabilis*. The variables that showed high permutation importance were the distance from the road (21%) and population (16.5%). The sensitivity analysis using the jackknife technique also showed a higher gain for annual mean temperature (BIO1) with a gain of 0.82, whereas the aspect showed a lower test gain (Figure 4.19). The highest permutation importance is observed for the bioclimatic variable precipitation of the warmest quarter (19.3), followed by aspect and distance from the road (17). Nevertheless, the isothermality variable, which showed the highest percentage contribution to the distribution of the invasive species, showed the least permutation importance.

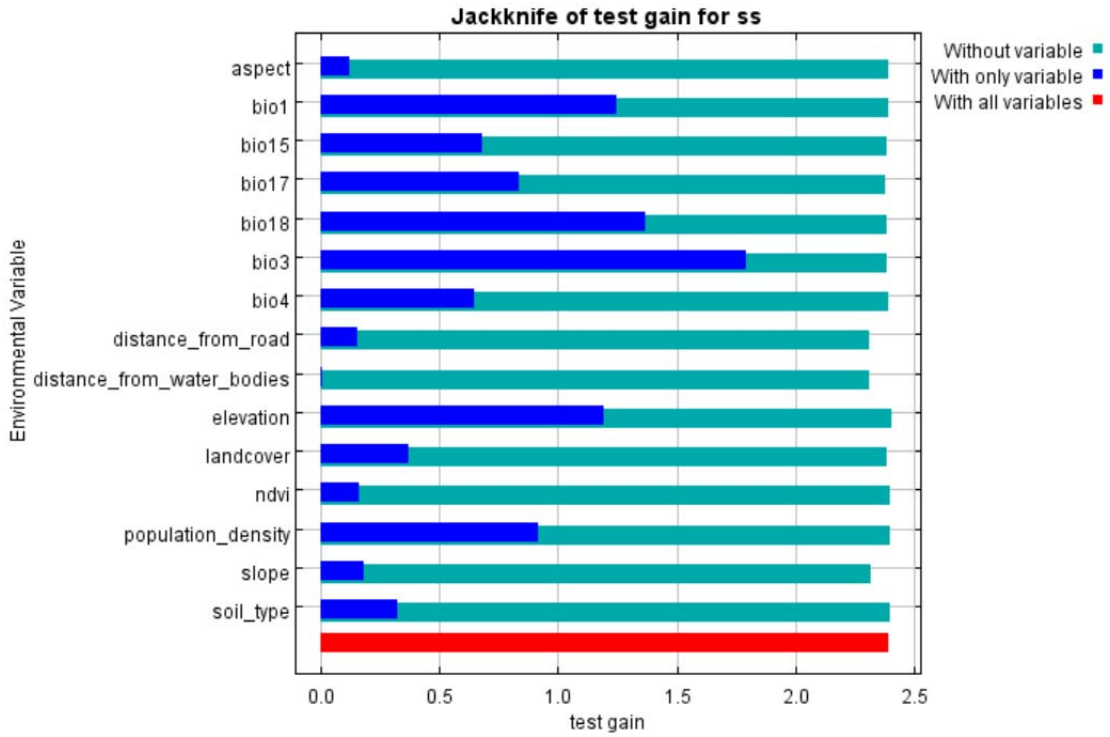


Fig.4.19. Results of the Jackknife test showing the influence (test gain) of each environmental variable relative to all environment variables in the MaxEnt modelling of the current distribution of *S. spectabilis* in Wayanad using selected variables

4.1.3.4. Response of *S. spectabilis* to environmental variables

The supplementary response curves also provided insights into the graphical representation of the impact of individual environmental variables on the distribution of the invasive species, as illustrated in Figure 4.20. Each response curve corresponds to a distinct model, specifically a MaxEnt model constructed solely based on the respective variable. The species' response to annual mean temperature (BIO I) (Fig.4.20.b) increased gradually when the temperature range was between 14^0C and 22.5^0C. When the isothermality (BIO3) (Fig.4.20.a) was in the range 55 to 57, the probability of presence for the *S. spectabilis* was greater than 90 per cent. However, the response of species to BIO3 remains constant. The probable presence of *S. spectabilis* was highest when temperature seasonality (BIO4) (Fig.4.20.c) was 165^0C to 170^0C. The probability occurrence of *S. spectabilis* was observed to be 95% when precipitation seasonality has value of 143 (Fig.4.20.d). The response curve followed a J-shaped curve and when BIO15 is between the ranges of value 40 – 90, a lower presence of the species is observed (15%) that remains constant. The precipitation of the Driest Quarter affects (Fig.4.20.e) model prediction as the species distribution

escalated when BIO14 rises above 15mm – 23mm, also the maximum probability of presence for the *S. spectabilis* (90%) when BIO17 is at 23mm (Fig.4.20.e).The similar pattern of two troughs and a peak is shown by response curves of elevation (Fig.13.g),

The highest distribution of *S. spectabilis* is observed (100%) when the precipitation of the warmest quarter (Fig.4.20.f) was 130mm – 200mm. The probability of occurrence of *S. spectabilis* was 95% when precipitation seasonality had a value of 143. A negative response of the aspect to *S. spectabilis* was observed when the aspect was 400m (Fig.4.20.h), and the chance of occupancy of *S. spectabilis* was comparatively less (50%). Consider the slop, probable presence of *S. spectabilis* gradually increased to 89m, followed by a sudden acceleration in distribution up to 74%, where the species distribution is at its peak (Fig.4.20.i).

When the land cover was deciduous forest or degraded/Scrub forest (Fig.4.20.l), the presence of *S. spectabilis* was highest (> 0.75). Variables such as distance from the road (Fig.4.20.m), distance from water bodies (Fig.4.20.n),NDVI(Fig.4.20.k), population density (Fig.4.20.j) also had a role in the probability of distribution of the *S. spectabilis*. The potential distribution of *S. spectabilis* decreased significantly with the increase in distance from water bodies and distance from the road.

S. spectabilis had a higher potential of occurrence in the forest soils, Black soils, Laterite plateau, Marayur soils, soils of Wayanad uplands, Upland soils of Palakkad central plain, lowland soils of Palakkad central plain and Poonthal padam soils of Palakkad eastern plain (>70%). The other soil types such as Gravelly laterite, Red soils, Brown hydromorphic soils, Riverine alluvium, Coastal alluvial oils, Coastal sandysoils, Onattukara sandy soils remained constant and had a probable positive response by *S. spectabilis* (40% chance of occupancy) which is given in (Fig.4.20.o) and soil type showed no significant change to the survival of the species. Response of *S. spectabilis* remained constant when population density increased above 1000 persons per km^2. The probable presence of the species was at its peak (85%) when the population density was 500 persons per km^2. When the normalized difference vegetation index (NDVI) increased (> $1*10^6$), the potential distribution of the species decreased. There is a 40-75% chance of occupancy of the invasive species when the index value is below $1*10^6$.

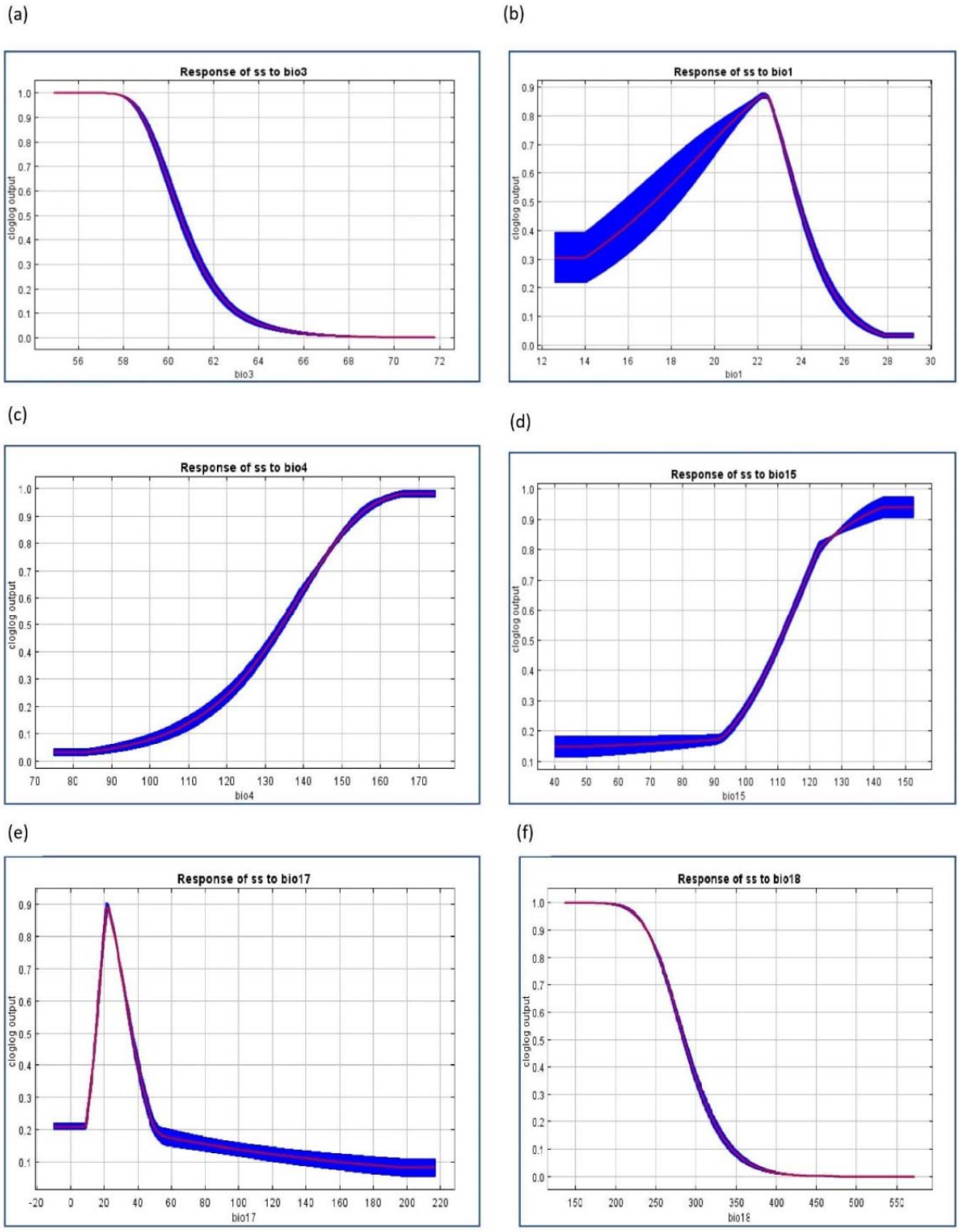


Fig. 4.20 Response curves of variables in determining the distribution of the *S. spectabilis* MaxEnt modelling, (a). Isothermality (BIO3), (b). Annual mean temperature (BIO1), (c). Temperature seasonality (BIO4), (d). Precipitation seasonality (BIO15), (e). Precipitation of driest quarter (BIO17), (f). Precipitation of warmest quarter (BIO18).

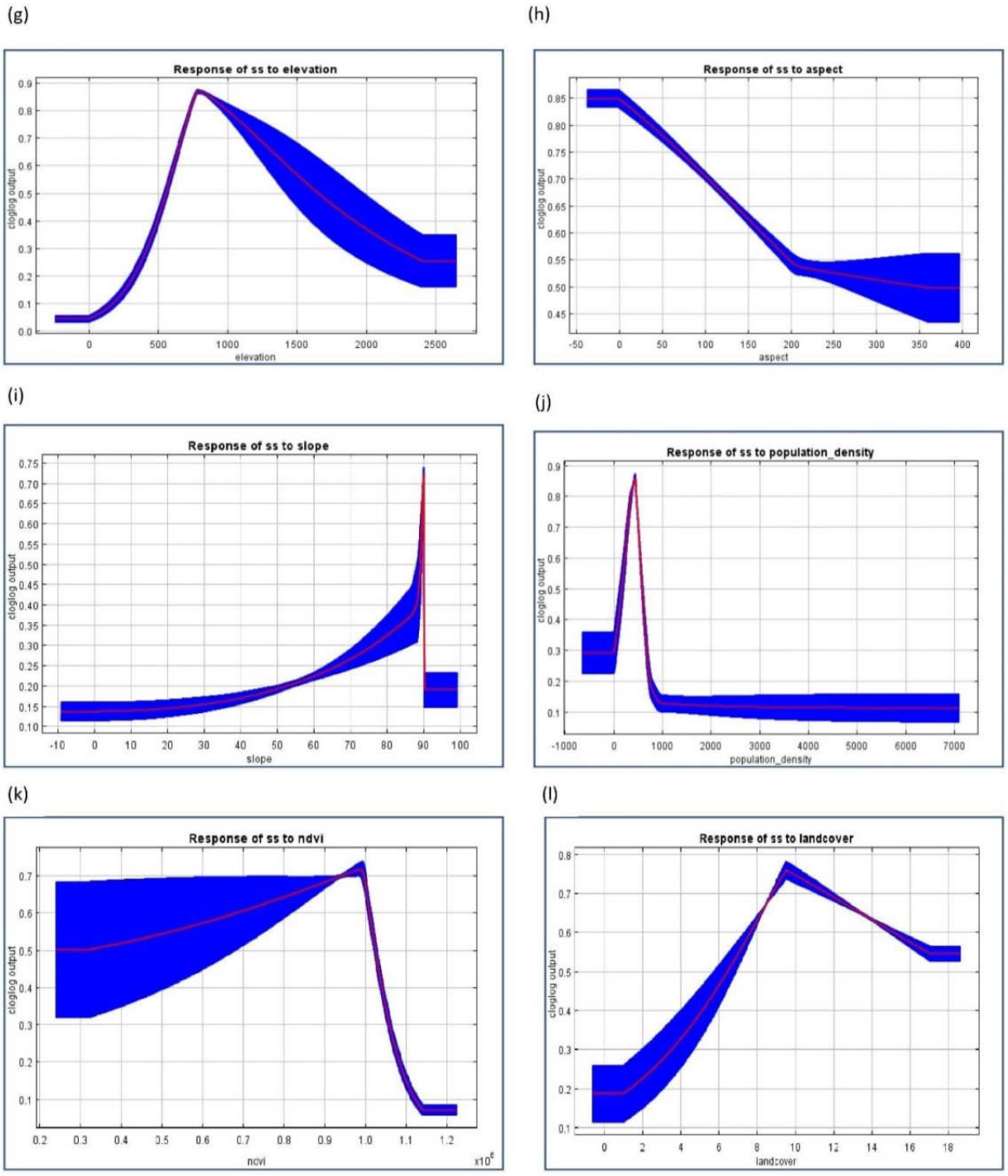


Fig. 4.20 Response curves of variables in determining the distribution of the *S. spectabilis* MaxEnt modelling (g). Elevation, (h).Aspect (i).Slope, (j). Population density, (k). NDVI, (l).Landcover

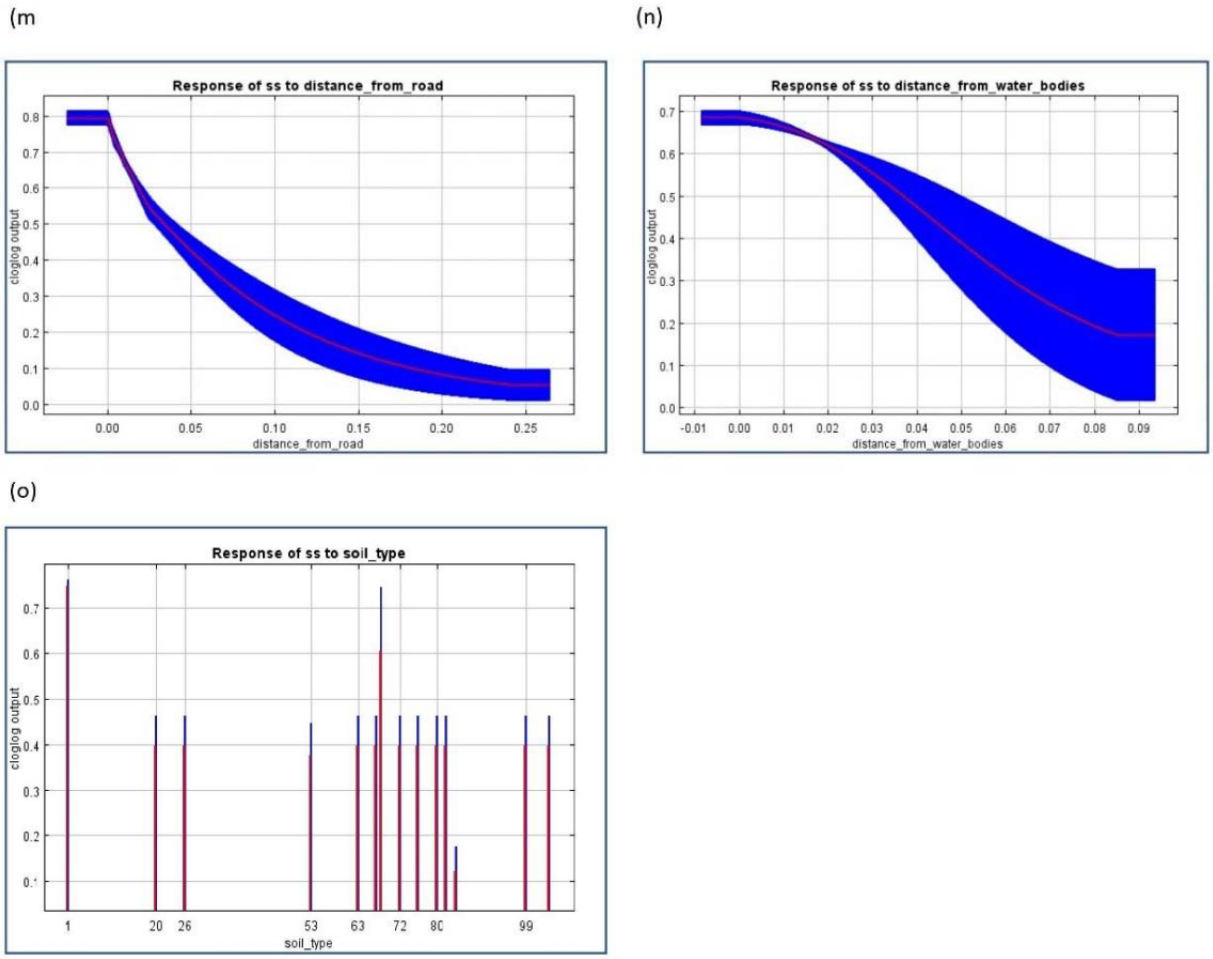


Fig. 4.20 Response curves of variables in determining the distribution of the *S. spectabilis* MaxEnt modelling (m). Distance from the road, (n). Distance from water bodies, (O). Soil type

The omission rate and predicted area of the test data average over the 10 replicate runs is given in Figure 4.21. The omission on test samples (blue line) showed a very good match to the predicted omission (black line) although, the predicted omission rate is a straight line. The test omission line is observed to be well below the predicted omission line considering the test data (75%) and training data (25%) are not independent.

4.1.3.5. The current distribution of *S. spectabilis*

The currently suitable habitat of invasive alien tree species *S. spectabilis* based on the presence records as given by the MaxEnt model is given in Figure 4.21.

The MaxEnt output ASCII files were reclassified using ArcGIS ver.10.7.1ESRI to obtain a logistic distribution which was then converted to binary raster for the easy

interpretation of suitable and unsuitable areas based on the 'max SSS' logistic threshold (0.52) obtained from the MaxEnt output. Out of 2364 km^2 total area, 1572 km^2 (66%) is suitable for *S. spectabilis*, and the remaining 821 km^2 (34%) area is unsuitable for its distribution. The logistic output is shown in Figure 4.21. The area is classified into low suitability areas with 344 km^2 (0 – 0.2), 249 km^2 with moderate suitability (0.2 – 0.4) potential, 428 km^2 with a good suitability potential (0.4 – 0.6), high suitability potential class (0.6 – 0.8) with 572 km^2 area and a very high suitability potential (0.8 – 1) area consisting of about 800 km^2 for the invasive species *S. spectabilis*. Most very high suitability areas were distributed in the North-eastern and South-eastern parts of Wayanad. Most high and very high suitability areas were covered in deciduous forests, degraded/scrub forests, plantation areas, barren areas/ wasteland areas and built-up areas.

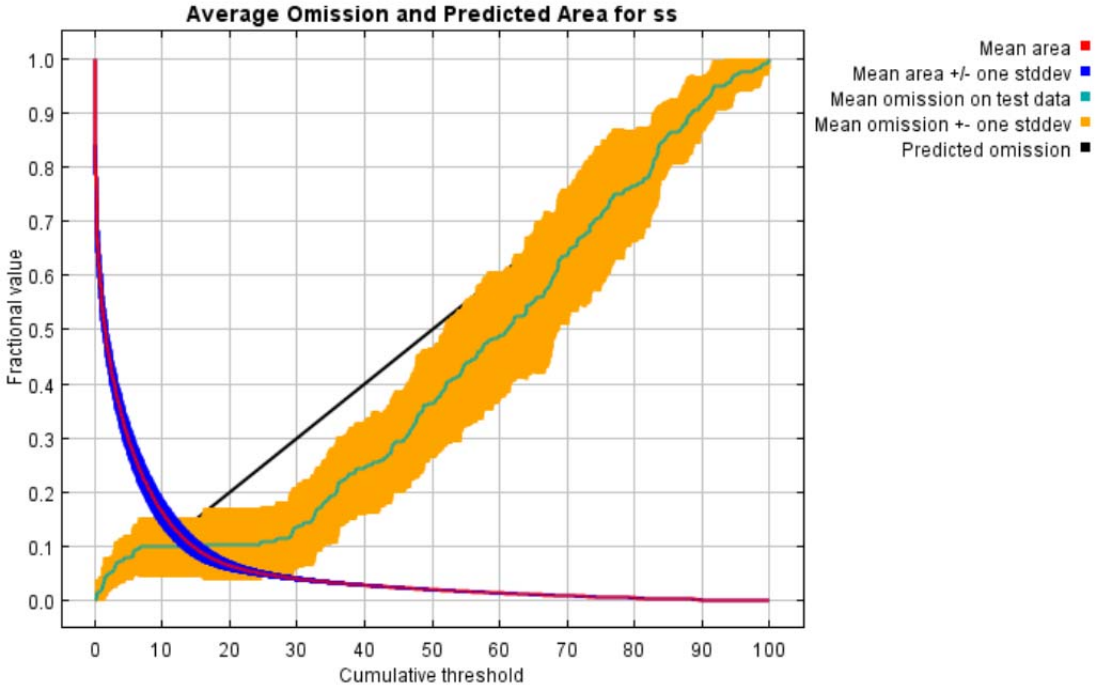


Fig. 4.21 Test omission rate and predicted area for *S. spectabilis* in the currentdistribution as a function of the cumulative threshold, averaged over the replicate runs for the current potential distribution MaxEnt modelling in Wayanad

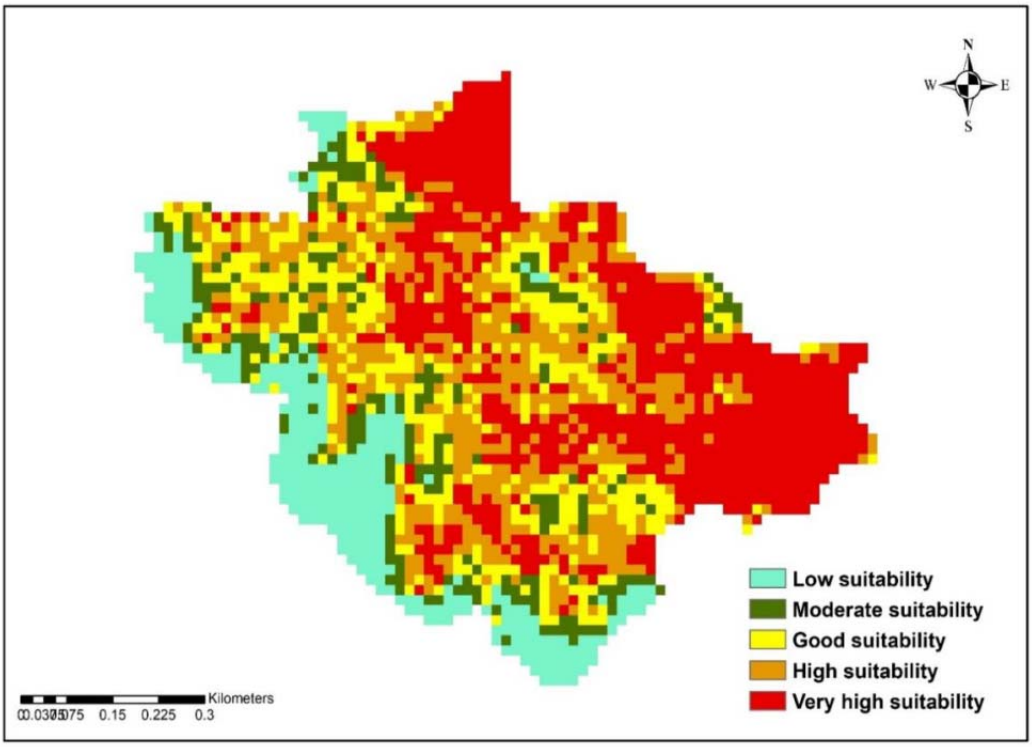


Fig. 4.22 Under current climatic conditions, the logistic output and the potential distribution MaxEnt modelling of *S. spectabilis* in Wayanad district.

4.1.3.6. Future changes in potential species distribution of *S. spectabilis*

The very high habitat suitability of the current scenario is observed to reduce to moderate and good suitability habitat areas. The low-suitability habitat areas are seen in the western parts of Wayanad, and the model showed a probable prediction of low-suitability areas in western parts of Wayanad in the future. The eastern parts of Wayanad, especially Wayanad Wildlife Sanctuary, are predicted to be under very high and high habitat suitability areas. The suitability class distribution of *S. spectabilis* in the 2050s indicates very high suitability (0.8 - 1) is in the RCP 4.5 scenario (475 km^2) compared to other RCP scenarios (Fig.4.23); however, the current scenario is found to be highest in the very high suitability area (800 km^2) for *S. spectabilis* than RCP 4.5 scenario. The predicted high habitat suitability (0.6 – 0.8) is higher in the RCP 4.5 scenario in the 2050s and lower in the RCP 6 scenario. Suitable habitat suitability is predicted to be higher in the RCP 6 scenario than in the RCP 4.5 scenario.

Moderate and low habitat suitability for the probable distribution of *S. spectabilis* is predicted to be more significant in the RCP 6 scenario among other RCP scenarios and lower in the RCP 4.5 scenario. In the 2070s, the predicted habitat

suitability areas of the RCP scenarios varied from the 2050s. The suitability class is higher in RCP 2.6 (475 km^2) than in the other representative concentration pathways. In general, there is a decreasing trend in the high habitat suitability class among RCP scenarios from 2.6 to 8.5 watts/km^2, although the RCP 4.5 scenario has the most minor habitat suitability area. Comparing the excellent habitat suitability area among all the observed RCPs, the RCP 6 scenario has the highest suitability area, and RCP 4.5 has the least suitability. The moderate habitat suitability (0.4 – 0.6) and the low suitability (0 – 0.2) are predicted to be higher in the RCP 4.5 scenario and lower in the RCP 2.6 scenario.

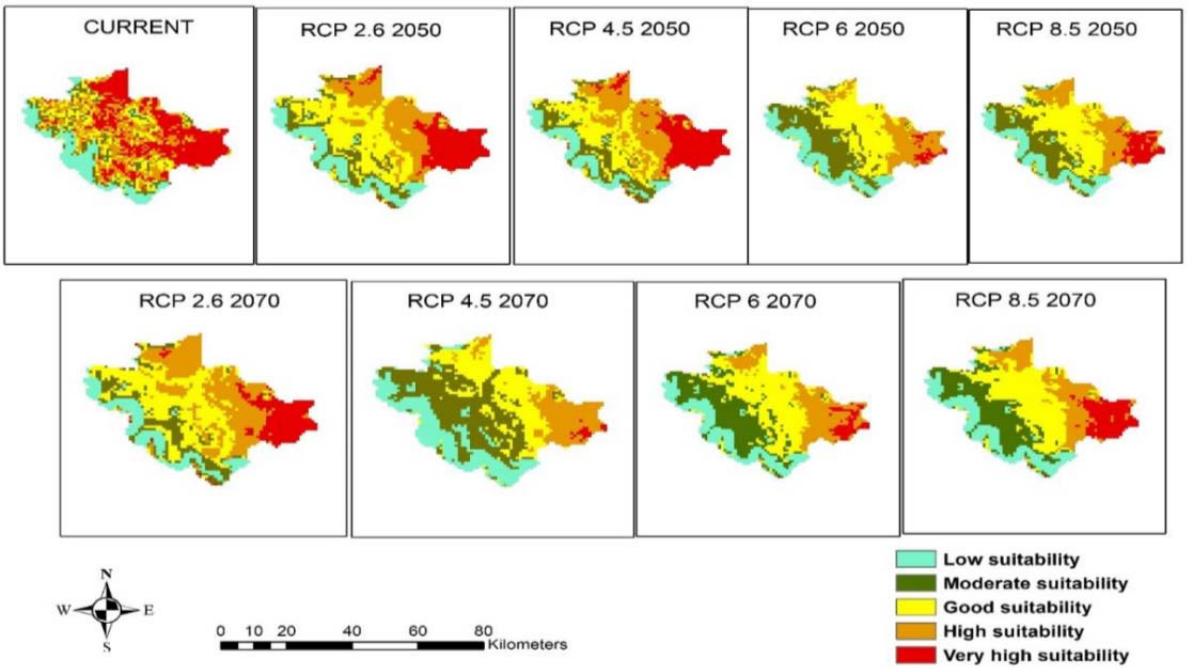


Fig. 4.23 Predicted potential distribution of *S. spectabilis* under the current scenario and various RCP scenarios for both 2050s and 2070s in Wayanad.

4.1.3.7. Distribution changes

By estimating the difference between current and future binary distribution maps, the relative changes in the future potential species distribution and the impact of climate change were observed. The criteria used for change analysis were range expansion, range contraction, no change (presence in both), and no occupancy (absence in both). Comparing both 2050 and 2070 RCP scenarios (Fig.4.24), the range expansion is higher shortly (2050s) than in the distant future (2070s), and RCP 4.5 is the highest. The highest range contraction is in 2070, RCP 4.5. Comparatively, the range

contraction is higher in the 2070s than in the 2050s. The no-changing area (presence in both) is higher in the 2050s, whereas no occupancy is higher in the 2070s.

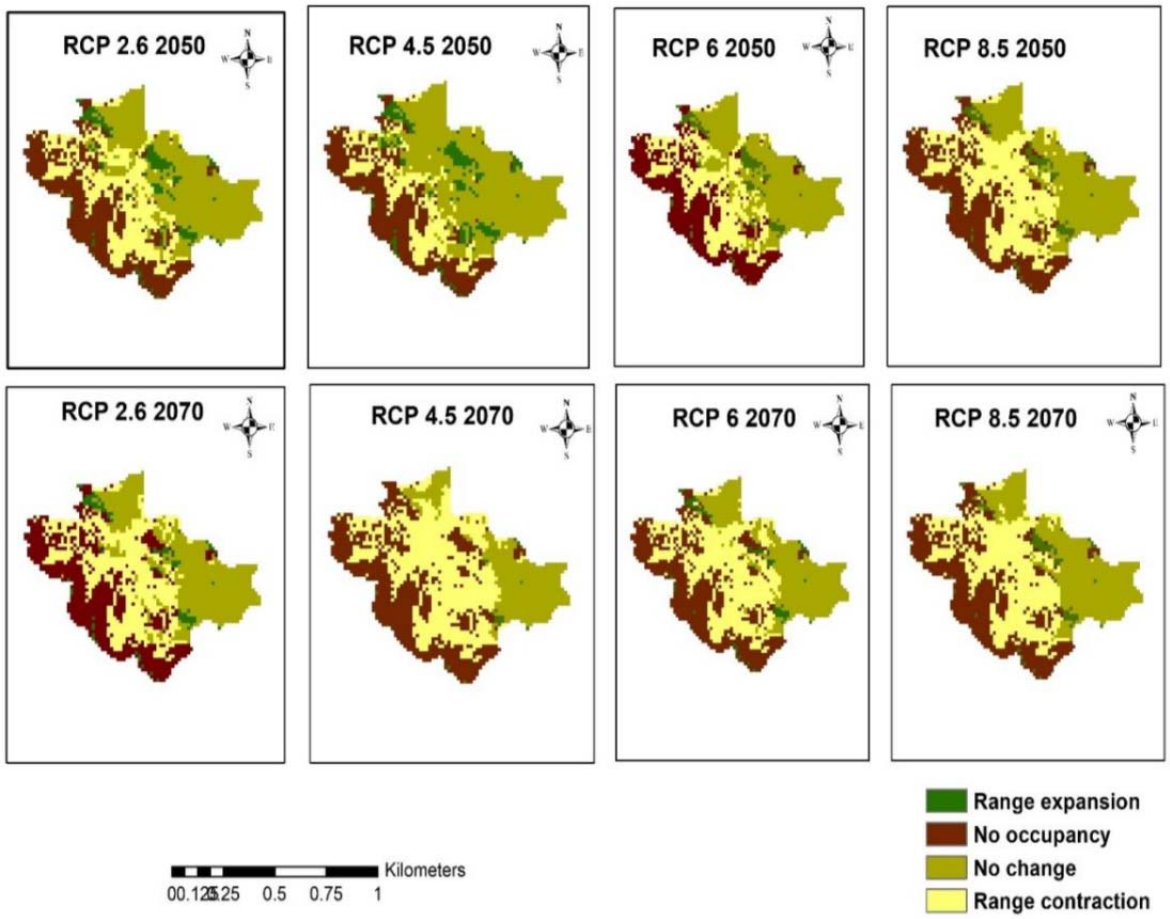


Fig. 4.24 Distributional changes of *S. spectabilis* in Wayanad under RCP scenarios in the near future (2050s) and the distant future (2070s).

The research findings indicated that the proliferation and abundance of invasive species were subsequently facilitated by the increasing temperature, modified precipitation patterns, and various anthropogenic disruptions (Easterling *et al.,* 2000; Hellmann *et al.,* 2008; Walther *et al.,* 2009). According to Tripathi *et al.,* (2019), when the mean temperature increased, it was convenient for the invasive species as the increase in mean temperature escalated the growing season length, thus creating many vacant spaces where invasive species successfully adjust. Temperature-related bioclimatic variables BIO3 and BIO1 (Isothermality and annual mean temperature) have contributed significantly to the distribution with a cumulative contribution of 45.2%. These variables indicated that the temperature was an inevitable factor in

determining the distribution of this invasive species. Similarly, Weldemariam and Dejene (2021) confirmed that temperature variables were the most critical for establishing the *S. spectabilis*. Averett *et al.,* (2016) established that temperature variables are the most influential predictor variables that limit the distribution of non-native species richness. At the same time, the precipitation variables, precipitation of the warmest quarter (BIO18), precipitation seasonality (BIO15), and precipitation of the driest quarter (BIO17) have a cumulative contribution of 10%.

Compared with the other temperature variables, temperature seasonality (BIO4) has had a lower percentage contribution (3.3%). Elevation was the second most influencing variable, contributing 20.8%. Other than these variables, slope, landcover, and distance from the road had a significant role in the distribution of *S. spectabilis,* as they contributed 6.4, 6 and 4.3 per cent, respectively. NDVI, aspect, soil type, and population density had less than one per cent contribution, whereas distance from water bodies contributed 2.1% to the distribution model. Additionally, the current distribution model result concurs with that of the report made by Sajeev *et al.,* (2012) and, Satyanarayana and Gnanasekaran (2013), Singh (2001), who put *S. spectabilis* as among the most occurring alien invasive species in many habitats of Peninsular India as well as categorised under the medium risk category. Besides, this current invasion distribution model result was in parallel with studies conducted in Wayanad (Anoop *et al.*, 2021) because the model predicted that 66% of the total area of the Wayanad district (1572 km^2) is currently suitable for *S. spectabilis.* According to Anoop *et al.,* (2021), 23% of the Wayanad Wildlife Sanctuary area has been covered with this invasive species in 40 years since the 1980s. The very high habitat suitability area in the Wayanad wildlife sanctuary in the current scenario accounted for 83% of the potential area compared to the area of Wayanad wildlife sanctuary (344.44 km2) followed by a high habitat suitability area with 14% potential area in the wildlife sanctuary that included Tholpetty range, Kurichyad range, Sulthan Bathery range, Muthanga range. The species distribution was highest (85%) when the mean annual temperature was 22.5^0C.

The mean annual temperature (BIO1) in the Wayanad district under the current scenario was 22.23^0C, which is highly suitable for the distribution of *S. spectabilis* (CABI, 2021). Besides, isothermality (BIO3) was found to be 59.46, shown in Figure 6, which seemed to increase the species distribution by about 98%, although the response of isothermality to *S. spectabilis* distribution was a negative relationship.

The isothermality variable was the most influencing variable. Besides, the test gain values were the highest for the isothermality variable, giving the most helpful information. The most abundant distribution in the Wayanad district was seen in the deciduous forest, which was favourable for 86% of the potential suitability area. However, a low suitability habitat distribution was seen in the evergreen forest, part of the southern Western Ghats in the western Wayanad. Because of this variable, the *S. spectabilis* could not be established under a whole canopy forest (PIER, 2014). Most modelled high habitat suitability of *S. spectabilis* in the current scenario was seen in the elevation range between 500 – 750 m, which is the eastern and central parts of Wayanad, including Wayanad wildlife sanctuary. The response of elevation to the invasive species *S. spectabilis* distribution increased with the increased elevation; beyond the elevation of 750m further, increased elevation had a decreased response in species distribution, reaching no change at 2400m, shown in Figure 4.20.g. During the survey for documenting the *S. spectabilis* occurrence and populations, the invasive nature of this species was found only in between 500-1200 msl in Kerala, such as Attapady, Wayanad and Periyar regions. However, distance from road and distance from water bodies showed a negative response to *S. spectabilis* distribution in the Wayanad district as predicted by the MaxEnt model in the study. In current climatic conditions high habitat suitability potential was found in the low disturbance area. Additionally, distance from roads had the most information not present in the other variables. Furthermore, the model predicts that the influence of distance from the road would play an essential role in the distribution of *S. spectabilis*.

The abundance of *S. spectabilis* was mainly because of its high adaptive capacity in any conditions, including its high coppicing ability, allelopathic effect, seed viability, and lack of natural enemies. Moreover, the study of Anoop *et al.* (2021) suggested that native mammals, specifically elephants, transport *S. spectabilis* to a considerable distance and thereby could play a role in the current and future distribution of the species, which may indicate that the biotic factors contributed to the spread of this species. Furthermore, studies observed a temporal overlap between *S. spectabilis* seed fruiting and high elephant density in the Nilgiri biosphere reserve (Anoop *et al*., 2021). Thus, the dispersal mechanisms also played a significant role in the distribution of *S. spectabilis* in the regions of Wayanad Wildlife sanctuary apart from the dispersal pathway.

The study also focused on finding the future distribution of *S. spectabilis* to understand whether future climatic conditions could promote the distribution and invasiveness. The decrease in the temperature profiles, BIO1 and BIO3, favoured the distribution. However, the temperature seasonality increases between the ranges 90 - 165^0C favoured the distribution. Regarding the precipitation variables, BIO17 and BIO15 increased precipitation in the specified range favoured the current distribution. BIO18, however, was unfavoured as it increased. Accordingly, the model results provided by this study have an important implication in the management measures.

The study modelled using the optimized variables under four different Representative Concentration Pathways (RCP) such as RCP2.6, RCP4.5, RCP6, and RCP8.5, predicted the future distribution of the *S. spectabilis* in Kerala for the years 2050s (average for 2041 –2060) and 2070s (average for 2061 – 2080). The distribution change of the *S. spectabilis* showed that in all the greenhouse gas pathways compared to the current scenario, most parts of the Wayanad district and mostly Wayanad wildlife sanctuary had no change in distribution. In the period 2050s, the RCP 4.5 scenario showed nearly half of the suitability area had no change compared to the current scenario, followed by 40% of no change distribution potential in parts of Wayanad in the RCP 2.6 scenario. RCP 6 and RCP 8.5 scenarios also had considerable no-change distribution potential with 39% and 32%, respectively.

Focussing on the protected areas gave a shocking result as the entire Wayanad wildlife sanctuary had 99% no change distribution potential compared to the current RCP 2.6, RCP 4.5, RCP 6 scenarios and 96% in RCP 8.5 scenarios. It could be regarded as an alarming call for action as in the earlier studies (Anoop *et al.*, 2021) reported that 23% of the Wayanad wildlife sanctuary is found to be distributed with *S. spectabilis*, Prajitha and Sudhabai (2022) reported that the Wayanad Wildlife sanctuary been invaded by *S. spectabilis*, covering approximately 25% of the sanctuary. Moreover, this study showed that very high suitability for the invasive species accounted for 86% of the sanctuary. This invasion could hamper biodiversity and lead to species extinction of the flora and fauna, creating an imbalance in the ecosystem. Although in Wayanad district, the exceptionally high habitat suitability area in all the RCP scenarios decreased compared to the current scenario, the high, moderate and reasonable suitability areas showed an increase compared to the current scenario. However, the moderate suitability area decreased in the wildlife sanctuary. Wildlife sanctuaries showed chiefly high and very high habitat suitability areas, and

compared to the current scenario, the high suitability would increase, whereas very high suitability would decrease in all the RCPs. It could be chiefly attributed to the decrease in the isothermality variable (BIO3), which outperformed the rest of the contributed variables by far. There was a 100% distribution response to *S. spectabilis* in the area when isothermality was the lowest.

The distribution change in no occupancy would be comparatively more significant than range contraction in the RCP scenarios except the RCP 8.5 scenario. The no occupancy distribution regions would be mainly the western parts of Wayanad, with the evergreen forests and agricultural regions in the Western Ghats, which were classified under the low habitat suitability area for the *S. spectabilis*. The model predicted the highest range contraction in the RCP 8.5 scenario with 34% potential area compared to the current scenario in the 2050s. Similarly, the RCP 8.5 scenario is the highest in the wildlife sanctuary.

The rise in the annual mean temperature (BIO1) and increased annual precipitation (BIO12) increased range contraction in RCP scenarios as the ideal temperature for *S. spectabilis* is 19 - 22^{0}C and the precipitation range between 800 – 2000mm (CABI, 2021).

The report by Kerala State Action Plan on Climate Change (2014) predicted an adverse change in the variation of rainfall in Wayanad district in the 2050s. The range expansion in the Wayanad district and as well as in the wildlife sanctuary was predicted to be significantly less compared to range contraction. However, range expansion in wildlife sanctuaries would be greater than no occupancy distribution change under RCP scenarios in the 2050s. The highest range expansion would be found in the RCP 4.5 scenario in both the sanctuary and Wayanad district and the lowest in the RCP 8.5 scenario. The more significant range expansion in the RCP 4.5 scenario, among other RCPs, could be attributed to the favourable range of isothermality, precipitation seasonality, precipitation of driest quarter and temperature seasonality for the distribution of *S. spectabilis*. Therefore, for the period the 2050s, the RCP 4.5 scenario has the highest habitat suitability for *S. spectabilis* in both the wildlife sanctuary and Wayanad district. In contrast, the RCP 8.5 scenario would be unfavourable for habitat suitability, leading to a more significant range contraction in both the wildlife sanctuary and Wayanad district in the period of the 2050s.

Unlike the 2050s, the model projection of *S. spectabilis* in all four RCP scenarios in the period 2070s showed that the range contraction would be greater than the no

change, no occupancy, range-expansion distribution change areas in the Wayanad district whereas, in the Wayanad wildlife sanctuary, the no-change distributional change would be more significant than range contraction followed by no occupancy and range expansion. Range contraction was higher in the RCP 4.5 scenario, about 45% in the 2070s and the higher range expansion with the RCP 2.6 scenario among the RCP scenarios in the Wayanad district. Similarly, the range contraction in the wildlife sanctuary was highest in the RCP 4.5 scenario, with the highest range expansion in the RCP 2.6, RCP 6 and RCP 8.5 scenario. Moreover, RCP 4.5 also showed the highest no-occupancy distributional change in the area in the sanctuary. Furthermore, 96% of the potential wildlife sanctuary area would be no change distribution area in RCP 6, RCP 8.5 and RCP 2.6, while there would be a decrease in RCP 4.5 scenario. The very high suitability was higher in RCP 2.6, among other scenarios. The sole reason for less distribution of *S. spectabilis* in RCP 4.5 scenario could be the higher isothermality (BIO3) among other RCP scenarios.

The isothermality response to *S. spectabilis* distribution showed a negative relationship. Furthermore, the temperature seasonality (BIO4) was very low in RCP 4.5 compared to other RCPs. Moreover, the increased precipitation of the driest quarter and precipitation of the warmest quarter beyond the suitable range contributed to lower suitability than other RCPs in the 2070s.

The most favourable habitat in the RCP 2.6 scenario could be attributed to low precipitation of the driest quarter, which lies in the suitable range between 15–23 mm, higher temperature seasonality among the RCPs, and the isothermality values of RCP 2.6 in the suitable range. Even though the future projected model showed variability among RCPs, and among the periods of 2050s and 2070s, it predicted a decrease in habitat suitability compared to the current scenario. However, there was found to be no significant change in the Wayanad Wildlife Sanctuary, which is an alarming call for action.

The predicted results of climatic suitability in the future scenario in Wayanad have broader similarities with Adhikari *et al.,* (2015), who found high climatic suitability in Wayanad despite the study on combined model projection from all five continents. According to Shreshta *et al.,* (2012), the impact of climate change is likely to be more drastic in high-elevation regions, possibly due to more significant temperature change in those areas compared to lowlands and midlands. Furthermore, there are studies accounting for the species invading higher altitude areas, currently

than in the past (Shrestha *et al.*, 2015; Tiwari *et al*., 2005). However, Wayanad's high and very high habitat suitability areas tend to decrease in future scenarios compared to the current scenario. Additionally, it was visualized that very high suitable areas under current climate conditions are prone to lose their suitability into good, moderate and low suitability ranges under future climatic conditions, which was in line with the study of Weldemariam and Dejene (2021) predicting the invasion hotspots of *Senna didymobotrya* in Africa. Biological invasion of *S. spectabilis* will enhance pressure and add risks to vulnerable ecosystems in future in eastern parts of Wayanad, especially the Wayanad Wildlife Sanctuary, as it is already vulnerable to climate change and experiencing its repercussions. Furthermore, the high coppicing ability, allelopathic nature and viability of the seeds of *S. spectabilis* give it an advantage for establishment in the invaded region and ecosystem destruction. Although climate change created some novel climatically suitable habitats for *S. spectabilis*, the model predicted significant contraction, no occupancy and no change areas in Wayanad, which had the upper hand compared to range expansion in future scenarios in both periods. Furthermore, the high habitat suitability decreased compared to the current scenario. Additionally, there was no significant shift in the future range. Therefore, the undertaken study articulated that Wayanad had no large range of new invading areas.

The study's hypothesis, that climate change would likely increase its occurrence probabilities was thus proven wrong. There was no profuse expansion due to increased temperature and rainfall; instead, there was a large range contraction and no change areas. The protected areas (Tholpetty range, Muthanga range, Sulthan Bathery range and Kurichyad range) were in danger due to invasion risk in both current and future scenarios, although there would be less expansion. With this result, management measures can focus on the predicted habitat suitability areas, primerly to the high-risk areas then monitor the potential invasion areas. The distribution modelling can aid in the risk assessment measures and, thus, the eradication procedures.

The result emphasized the very high suitability of the *S. spectabilis* in the current scenario in Wayanad and the wildlife sanctuary. Currently, 86% of the wildlife sanctuary area is under the very high habitat suitability category.

The astonishing result is that despite no considerable range expansion, more than 90% of the potential wildlife sanctuary area under the habitat suitability remained as

such in future scenarios as in the current scenario. Looking into the Wayanad district, there is a considerable range contraction, and in western Wayanad, part of Western ghats with no occupancy or low suitability areas for *S.spectabilis,* which is a relief. However, the distribution of *S. spectabilis* could lead to the extinction risk of many flora and fauna if the right measures are not taken at the right time.

Chapter 4.2

Study of pollination, seed dispersal, phenology and variability of *Senna spectabilis* populations in Kerala

4.2.1 Phenological pattern of *Senna spectabilis* (DC.) H.S. Irwin & Barneby.

The phenological patterns of the invasive tree *Senna spectabilis* were observed in Wayanad Wildlife Sanctuary and Anaikkaty regions. The influence of climatic factors was observed in the vegetative and reproductive patterns. The evaluated phenological events were vegetative events, such as leaf flushing, mature leaf and leaf abscission, and reproductive phenophases classified as flower bud initiation and anthesis or open flower for flowering and immature fruit and mature fruits for fruiting phenophases. The phenological stages were examined in relation to climatic factors such as maximum and minimum temperature, as well as rainfall, employing Spearman's rank correlation. The vegetative phenological stages were consistently observed throughout the entire duration of the study.

Observations on Leaf Phenology: The phenology of this invasive tree was dependent on meteorological variables such as rainfall and temperature. Leaf phenophases such as leaf flushing and leaf abscission were more dependent. The leaf flushing started in the pre-monsoon time, and its maturation continued with the rainy season. The observed results revealed that leaf flushing had a high positive correlation with the rainfall of the three assessment years (r_s>0.75) (2018-2021) (r_s=0.787; $p<0.05$, r_s=0.779; $p<0.05$, r_s=0.833; $p<0.05$.) (Table 4.13). The average rainfall during the study period was 814.4, 248.3 and 293.1 mm, respectively (Figure 4.25). In these corresponding years, June recorded >80% leaf flushing intensity (Figure 4. 26). Leaf flushing started in the last week of May and ended in the first week of December. All three assessment years showed the same trend: the temperature results were negatively correlated. The average maximum temperature was 33^0C, and the

minimum was 21^{0}C (Figure 4.25). However, the phenol events of leaf maturation and leaf abscission did not correlate with the rainfall.

The intensity index calculated for the leaf phenophases (Figure 4.26) demonstrated that matured leaves were seen throughout the studied years. The prominent peak was observed during the post-monsoon period, and the minor peak was during the pre-monsoon period. However, the invasive tree lacked a complete deciduous nature. The intensity index showed that the peak of leaf abscission was observed in the summer months, such as March, April and May. It was a moderate positive correlation and significant with the maximum temperature (r_s=0.743; p<0.0.5) (Table 4.13) during the observed year of 2019-20 when an average maximum temperature of 36^{0}C and minimum of 25^{0}C (Figure 4.25) were recorded.

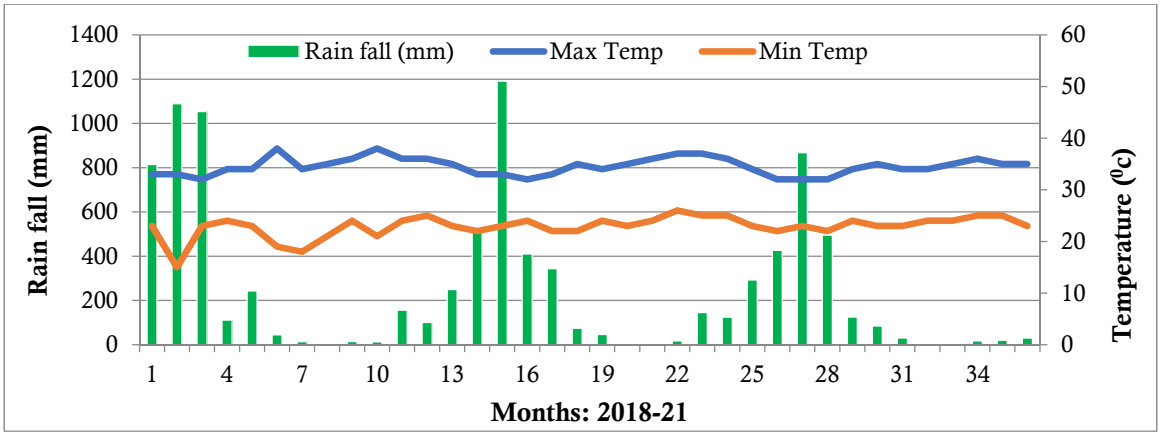


Fig. 4.25 Ombrothermic diagram for the study areas mean monthly temperature and rainfall.

During the assessment years of 2018-19 and 2020-21, leaf-falling activity positively correlated, but the association between the two variables could not be considered significant. Leaf abscission≥ 80% was observed in the pre-monsoon period, April and May. The effectiveness of management methods in controlling the invasive tree *S. spectabilis* will be higher in April and May due to low physiological activities during this period. Leaf flushing intensity is high (>80%) (Figure 4.26) in June, i.e., at the onset of monsoon, so that management methods can be effectively applied before the start of monsoon.

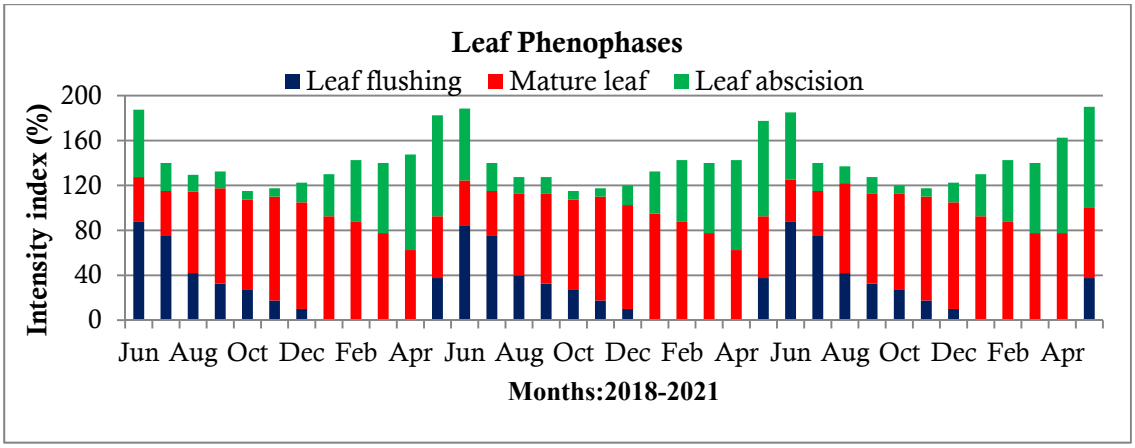


Fig. 4.26 Fournier intensity index of leaf phenophases for *S. spectabilis*

Observations on Flowering Phenology: High reproductive capacity is an important characteristic that makes species invasive. Seasonal and climatic factors also aid this characteristic.

Flowering Phenophases were recorded from August to March, and the intensity index demonstrated that ≥80% (Figure 4. 27) of flowering phenophases were observed from November to January. Flower bud initiation, along with the opening of the flower, maintained the peak months. No significant positive correlation with the meteorological factors was recorded in these two phenophases. During 2018-19, Spearman's rank correlation coefficient has shown maximum temperatures with a high positive correlation but is not significant (r_s=0.109; p>0.05, r_s=0.239; p>0.05) (Table 4.13). Except for the rainfall data of 2020-21 (r_s=0.007; p>0.0.5) (Table 1) and the maximum and minimum temperature of that year, other meteorological variables were negatively correlated. Though flowering events were observed in all the marked trees synchronously during the flowering seasons of 2018-19, 2019-20 and 2020-21, flowering was not simultaneous. The observations of the flowering phenology of S. spectabilis showed that mass flowering occurred regularly and flowering intensity remained the same from year to year. Such a flowering pattern appears to function as a reproductive success for invasion.

Table 4.13 Spearman's correlation coefficients (r_s) for phenology of *S. spectabilis*

Pheno-phase	2018-19			2019-20			2020-21		
	Rainfall (mm)	Temperature(^{0}C)		Rainfall (mm)	Temperature(^{0}C)		Rainfall (mm)	Temperature(^{0}C)	
		Max	Min		Max	Min		Max	Min
LF	0.787*	-0.712*	-0.033	0.779*	-0.578*	-0.450*	0.833*	-0.619*	-0.769*
LM	-0.723*	0.341	0.251	-0.594*	-0.057	-0.101	-0.309	0.05	0.16
LA	-0.189	0.316	0.406	-0.316	0.743*	0.682*	-0.533	0.524	0.385
FBI	-0.413	0.109	-0.376	-0.214	-0.391	-0.386	0.007	-0.11	-0.037
FOP	-0.619*	0.239	-0.398	-0.446	-0.237	-0.248	-0.218	0.019	0.069
FIM	-0.688*	0.368	-0.338	-0.582*	-0.102	-0.197	-0.443	0.209	0.272
FM	-0.951*	0.609*	-0.224	-0.961*	0.663*	0.318	-0.886*	0.611*	0.553

LF: Leaf flushing, LM: Mature leaf, LA: Leaf abscission, FBI: Flower bud Initiation, FOP: Anthesis/Open flower, FIM: Immature fruit, FM: Mature fruit

Until the middle of the flowering period, younger inflorescences were documented in *S. spectabilis* and mature inflorescence within the same tree. Developing inflorescences that have not reached the stage of dormancy will never flower. Dormant inflorescences started to swell and flower their buds. Later, they entered into anthesis when the older inflorescences were wilted and abscised or formed fruit. Incomplete flowering of single inflorescences happened throughout the flowering of single tree individuals in a single flowering season. So, this species has a lengthy flowering period. The length and intensity of the flowering period were a favourable indication of the invasive nature of this tree and may promote the species' reproductive capacity.

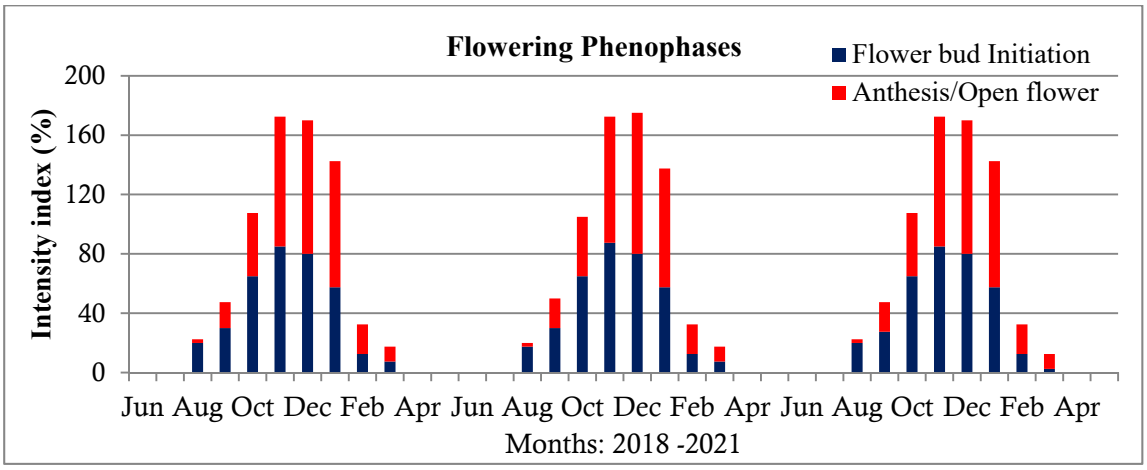


Fig. 4.27 Fournier intensity index of flowering phenophases for *S. spectabilis*

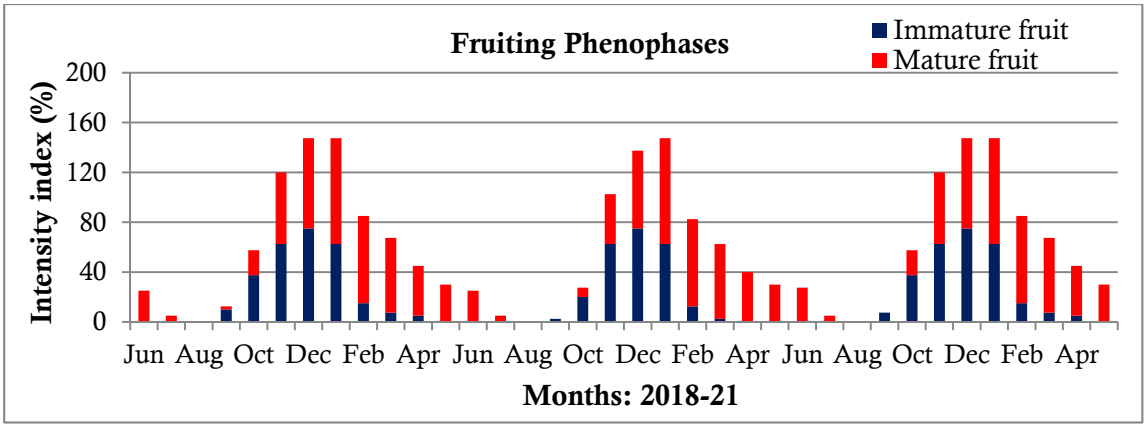


Fig.4.28 Fournier intensity index of fruiting phenophases for *S. spectabilis*

Observations of fruiting phenology: Immature fruit were found from September to March, and mature fruit were observed from October to June during the entire study period. According to the intensity index, >60% of immature fruits were recorded in November, December and January, and >80% of mature fruits were observed in January (Figure 4.28). February onwards, the mature fruits, i.e., in *S. spectabilis*, the dry pods started dispersing, or the pods were shed into the ground by barochory. In the three observation years recorded, the maximum temperature showed a high positive correlation and significance with the maturation of fruit (r_s=0.609; $p<0.05$, r_s=0.663; $p<0.05$, r_s=0.661; $p<0.05$) (Table 4.13). In the case of the immature fruit period in 2018-19, maximum temperature showed a high positive correlation but was not significant, where as in 2020-21 maximum and minimum temperatures showed a high positive correlation even though it was also not significant. An average temperature of a maximum of 36^0C and a minimum of 24^0C were recorded in the peak phenophases during the study period. The fruit's maturation corresponded to the maximum temperature in the dry summer season in the studied area. Immature fruits were green; later, they turned dark green to fully matured dark brownish pods. According to the field observation, the total period required to complete the entire process of fruit formation was 80 to 90 days. This fruiting behaviour has helped *S. spectabilis* to disperse the seeds timely and thus facilitate successful seed germination during the monsoon season that begins in June in the studied area.

The phenological events of *S. spectabilis* were distinctly seasonal in the studied areas. Leaves emerge and mature in a period with rainfall, and leaf flush begins with the pre-monsoon shower. High temperature and increasing length of the day trigger the

abscission of leaves. A significant peak in reproductive events was observed at the start of the post-monsoon period.

To date, no conclusive documentation or research on the phenological occurrences of *S. spectabilis* exists. Satyanarayana *et al.,* (2013) have noted that the flowering of *S. spectabilis* typically takes place between August and March, while http://www.flowersofindia.net reports that the flowering period is from October to December. As per Agroforestry Database 4.0 (Orwa *et al*., 2009), in Zambia, flowering occurs between January and February, while the ripening of fruits typically transpires in September or October. Conversely, in the United States, the process of flowering is observed consistently throughout the entire year. However, detailed phenological records have yet to be reported from its native range and other regions. This study illustrates the phenological pattern of *S. spectabilis* within its invasion range.

Linking phenological patterns to plant functional traits may help better understand plant community function and assembly during the invasion. According to Sakai *et al.,* (1999), research indicates that phenological studies pertain to the chronology of recurring biological occurrences. The phenology of plants often exerts a substantial impact on animal populations by causing temporal fluctuations in the availability of resources. Furthermore, phenological patterns can be influenced by various biotic factors, including competition, herbivory, pollination, and seed dispersal, in addition to a range of climatic variables.

Numerous studies have revealed a favourable correlation between the flowering phenology of non-indigenous species and their capacity for invasiveness. The flowering phenology of invasive species, along with any potential disparities in comparison to native species, is merely an outcome of distinct historical occurrences of human-mediated introductions (Goodwin *et al.,* 1999; Pysek et al., 2003; Lake and Leishman 2004; Cadotte and Lovett-Doust, 2001).

According to Taylor *et al.,* (2020) enhanced comprehension of the biology and temporal patterns of invasive plant species phenology can enhance the efficacy of land managers in selecting a suitable treatment approach and executing it in the field.

4.2.2. Reproductive biology of *Senna spectabilis* (DC.) H.S. Irwin & Barneby

Floral Biology: The phonological observations revealed that the peek flowering of *S. spectabilis* usually begins in September and continues up to December. Inflorescence is a raceme, terminal or axillary, with 10 to 15 cm long corymbose panicles. It has 120 to 140 flowers; peduncles are 2 to 3 cm long, while pedicels are 2 to 3 cm long. Its bracts are observed to be narrowly ovate or lanceolate, acute or subacuminate at apex and are caduceus. It has five sepals, which are unequal and reflexed. Among them, the outer two green are ovate, 5.50 x 3 mm long, concave and pubescent. The inner three petaloids are rotund or ovoid, 9 to 10 x 10 to 13 mm long, inconspicuously veined and pubescent. 5 petals are unequal, ovoid and 2 to 2.50 cm long. They are clawed at the base, entire at the margin and obovoid. There are two types of anthers: fertile stamens and three sterile stamens or staminodes. Fertile stamens are equal and glabrous, filaments are ca. 3 mm long, anthers are ca. 5 mm long, bi-porose at the apex and reflexed.

The anther is dehisced by apical slits, which open or close according to ambient humidity. Three sterile stamens, or staminodes, are 4 mm long, glabrous, and deeply cordate at both ends. The ovary is curved, 0.20 cm long, stigma up to 2.30 cm long, glabrous, stigma fringed with cilia. Style is bent down—the sickle-shaped pistil projects into the fertile stamens. The average number of pollen grains per anther is 6580 ± 5.20, which has moderate viability. Pods are pendulous, 17 to 25 x 1 to 1.50 cm long, shortly stipitate, linear-cylindric, 100 to 108 seeded, nearly terete, turgid, septate, and dehiscing along one margin. Seeds are orbicular, 4 to 6 x 3 to 5 mm, brown and rugulose. Floral morphology observations are detailed in Table 4.14. The dimensions of the floral parts of *S. spectabilis* are given in the plate 4.4.1a-e.

Table 4.14 Observations on floral characters of *Senna spectabilis*

Floral Characters	**Observations**
1. Flowering period	September to December
2. Flower colour	Rich yellow to Dark-veined
3. Odour	Present
4. Nectar	Present
5. No. of primary branch	16 ± 1.73
6. No. of inflorescence/branch	2262.75 ± 527.74
7. No. of flowers/inflorescence	120 to 140
8. Sepals/ flower	5
9. Petals/ flower	5
10. No. of anthers/ flower	7 fertile stamens,3 sterile staminodes
11. No. of pollen grains /anther	6580 ± 5.20
12. No. of ovules/ flower	80 to 120
13. Pollen/ ovule ratio	59.81
14. Length of stigma ± style (in cm)	02.35 ± 0.19
15. Length of ovary (in cm)	0.20
16. Anthesis time	06.00 to 09.00 hrs
17. Anther dehiscence time	8.00 to 12.00 hrs
18. Sugar concentration (in %)	4.11 ± 0.79
19. Pollen type	Tri-colporate
20. Pollen size	35.05 ± 2.19 µm
21. Stigma type	Above anther level
22. Fruit setting/inflorescence	10.55 ± 0.95
23. No. of seeds/pods	108.91 ± 9.69

Plate 4.4-1. a-e. Floral morphology (a- inflorescence, b-sepals, c-petals, d-stamen & carpel, e-ovules

Plate.4. 4-2.a-f. Some of the floral visitors(a-Formicidae, b- Dammar Bee, c- Housefly, d- Wasp Moth, e- Stink bug, f-Coptosoma)

Anthesis and Pollination: The duration of anthesis was from 6.00 to 9.00 hrs, and anther dehiscence started at 8.00 hrs and continued up to 12.00 hrs. The stigma became receptive at 8.00 hrs. The anthesis process is diurnal and sometimes asynchronous, meaning some flowers open completely by 10:00, while some start opening early. The flowers remain open until the next day, probably due to increasing temperature favouring the anthesis. The anthesis exhibited two days of positive stigmatic receptivity under this condition. The flowers open partially on the first day. Then, they gradually open fully and expose the sexual whorls for visitors. A fluid-like substance in the basal portion of the flower and tender floral parts of newly opened flowers were used for sugar concentration, and the mean nectar sugar concentration was 4.11 ± 0.79 brix; no distinct nectaries or extra floral nectaries were found. Extra floral nectar was absent in the case of *S. spectabilis* var. *excelsa*.

The peak arrival time of insect visitors was observed from 9.00 to 12.30 hrs. Dammar Bee is a significant visitor to *S. spectabilis,* while Violet carpenter bee is a regular visitor. Some Formicidae members, like Weaver Ant and Large Myrmicine Ant, are residents of the flowers of this species. They feed on the floral parts, like the tender petals and sepals, even during night hours. Rice Swift is an occasional visitor. Other visitors, such as Stink Bugs and Wasp Moths, came to consume the sap from tender pedicels and branches. The list of flower visitors is recorded (Table 4.15, Plate 4.4-2). The Indian stingless bee, a significant visitor, starts its nectar-foraging activity as early as the dawn hours, from 8.00 to 12.30 hrs, and resumes forging during the dusk hours, from 16.00 to 17.30 hrs. The Violet Carpenter Bee species foraged from 10.00 to 11.30 hrs in the morning. Dammar Bee, a persistent visitor, only visited open flowers. This foraging behaviour is thought to be boosting the chances of cross-pollination.

Breeding Systems: Studies carried out on artificial breeding experiments and observations of natural and open pollination showed that 20% of fruits were set in crossing experiments such as hand-geitonogamy, while 25% were set in hand-xenogamy and 20% of fruits in autogamy. Our tagged flowers' natural and open pollination set 30% of fruits (Table 4.16) (Plate 4.6).The fruit set per inflorescence in open pollination is 10.55 ± 0.96. The number of flowers per inflorescence is 114 ± 4.27. After observing 20 trees and their tagged uniform inflorescence, 10% of fruits were found to be finally maturing following the abortion of immature flowers, immature fruits and unripe fruits.

Plate.4.5. Artificial breeding experiments in studied locations of *S. spectabilis*

The examination of the futile percentage also demonstrates that 13.58% of opened flowers were lost, while 90.84% represents the final ripened pod futile percentage (Table 4.17). Despite these findings, the remaining 10% of ripened pods proved sufficient for additional dispersal mechanisms and the successful invasion of this particular tree species. The results of the breeding system indicated that the flowers are self-compatible and self-pollinating, and they also facilitate cross-pollination. As an out-crosser and a self-pollinating species, *S. spectabilis* has different reproduction methods in this invasion area.

Table 4.15 List of Flower foragers on *S. spectabilis*

Sl.No.	**Scientific Name**	**Common Name**	**Visiting status**
1.	*Tetragonula iridipennis* Smith	Dammar Bee	Regular
2.	*Xylocopa violaceae* L.	Violet carpenter bee	Regular
3.	*Amata huebneri* Boisdual	Wasp Moth	Occasional
4.	*Bocana manifestalis* Walker	-	Occasional
5.	*Camponotus mitis* Smith	Carpenter Ant	Regular
6.	*Myrmicaria brunnea* Saunders	-	Resident
7.	*Oecophylla smaragdina* Fabricius	Weaver Ant	Resident
8.	*Tapinoma melanocephalum* Fabricius	Ghost Ant	Occasional
9.	*Borbo cinnara* Wallace	Rice Swift	Occasional
10.	*Musca domestica* L.	Housefly	Occasional
11.	*Halyomorpha halys* Stal	Stink Bug	Occasional
12.	*Coptosoma* Laporte	-	Occasional

Table 4.16 Modes of breeding pattern in *Senna spectabilis*

Sl.No.	Treatments	n	No. of flowers		Fruit set (%)
			Pollinated	Set fruit	
1.	Autogamy	20	8	4	20
2.	Geitonogamy	20	11	4	20
3.	Xenogamy	20	9	5	25
4.	Apomixis	20	-	-	0
5.	Open pollination	20	16	6	30

Table 4.17 Flower and fruit set per inflorescence

Tree No.	Flower			Fruit-Pod		
	Bud	Young	Opened	Bud	Young	Ripened
1	140	124	120	76	24	12
2	138	137	121	68	16	15
3	139	128	114	59	17	10
4	132	130	116	72	20	13
5	128	119	114	60	28	12
Mean	135.4	127.6	117	57.4	21	12.4
Futile (%)		**5.7**	**13.58**	**57.6**	**84.49**	**90.84**

Pollen Viability: The present study reports on the viability and germination of fresh pollen grains of *S. spectabilis*. The viability of these pollen grains was determined to be 30% when stained with acetocarmine (1%). In vitro experiments were conducted to assess the germination of these pollen grains when dusted in different media. After 20 minutes of observation under the microscope, germination rates ranging from 32% to 100% were recorded (Table 4.18, Plate 4.5). The highest germination rate of 100% was observed in medium 1, while the lowest germination rate of 32% was observed in medium IV, which lacked sucrose. These findings suggest that the presence of sucrose in the medium is crucial for the germination of *S. spectabilis* pollen grains.

Table 4.18 Composition of the pollen germination media

Composition	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**
Sucrose (g)	10	10	10	0	10	5	5	5	5
Boric acid (g)	0.01	0.01	0	0.01	0	0.01	0.01	0	0
Calcium nitrate (g)	0.03	0	0.03	0.03	0	0.03	0	0.03	0
Distilled water (ml)	100	100	100	100	100	100	100	100	100
Germination %	100	91	72	32	64	75	71	54	44
Duration (min)	20	20	20	30	20	20	20	20	20

Information on floral characters and pollination systems is essential in the breeding system, especially in the case of *Senna spectabilis*, which poses a major threat and negatively impacts the structure and diversity of the forest and its ecosystem. In order to manage this species in the invaded forest areas, observation of reproductive biology is very important.

The anthesis time of this species is diurnal; in this case, flowers and flower buds are at different development stages on the same inflorescence as observed in *Sesbania virgata* (Cav.) Pers. Apart from this, extended flower opening was found to favour pollinator activity throughout the day (Souza 2016). Here, the *S. spectabilis* flowers remained open until the next day. The implementation of this particular strategy has the potential to facilitate cross-pollination by rendering pollen accessible as a resource across multiple plants and flowers, thereby enabling flower visitors to engage in simultaneous pollination activities.Fabaceae types of flowers exhibit specific and highly efficient pollination mechanisms with their vectors. Different biotic vectors, such as bees and birds, were reported in Fabaceae flowers.

The present study found that the significant pollen vectors are the Dammar Bee, a widespread species in India, and the Violet Carpenter Bee. These species are confirmed as pollinators based on pollen load and seed setting (Rasmussen, 2013).

Studies showed that in the genus, *Senna* pollen-collecting bees extract pollen by vibrating the middle feeding stamens, which they clasp with their legs (Marazzi and Endress 2008).

This species has poricidal dehiscence of anthers, minute terminal stigmas and curved styles. Pollens are released when bees vibrate anthers (Buchmann 1974). These floral features showed that this species has buzz pollination syndrome.

According to Marazzi *et al*., (2013), when they studied the diversity and evolution of a trait mediating ant-plant interactions relating to extra floral nectaries in *Senna* (Leguminosae), they excluded *S. spectabilis* because they did not observe any ants around floral buds or leaves in that field. During field observation, Formicidae species were spotted abundantly and acted as residents of these flowers. They were observed to be feeding on tender floral parts. Both diurnal and nocturnal activities were observed. A moth species, *Bocana manifestalis,* was observed on the flower at night.

According to Almeida *et al*., (2015) *S. spectabilis* is listed as an Enantiostylous type of species. They classified Cassiinae species into seven types based on morph distribution among plants and grouped species with different flower morphologies and diverse reproductive strategies of these types. *S. spectabilis* comes under Type five, which is the Amicella group. *Chamaecrista amiciella* is the model species. The observed pattern in these species entails the deposition of pollen grains on the dorsal

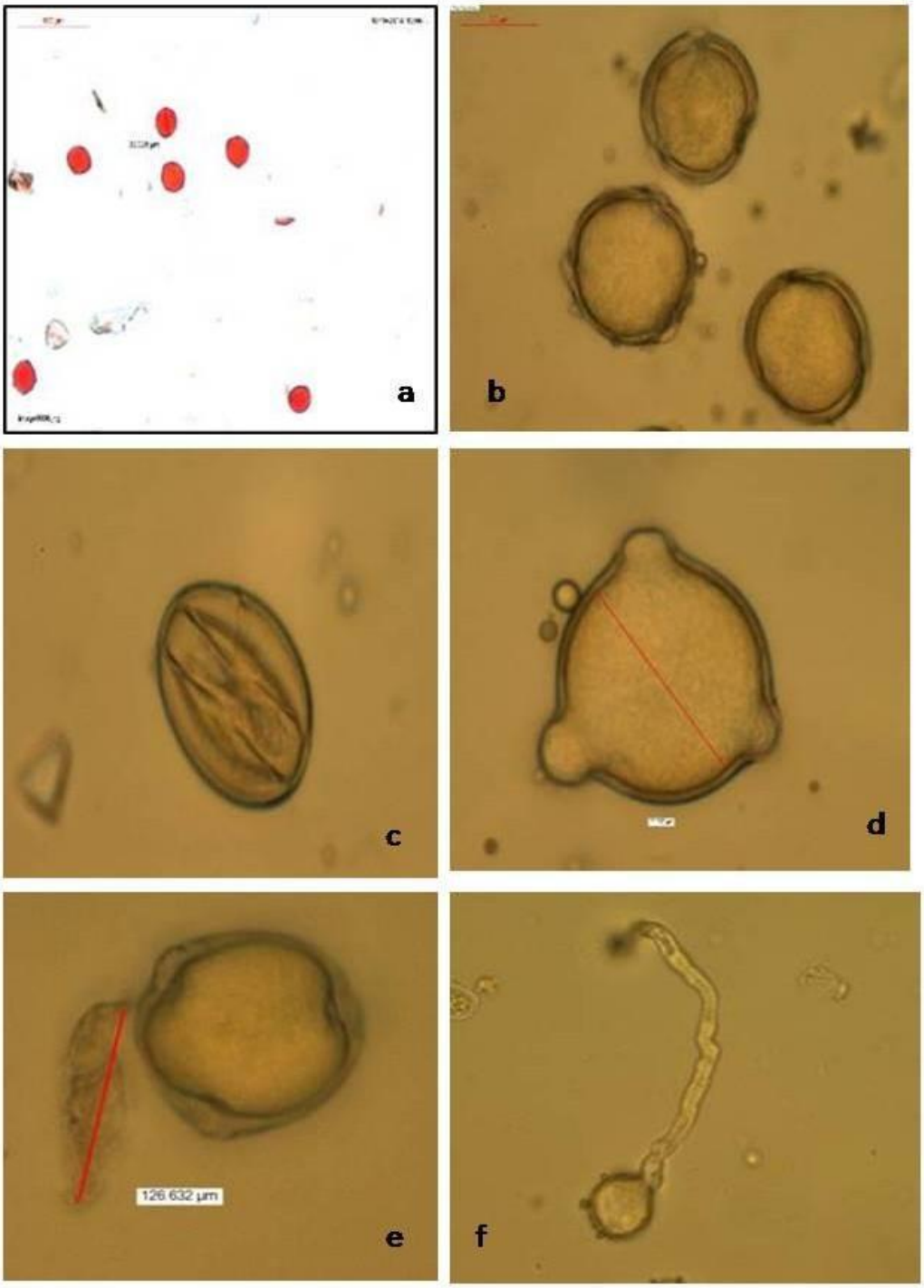


Plate.4.6.a-f. Stages of Pollen Germination (a: Viable pollen, b to d: Pollen germination, e and f: Pollen tube development)

portion of the pollinator subsequent to their traversal through all the extensions of a modified, tube-shaped petal (Almeida *et al*., 2013). The pollen produced by the anthers involved in pollination is deposited in a position opposite to the stigma. The Amicella type is considered the second most prevalent, as it exhibits the same mechanisms observed in the Ramosa type (type seven), with the exception of employing a cluster of petals (whereas only one petal fulfills this function). This particular type is unique to Chamaecrista and Senna species.

In the case of *S. spectabilis,* the body washing of a dammer bee results in both the dorsal and ventral sides having deposited with pollen grains. The number of pollen grains is higher on the ventral side. Pollen is the most sought-after floral reward. It is a vital food for many insects, especially Apidae, beetles, flies, thrips springtails, and some orthopteroids and butterflies (Anderson 1996). Pollen is highly nutritive and contains essential and quasi-essential amino acids (Haydak 1970). In the case of *S. spectabilis*, pollen is also the primary reward because the nectar-sugar concentration is deficient and proper nectar secretion was absent in this flower (Table 4.19).

Table 4.19 Nectar sugar concentration in *Senna spectabilis*

Time of testing	05.30	06.00	07.00	08.00	09.00	10.00	11.00	12.00	14.00
Brix %	3.02± 0.14	3.30± 0.57	3.58± 0.40	4.32± 0.34	4.90 ± 0.26	5.00 ± 0.12	5.10 ± 0.17	4.38± 0.57	3.42± 0.86

Tamnet (2011) studied how to optimise the preservation of pollen grain germination of *S. spectabilis*. For the study, they selected this invasive tree species, a large species of bee flora facing extinction in the Adamawa region of northern Cameroon. They claimed to have conducted the study to help beekeepers. They investigate the in vitro germination and storage of pollen. The findings demonstrate that the pollen exhibits a preference for germination, with a rate of up to 38.36%, when cultivated in Brewbaker medium supplemented with an optimal concentration of 25% sucrose. Furthermore, the pollen samples were subjected to storage conditions at temperatures of 10°C and 20°C, and their germination length was assessed over a period of 22 weeks.

In vitro germination was found to be good in the present study, and 32 to 100% germination was found in different media, also proven in the experiments (Table 4.18). During field observations for pollinator interactions, the Indian honey bee (*Apiscerana indica*) was consistently found to be hovering around the flowers of *S. spectabilis* and visiting only the associated plants. However, it never made a single visit to *S. spectabilis* flowers. Further observations and research experiments are required to determine the reason behind it, as this could be due to a lack of sufficient forage or any repellent factors. It also possesses a self-pollination mechanism. Autogamy is a reproductive characteristic of invasive and pioneer species that occupy clearings and forest edges (Williamson, 1996, Holsinger, 2000). Here, the case of *S. spectabilis* occurred in areas similar to clearings, such as massive bamboo flowering in open areas, other open areas of deciduous forest patches and the edges of Vayal ecosystems. In breeding experiments, 25 to 20% of fruit sets occurred, and autogamy also accounts for 20% of fruit sets. It reveals that *S. spectabilis* possesses a mixed reproductive system composed of cross-pollination and autogamy. This system is probably related to its success as an invasive species, which helps it spread and colonise new habitats.

Baker and Baker (1979) conducted an observation which revealed that the optimal balance between self-compatibility and cross-pollination confers advantages to weeds. The author asserts that the establishment of a new population, following the dispersal of a seed to a distant location, is contingent upon the self-pollination capacity of the species. *S. spectabilis*, for instance, exhibits both autogamous and outcrossing characteristics, which appears to be a favorable strategy when coupled with its ability to colonize degraded lands, such as forest areas. Numerous invasive plant species have been documented as self-compatible in their introduced ranges (Rambuda and Johnson 2004, Kleunen and Johnson 2007, Stout 2007,Rodger *et al.*,2010, Hao *et al.,* 2011) and this has been posited as a contributing factor to their successful invasion (Williamson and Fitter, 1996, Pannel and Barret, 1998).

Invasive species typically exhibit a notable sexual reproductive capacity, the potential for asexual reproduction, the ability to progress from seed to sexual maturity, efficient dispersal and colonization capabilities, a considerable tolerance towards environmental heterogeneity and disturbances, a heightened adaptation to

environmental stress, and a superior competitive capacity compared to native species (Sakai *et al.,* 2001; Vila and Weiner, 2004; Werner *et al.*, 2009).

The scholarly article by Winkler *et al.,* (2019) regarding the invasive Sahara mustard (*Brassica tournefortii*) has the potential to enhance invasive species' proliferation. In the case of invasive species, the attribute of self-fertilization is deemed advantageous as it enables a solitary organism to initiate a new population without relying on other individuals for reproduction. Consequently, self-fertilizing invasive species are perceived as notably challenging to manage due to their exceptional efficacy in disseminating and establishing themselves in novel territories.

As an invasive tree species in Wayanad Wildlife Sanctuary forest areas, forest officials and locals try to eradicate this species by cutting the tree. However, the tree re-sprouts profusely. During five years, this tree was observed to have grown more branches after re-sprouting, while each branch produced flowers vigorously in three years. Re-sprouting ability is a positive reflection of its invasiveness.

Research conducted on invasive Australian *Acacias* by Milton and Hall, 1981, elucidated that this species possesses various reproductive characteristics that potentially contribute to their invasiveness. These traits include extensive and enduring floral displays, pollination syndromes that cater to a wide range of pollinators, early production of a substantial quantity of long-living and highly viable seeds, leading to the formation of extensive seed banks, adaptations for seed dispersal, and mass germination. These findings were also observed in *S. spectabilis*, which displayed comparable behaviour and responses.

The study revealed that the high rate of seed production in *S. Spectabilis* can be attributed to various factors, including the pollen viability and vigour of the pollen tube, the timing of anther dehiscence and stigma receptivity, the presence of multiple pollinators, and adequate pollen rewards. The pods of *S. Spectabilis* were observed to contain an average of 108.91 ± 9.69 seeds. Notably, the plant exhibited no sexual incompatibility or pollination difficulties. The reproductive syndrome of *S. Spectabilis* is conducive to achieving maximum fertilization.

4.2.3 Seed germination and seed dispersal characters of *S. spectabilis*

S. spectabilis generates substantial quantities of seeds that exhibit a remarkable longevity of up to three years and a notable resistance to mortality. This characteristic enables the species to proliferate rapidly, as the pods rupture and scatter the seeds upon detachment from the plant, and can further disseminate through water channels (PIER, 2014).

Seed Viability

Viability was calculated by conducting rapid cutting tests with the help of seed cutters manually using the seed collected from Wayand, Anaikkaty and Thiruvanathapuram, Kerala. Eight seed lots were directed to cuttings tests, and seed viability ranged between 79% and 87.2, and an average of 84.45% viable seeds were recorded (Table 4.20).

Table 4.20 Seed viability (Cutting test)

Sample	No. of Seeds	No. of viable seeds	Viable seeds (%)	Empty seeds (%)
1	1000	860	86	14
2	1000	850	85	15
3	1000	790	79	21
4	1000	847	84.7	15.3
5	1000	890	89	11
6	1000	795	79.5	20.5
7	1000	852	85.2	14.8
8	1000	872	87.2	12.8
Mean	1000	844.5	84.45	15.55

Pre-Sowing Treatments and Germination Experiments

The germinability of the seeds was determined through the implementation of germination trials. The results obtained from these experiments indicate that initial germination takes place seven days after sowing, while germination concludes - days after sowing (as shown in Table 4.21;Plates.4.6 and 4.7). The germination process spans 7 to 20 days after sowing, requiring a week to complete. The highest germination rate is

observed on the first day of germination (7th day), reaching 76%, and gradually decreases until the 20th day, when the minimum germination rate of 96% is recorded. The maximum germination rate observed in the experiment is 96%. Throughout the germination period, there is a noticeable decline in daily germination from the first to the final day.

In a controlled laboratory setting, it was observed that *S. Spectabilis* displayed a dormancy imposed by the seed coat, resulting in delayed germination. Based on this observation, the effects of various pre-treatment methods on the germination of *S. spectabilis* seeds were evaluated. Six pre-sowing treatments, including a control group, were tested. Among the different pre-sowing treatments, the seeds that were not subjected to any treatment (control group - sown in a plastic tray and placed in a mist chamber with intermittent watering) exhibited only 36% final germination.

The pre-sowing treatments yielded a maximum germination rate of 96% when the seeds were soaked in 2M H_2SO_4. It was followed by nicking the seeds and soaking them in water for 24 hours, resulting in a 90% germination rate. Additionally, seeds soaked in boiled water and allowed to cool for 48 hours exhibited an 88% germination rate. These seeds were collected from different populations in Kerala. Recent studies have supported these findings and conclusions. Prajitha and Sudhabai (2022) observed that *S. spectabilis* seeds from the Muthanga in Wayanad Wildlife Sanctuary, which are considered orthodox, germinate more quickly in forest soil compared to laboratory conditions. Without any dormancy treatments, the seeds fail to germinate, and concentrated acid treatment or manual scarification is required for germination.

A study conducted in Malawi by Zembele and Ngulubeto (2022) also evaluated the effects of pre-treatment methods on the germination of *Senna spectabilis* seeds. It has been determined that the seeds exhibit a dormancy state enforced by the seed coat, leading to a postponement of the germination process. The treatment method identified as the most efficacious involves nicking the seed coat followed by a 24-hour soaking in hot water.

Table 4.21 Pre-sowing treatments germination percentage of *Senna spectabilis*

Sl. No.	Treatments	Germination medium and scarification-Vermiculate28°C(+/-2), 100% humidity constant with 12h light and 12h dark	Initial ger. day	Final ger. day	Ger.mination%
1	Control		12	18	36 ± 0.00
2	The seeds were subjected to a 24- hour water soaking treatment		16	21	42 ± 0.00
3	The seeds were subjected to a 48- hour water soaking treatment		19	23	46 ± 0.58
4	Soaking in 2M H_2SO_4.		38	48	96 ± 0.58
5	The seed was immersed in water that had been boild and subsequently left to cool for duration of 48 hours.		34	44	88 ± 0.58
6	Nicking &soaked in water for 24 hrs		36	45	90 ± 0.58

4.2.3.1. Seed Predation and Seed Dispersal Mechanisms

Dispersal of seeds by herbivores was observed directly and through opportunistic data collection: The droppings containing seeds of *S. spectabilis* were randomly selected and collected. These droppings were then separated, and the seeds were counted after being placed on tissue paper. It was observed that herbivores such as Sambar deer, elephants, and Chital either consumed or regurgitated the dry pods of *S. spectabilis*. The density of *S. spectabilis* seeds in the faeces differed significantly among these dispersers. The Sambar deer (*Rusa unicolor)* had the highest faecal seed density at 36%, followed by Asian elephants (*Elephas maximus*) at 16%, and Chitals (*Axis axis)* at 8% in Wayanad Wildlife Sanctuary (Table 4.22). Terrestrial herbivores play a significant role in the dispersal of S. spectabilis in Wayanad Wildlife Sanctuary.

Fate of Seeds: To ascertain the fate of seeds, all fruits that had fallen beneath the tree canopy were permitted to undergo natural scarification, which resulted in regeneration within 4 to 8 months. As a result, barochory was predominantly observed in the study plots.Occasionally, herbivores would consume the desiccated pods of *S. spectabilis* and transport the seeds to unexplored regions within the forest ecosystem, leading to new populations in patches, which have been documented in Wayanad Wildlife Sanctuary.

Table 4.22 Observed faecal samples on various herbivores in Wayanad WLS

Sl. No.	Name of the species	No. of samples	Samples with seeds	Seeds
1.	Sambar deer	50	18	400 ± 5
2.	Asian elephant	50	8	120 ± 6
3.	Chital	50	4	34 ± 2
4.	Indian hare	50	0	0

The fruits of *S. spectabilis* exhibit a long, cylindrical shape and hang down from the plant in pendulous pods. These pods contain brown seeds, as documented by Satyanarayana and Gnanasekaran (2013). These pods' taste, nutrient composition, and aroma likely played a significant role in attracting herbivores. Previous research has indicated that the invasion of *S. spectabilis* in certain areas has resulted in a scarcity of forage for herbivores, leading them to adapt to consuming the seeds of *S. spectabilis* due to their abundant availability.

A study conducted by Anoop *et al.*, (2021) has revealed that the Asian elephant plays a crucial role in the distribution of *S. spectabilis* within the Wayanad Wildlife Sanctuary (WLS). The researchers have observed that the fruiting season of *S. spectabilis* coincides with the peak density of elephants in Wayanad, as these animals migrate to this region during the summer months (March to May) from other parts of the Nilgiri Biosphere Reserve.

In this study, samples were collected during various periods of fruiting phenology, and the findings indicate that the Sambar deer is the most efficient seed disperser of *S. spectabilis*, effectively moving the seeds away from the tree crown. Successful germination of seedlings occurred in the distinct faecal pellets of Sambar deer (Plate.4.7.d) and elephants, mainly when they were located away from the *S. spectabilis* tree crown. This phenomenon may have contributed to the widespread proliferation of *S. spectabilis* within the forest ecosystem. The observations also revealed that both barochory and terrestrial animals played a role in the dispersal mechanisms of *S. spectabilis* in Wayanad Wildlife Sanctuary. However, no seeds were found in the faecal samples of the Indian hare (*Lepus nigricollis*), commonly observed in the areas invaded by *S. spectabilis*. The promotion of naturalization success in invasive plant species has been significantly influenced by increased germination rates and seedling recruitment, as stated by Udo *et al.,* (2017).

Plate.4.7.a. Seeds of *Senna spectabilis*

Plate 4.7.b. Sowing the seeds of *S.spectabilis*

Plate.4.7.c. Foraging of herbivores in *Senna spectablis* invaded areas at Wayanad

Plate.4.7.d. Sambar deer at Muthanga forest area; *S.Spectabilis* seedlings germinated in the segregated fecal pellets of Sambar deer

4.2.4. Study on Morphological variations of *Senna spectabilis* in different populations in Kerala

Population variation is analysed worldwide by examining phenotypic and molecular characteristics. Scholars have chosen various characters to analyze variation using phenotypic characteristics. The success of invasive plants is attributed to their morphological traits. This study aimed to test the hypothesis that genetic shifts in morphological traits or any adaptive variation have occurred in the invasive populations of *S. spectabilis* in Kerala.

4.2.4.1. Estimation of Morphological variability of *Senna spectabilis* seedlings in different populations in Kerala

Seedlings were cultivated in the KFRI Mist chamber, utilizing seeds collected from seven populations of *S.spectabilis* in Kerala such as Muthanga, Vythiri, Tholpetti, Meppadi, Azhinjilam, Thiruvanathapuram, Anikkaty(Plate.4.8). Our investigation focused on a range of functional traits, including SH (Shoot height), RL (Root length), RCD (Root collar diameter), LL (Leaf length), LW (Leaf width), RN (Root number), TN (Twig number), and CC (Chlorophyll content). These characteristics, frequently observed in invasive plant species, may be indicative of their intrinsic attributes acquired through prior evolution in their original habitat before introduction, or as a result of new adaptations in response to evolutionary pressures aimed at evading natural adversaries in the introduced environment (Callaway and Aschehoug, 2000; Leger and Rice, 2003; Erfmeier and Bruelheide, 2005; Güsewell, Jakobs and Weber, 2006).

Analysis of variance revealed significant differences ($p<0.01$) among populations for the seedling growth characteristics of *S. spectabilis* (Table 4.23). Considering the six-month seedlings, seedling height varied significantly among populations with a range of 16.4-17.93 cm. The maximum seedling height was observed for the Anikkaty population (17.29), followed by Thiruvanathapuram (17.13cm). The average root length and root collar diameter were 13.08cm and 0.28 cm between the population. The highest root collar diameter was noted for Meppadi, followed by Tholpatty and Azhinjilam, whereas the lowest value for this character was observed for Vythiri (0.24). Regarding the leaf length, the values ranged from 3.87–3.64 cm among the populations.

Plate.4.8. *Senna spectabilis* seedlings of different populations in Kerala

Variation in the Chlorophyll content of leaves was observed among the seven populations of six-month seeedlings, ranging from 16.2 to 18. In theory, chlorophyll is a crucial photosynthetic pigment in plants significantly impacted by environmental factors. The primary objective of most investigations concerning Chlorophyll content in natural communities has been to establish a correlation between chlorophyll and the functioning of ecosystems since chlorophyll significantly influences the photosynthetic capacity of leaves (Singsaas *et al.,* 2004).

In 12-month seedlings, traits such as leaf width (LW) and twig number (TN) show no significant difference among the populations. The trait, such as shoot height, ranges from 28.9 cm to 30.96 cm between the studied populations. The height of a plant, which is a significant morphological characteristic, exhibits a strong correlation with its life span and time to maturity and is regarded as a surrogate for competitive ability (Moles et al., 2009; Pérez-Harguindeguy *et al.,* 2013). Liu *et al.* (1993) observed that shoot growth and diameter growth of seedlings are more a consequence of evolved genetic responses to environmental stimuli.

Comparing the performance of *S. spectabilis* seedlings, six-month and 12-month seedlings (Figure 4.29) had significantly higher increments in all characters except the chlorophyll content.

Table 4.23 Morphological Variations of *S. spectabilis* seedlings growth parameters in different locations of Kerala

		MUT	VYT	THO	AZH	MEP	ANA	THI	Mean	FValue	P Value
6 MONTHS	SH	16.4	16.2	16.91	15.86	16.16	17.29	17.13	**16.56**	30.763	**0.000**
	RL	12.6	13.1	12.74	11.83	13.72	14.02	13.56	**13.08**	49.772	**0.000**
	RCD	0.28	0.24	0.29	0.29	0.30	0.26	0.28	**0.28**	9.782	**0.000**
	LL	3.65	3.76	3.64	3.87	3.68	3.84	3.75	**3.74**	4.030	**0.002**
	LW	0.93	0.93	0.97	0.94	0.99	0.97	0.92	**0.95**	14.962	**0.000**
	RN	13.6	14.2	13.4	14.9	13.5	13.0	13.3	**13.70**	2.962	**0.013**
	TN	0	0	0	0	0	0	0	**0.00**		
	CC	17.0	17.5	18.0	20.4	18.0	16.2	17.6	**17.81**	7.372	**0.000**
12 MONTHS	SH	32.4	31.7	32.34	28.9	31.8	32.96	30.93	**31.58**	235.326	**0.000**
	RL	30.1	25.7	29.96	26.41	27.53	27.2	28.84	**27.96**	121.771	**0**
	RCD	0.58	0.57	0.52	0.52	0.54	0.48	0.50	**0.53**	94.727	**0.000**
	LL	4.29	4.31	4.77	4.80	4.96	4.60	4.21	**4.56**	90.695	**0.000**
	LW	1.05	1.06	1.09	0.98	1.04	1.09	1.01	**1.05**	1.075	**0.387**
	RN	29.2	28.6	29.8	30.5	31.1	32.2	31.8	**30.46**	8.643	**0.000**
	TN	1.4	1.5	1.5	1.3	1.8	1.6	1.4	**1.50**	1.057	**0.398**
	CC	22.6	18.4	18.96	21.94	19.15	17.24	18.37	**19.52**	14.667	**0.000**

SH:Shoot height,RL:Root length,RCD:Root collar diameter,LL:Leaf length,LW:Leaf width,RN:Root number,TN:Twig number, CC: Chlorophyl content

Correlation matrix of seedlings characters of *S. spectabilis*

Correlation analysis was carried out using IBM SPSS statistics software (version26: 2018) to estimate the relationship and strength between the characters used for variation studies. The correlation analysis using Palaeontological Statistics (PAST) software graphically represents the relationship between the tree characters (Table.4.24, Figure.4.29). A positive correlation is displayed in blue colour dots, while a negative correlation is shown in red dots, and the dots' denseness indicates the strong relationship between the characters. The strength of association between the characters is represented in the form of the denseness of the beads. More dense bubbles show a strong bond, and low, dense dots show a weak relationship between the characters.

As per pearson linear correlation analysis of the characters of six-month-old *S. spectabilis* seedlings, few show a significant correlation between the characters, and twelve-month-old *S. spectabilis* seedlings also show a significant correlation between the characters. The correlation coefficient (*r*), along with all probable combinations of the variable of six months and twelve months, was summarized in Table (4.24 and 4.25). Among the seven characters of six-month-old *S. spectabilis* seedlings, root length significantly correlates with shoot height (0.62) and leaf width (0.07).Root collar diameter correlates significantly with leaf width (0.40) and chlorophyll content (0.47). Leaf length positively correlates with root number (0.41) and chlorophyll content (.30), and root number correlates with chlorophyll content (0.83).

Correlation analysis of the characters of twelve-month-old *S.spectabilis* seedlings (Table 4.25, Figure 4.30) shows a significant correlation between the characters. Among the seven characters, Shoot height shows a significant positive correlation with leaf width (0.92), root length (0.39) and twig number (0.55). Root collar diameter shows a significant correlation with chlorophyll content (0.55). Leaf length positively correlates with twig number (0.48), root number (0.23) and leaf width (0.01). Leaf width (0.48) and root number (0.27) correlate with twig number.

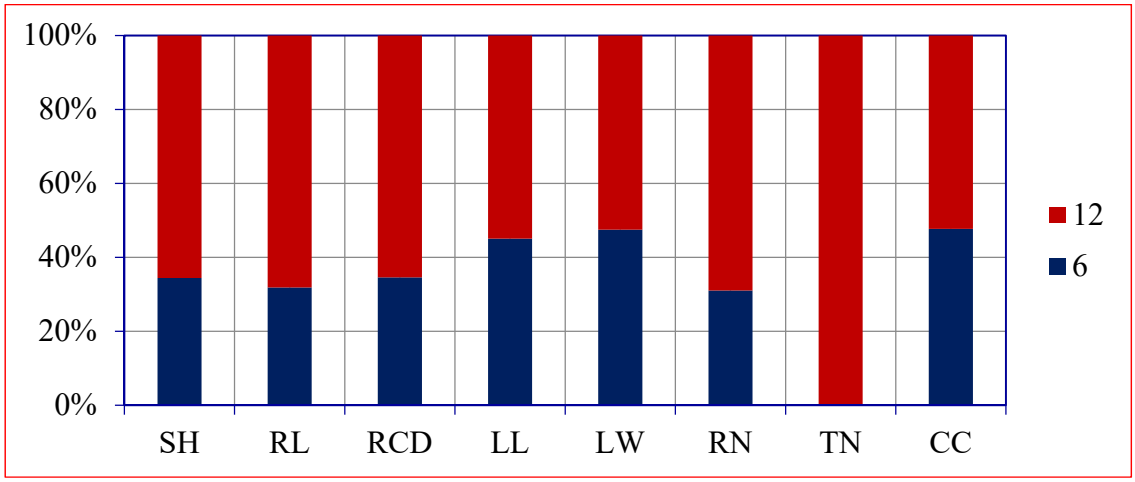
In the current investigation of *S. spectabilis*, a notable disparity in seedling growth was observed. This discrepancy is likely to be attributed to either the genetic makeup or the distinct genotypes present within its diverse populations. This finding aligns with the research conducted by Azad *et al.,* (2011) who similarly reported a

significant divergence in growth patterns among Acacia auriculiformis during the nursery phase. Furthermore, Zobel (1992) expounded upon the notion that approximately 30% of the plant genome is dedicated to the intricate processes of seedling growth and development.

A seedling's successful growth and development significantly enhances the likelihood of opportunist species or plant invaders effectively establishing themselves in new regions (Skálová *et al.,* 2012). Consequently, comprehending the crucial factors that can potentially impact seedling performance is of utmost significance. The variability in this performance may arise from genetic variation (Biere, 1991).

The current study has revealed variation in the germination and seedling growth characteristics of the offspring that were grown from seeds collected from different populations. This finding is consistent with previous research conducted on various species, such as another tree invader, *Prosopis juliflora* (Goel *et al.,* 1997), *Prosopis cineraria* (Manga and Sen, 1998), *Acacia nilotica* (Ginwal *et al.,* 1995), *Melia azedarach* (Thakur and Thakur, 2015), *Tectona grandis* (Gunaga *et al.,* 2010), *Grewia optiva* (Jaswal, 1992).The aforementioned studies have also documented heterogeneity in the germination and seedling growth characteristics of offspring from trees that are situated in diverse geographical regions, each with its own unique environmental circumstances. This variation can be attributed to the genetic composition of the parent trees and their offspring, as these trees originate from seedlings and are expected to exhibit heterozygosity.

Fig. 4.29 Performance percentage of *S. spectabilis* seedlings growth parameters in different locations of Kerala, 6-month and 12-month seedlings.



SH:Shoot height,RL:Root length,RCD:Root collar diameter,LL:Leaflength,LW:Leaf width,RN:Root number,TN:Twig number,CC: Chlorophyl content

Table 4.24 Pearson Linear R Correlation statistics of six-month seedling characters of *S. spectabilis* populations in Kerala

	RL	**RCD**	**LL**	**LW**	**RN**	**TN**	**CC**
SH	0.62	-0.17	-0.05	0.07	-0.84	0.00	-0.68
RL		-0.24	-0.06	0.37	-0.79	0.00	-0.75
RCD			-0.33	0.40	-0.01	0.00	0.47
LL				-0.18	0.41	0.00	0.30
LW					-0.35	0.00	-0.09
RN						0	0.83
TN							0

Fig. 4.30 Linear r correlation statistics of six-month seedling characters of *S.spectabilis* populations in Kerala (exclude lower triangle; significance p>0.05 blank)

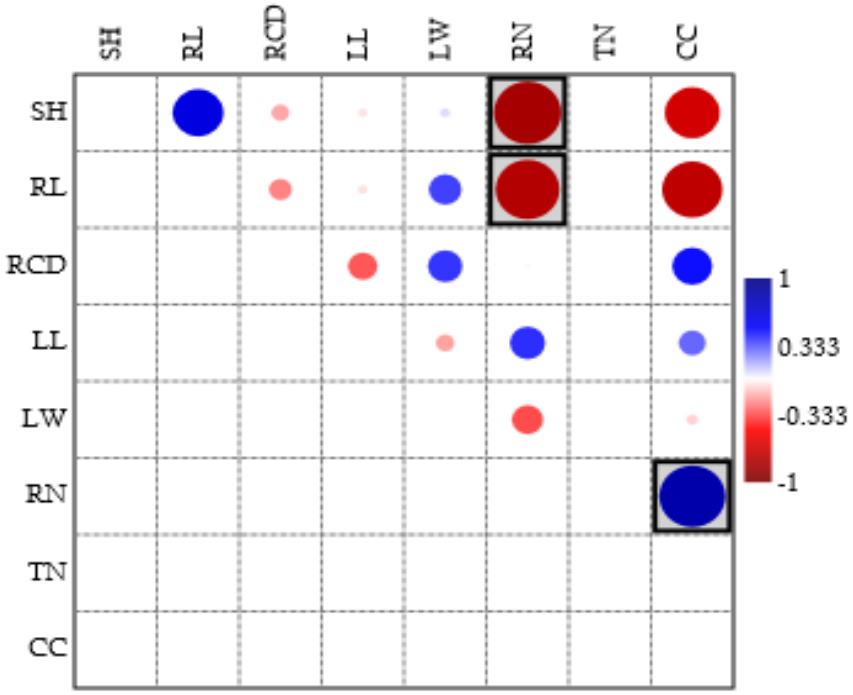
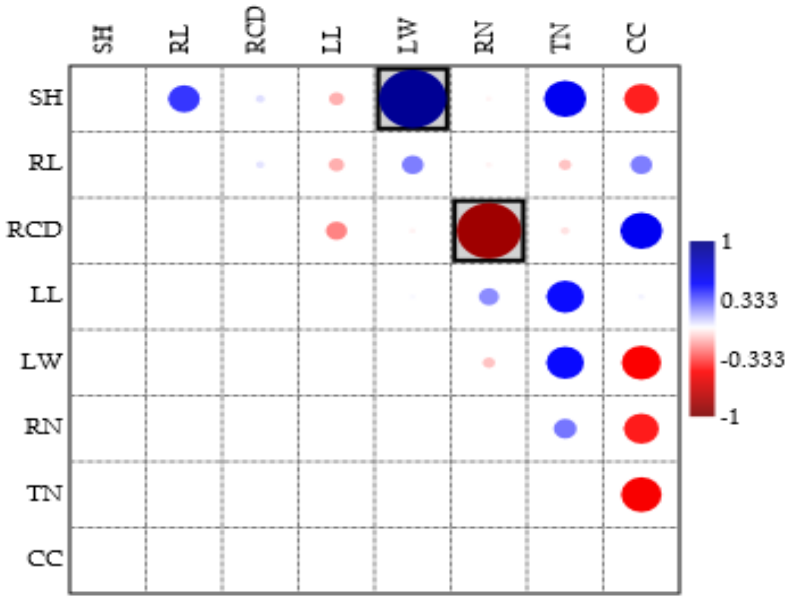


Table 4.25 Pearson Linear R Correlation statistics of 12-month seedling characters of *S. spectabilis* populations in Kerala

	RL	**RCD**	**LL**	**LW**	**RN**	**TN**	**CC**
SH	0.39	0.06	-0.15	0.92	-0.02	0.55	-0.44
RL		0.05	-0.16	0.25	-0.02	-0.12	0.25
RCD			-0.24	-0.02	-0.87	-0.06	0.55
LL				0.01	0.23	0.48	0.02
LW					-0.12	0.48	-0.51
RN						0.27	-0.45
TN							-0.52

Fig. 4.31 Linear r correlation statistics of twelve-month seedling characters of *S. spectabilis* populations in Kerala (exclude lower triangle; significance p>0.05 blank)



4.2.4. Cluster Analysis

The multivariate classical clustering of seedlings' morphological characters led to a hierarchical clustering of seven populations of *S. spectabilis* in Kerala (Fig.4.31). These populations were initially grouped into three clusters. The first cluster was further divided into two, with the populations of Muthanga forming a single branch and Vythiri and Tholpetty clustering together in another branch. In the second branch, Thiruvananthapuram was grouped, while in the third branch, Azhinjilam, Meppadi, and Anikkatti populations were clustered based on the selected characters.

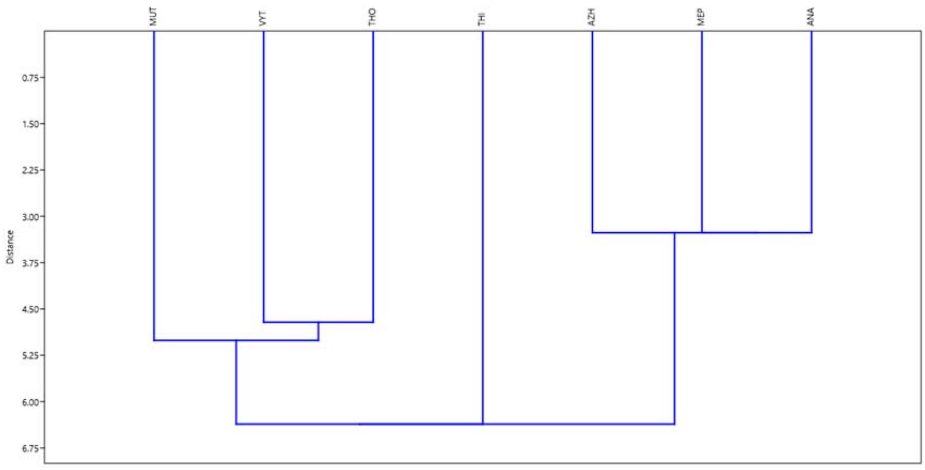


Fig. 4.32 Paired Group UPGMA algorithm of seedlings characters of *S. spectabilis* in Kerala. (Y-axis: Euclidean distance; X-axis: Populations).

4.2.4.2. Morphological Variations among the different tree populations of *S. spectabilis* in Kerala

This study encompassed an analysis of variations among the eleven populations of *S. spectabilis* in Kerala, focusing on morphological characteristics. These characteristics included girth at breast height (GBH), total tree height (HT), crown area (CA), number of primary branches (PB), number of secondary branches (SB), bark thickness (BT), sapwood moisture content (SM%), leaf chlorophyll content (LC), leaf area (LA), leaf fresh mass (LF), leaf dry mass (LD), diameter of the pods (DP), seeds per pods (SP), germination percentage (GM%), and viability percentage (VP).

The girth at breast height and tree height exhibited a range of 58.3 cm to 40 cm (with a mean of 48.19 m) and 13.42 m to 8.2 m (with a mean of 11.52 m), respectively, as depicted in figures 4.33.a-p, and table 4.26 and 4.27. The crown area of the individuals ranged between 88.7 m^2 and 35.3 m^2 (with a mean of 63.65 m2). The primary and secondary branches ranged between 2 and 9 (with a mean of 5.21) and between 6 and 32 (with a mean of 16.54), respectively, as illustrated in figures 4.33.a-p and Table 4.26 and 4.27.

The wood characteristics, such as bark thickness and sapwood moisture content, exhibited a range of 0.62 cm to 1.05 cm ($\overline{x}$= 0.76 cm) and between 50.8% and 73.7% ($\overline{x}$= 60.16%), respectively, as shown in figures 4.33.a-p, and table 4.26 and 4.27. The leaf characteristics, including chlorophyll content, area, fresh mass, and dry mass, ranged between 19.12 and 22.3 ($\overline{x}$= 21), 318 cm2 and 364 cm^2 (with a mean of 342.18 cm^2), 16 g and 21.6 g ($\overline{x}$= 19.62 g), and 3.53 g and 4.44 g ($\overline{x}$= 4.02 g), respectively, as depicted in figures 4.33.a-p, and table 4.26 and 4.27. The pod characteristics, such as the length and diameter of pods and seeds per pod, exhibited a range of 29.45 m to 31.11 m ($\overline{x}$= 30.52 m), 1.19 cm to 1.75 cm ($\overline{x}$= 1.38 cm), and between 110.7 and 117.2 ($\overline{x}$= 114.72), respectively, as shown in figure 4.33.a-p, table 4.26 and 4.27. The germination and viability percentage of seeds in the different studied populations ranged from 87.5% to 93.9% ($\overline{x}$= 91.41%) and between 93.69% and 99.63% ($\overline{x}$= 98.92%), respectively.

Sixteen tree characteristics were measured using standard methods and techniques from various tree populations, and the resulting data were tabulated in Microsoft Excel. Subsequently, the minimum and maximum values, mean, standard error, variance,

standard deviation, median, and coefficient of variation of all the tree characteristics were calculated and presented in a table 4.27. This study has contributed to documenting the mean values of *S. spectabilis* characteristics in its exotic invaded range. The variance, standard variance, and median values varied across different tree characteristics. The coefficient of variation indicated significant variation in tree characteristics among the populations, as shown in the table 4.27.

Avenue populations like Vythiri, Begur, and Periyar are characterized by a larger crown area than invaded populations like Muthanga, Ponkuzhy, and Meppadi. This difference may be attributed to the higher density of these populations in invaded regions. Vythiri and Azhinjilam populations have the highest bark thickness, followed by Muthnaga populations. Tholpetty and Begur populations have the highest number of primary and secondary branches. The sapwood moisture content is highest in Muthnaga populations, followed by Tholpetty. Attappady populations, such as Anaikkaty and Kottatthara, followed by Vythiri, exhibit the maximum leaf area. Leaf area is associated with light availability photosynthetic capacity and indicates soil resource availability and nutrient cycling.

Regarding fruit and seed characteristics, Muthanga and Tholpetty populations have the maximum number of seeds per pod. The Muthanga populations exhibit the highest germination and viability percentages of seeds. The number of seeds per capsule serves as an indicator of successful pollination and the allocation of resources towards reproductive output.

Table 4.26 Population-wise tree characters of *Senna spectabilis* in Kerala

Mean Value	GBH In Cm	HT In m	CA In m^2	PB In No.	SB In No.	BT In g	LA In cm^2	LF In g	LD In g	LC	SM In %	LP In Cm	DP In Cm	SP In No.	GM%	VP
PO	49.1	12	66.8	6.3	16.2	0.61	331	18.5	3.73	22.3	59.4	31.06	1.4	115.9	91.6	99.69
MU	48.1	9.45	50.2	4.7	17.9	0.83	335	18.4	3.56	21.6	73.7	32.1	1.19	117.8	93.9	99.15
VY	50.6	13.8	87.7	5.3	19.7	1.04	356	20.8	4.44	20.5	50.4	30.65	1.33	115.5	92.1	98.79
TH	50	11.6	54.8	7.1	25.4	0.67	332	20.8	4.4	21.4	70	30.49	1.2	116.6	91.7	98.46
BE	46.1	12	83.8	7.2	23.3	0.78	324	19	3.73	20.7	47.6	30.47	1.25	111	91	99.42
AZ	54.8	13.5	84.2	3.5	8.3	1.05	350	21.6	4.66	21.8	48.7	29.63	1.29	117.2	93.2	99.2
ME	58.3	10.3	40.5	5.11	12	0.62	341	21.6	4.11	20.9	69	31.11	1.38	110.7	93.6	98.53
AN	40	13.4	44.4	5.3	17.6	0.67	364	17.2	3.53	20.7	62.1	29.45	1.34	115.7	89.6	98.74
KO	40.5	8.2	35.3	3.1	11	0.64	363	16	3.34	20.9	62.2	29.28	1.53	111.5	90.8	98.71
PE	46.8	11.8	88.7	4.6	13.7	0.72	318	22.8	4.86	21.1	67.9	31.35	1.53	115.7	87.5	98.76
TR	45.8	10.7	63.8	5.1	16.8	0.73	350	19.1	3.89	19.1	50.8	30.13	1.75	114.3	90.5	98.63
Mean Value	48.19	11.52	63.65	5.21	16.54	0.76	342.18	19.62	4.02	21.00	60.16	30.52	1.38	114.72	91.41	98.92
F Value	4.86	6.05	4.96	6.83	14.12	12.09	9.81	16.38	8.65	2.94	12.00	3.06	9.770	4.23	3.89	1.04
P Value	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.003	0.000	0.002	0.000	0.000	0.000	0.414

Fig. 4.33 a-b. Variations of *S. spectabilis* growth parameters in different locations of Kerala

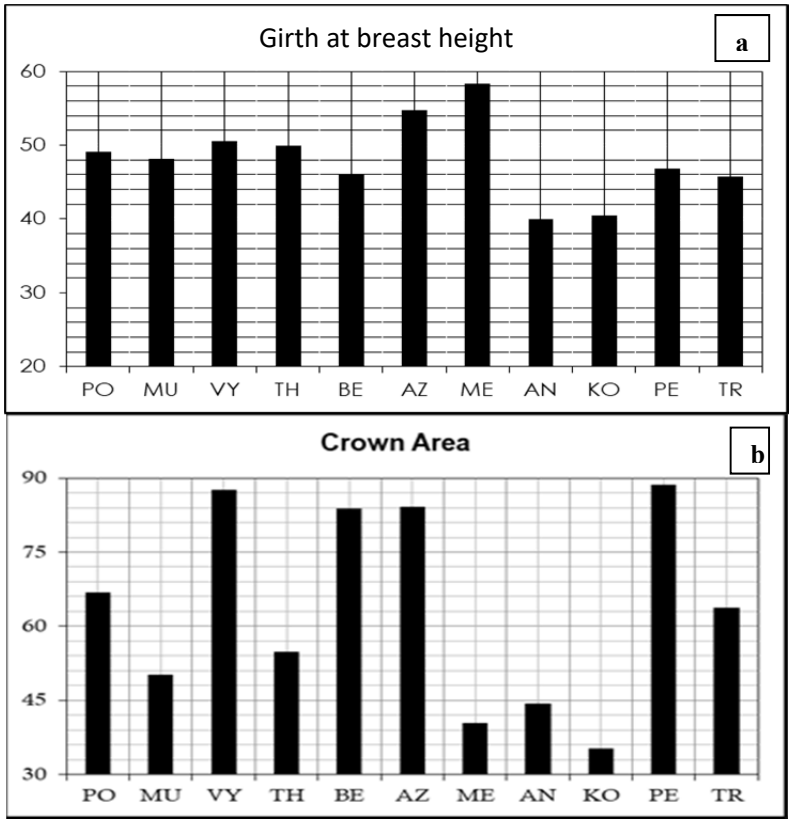
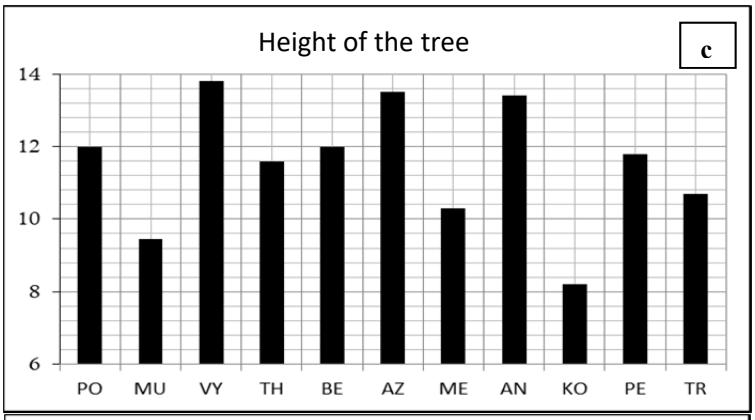
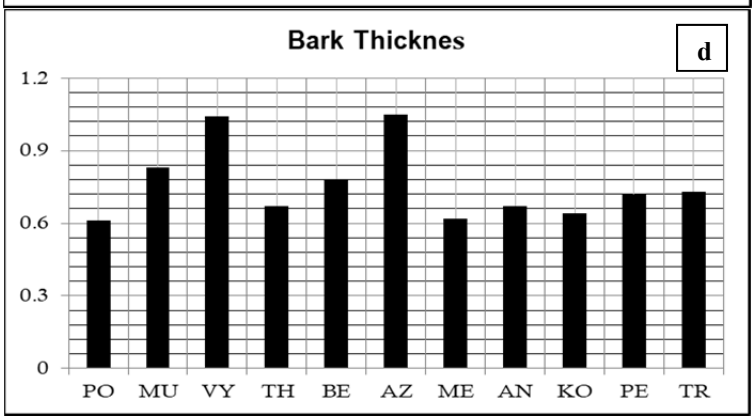
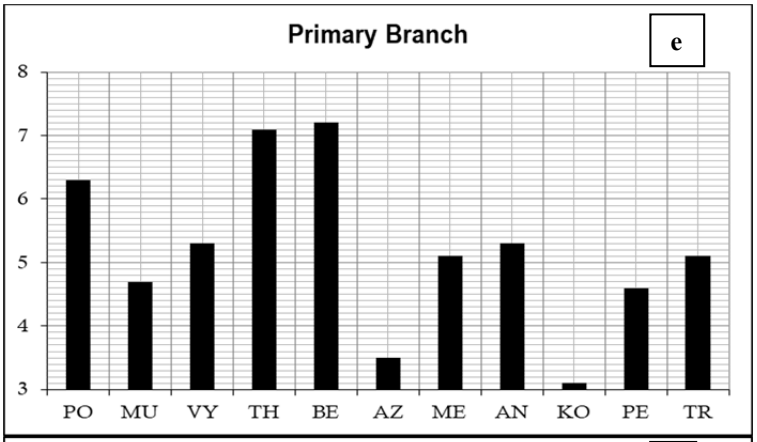
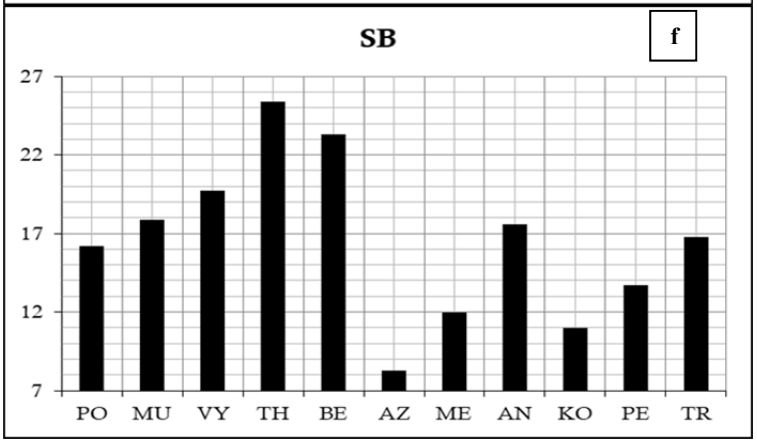


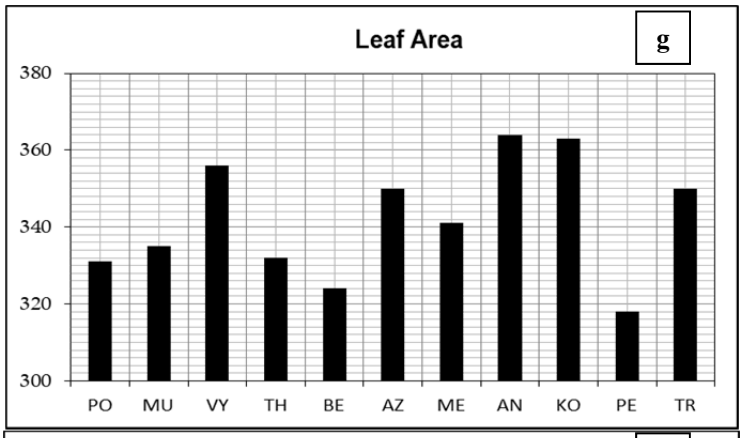
Fig. 4.33.c-j. Variations of *S.spectabilis* growth parameters in different locations of Kerala










Leaf Area
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320
300
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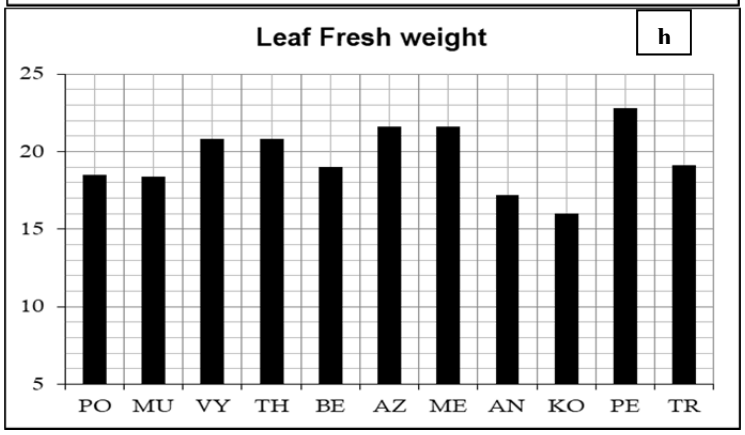
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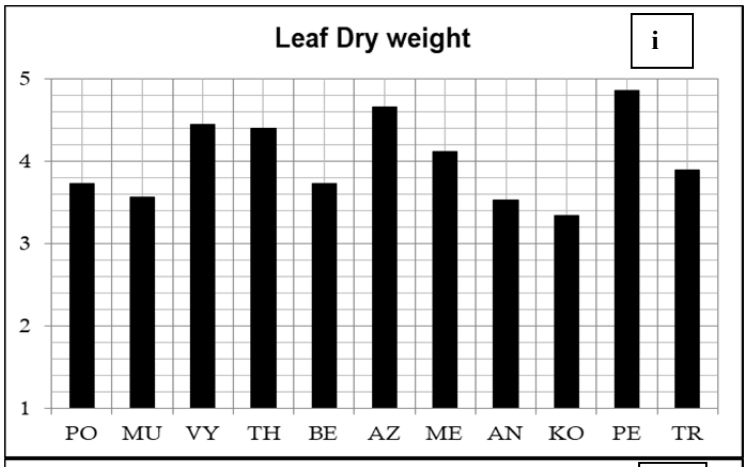
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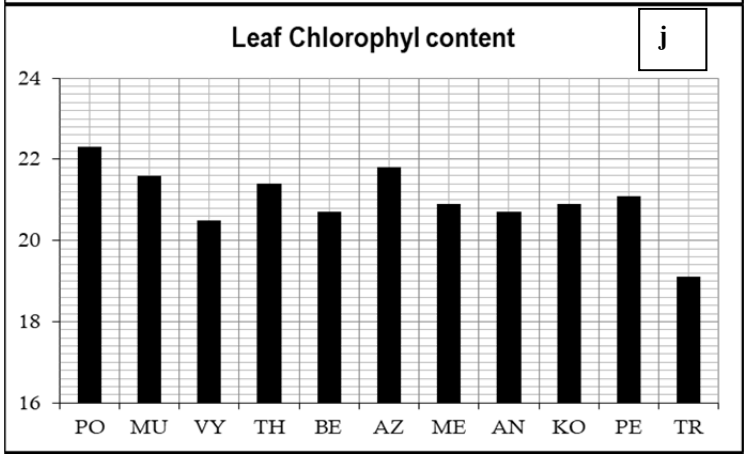
Leaf Chlorophyl content
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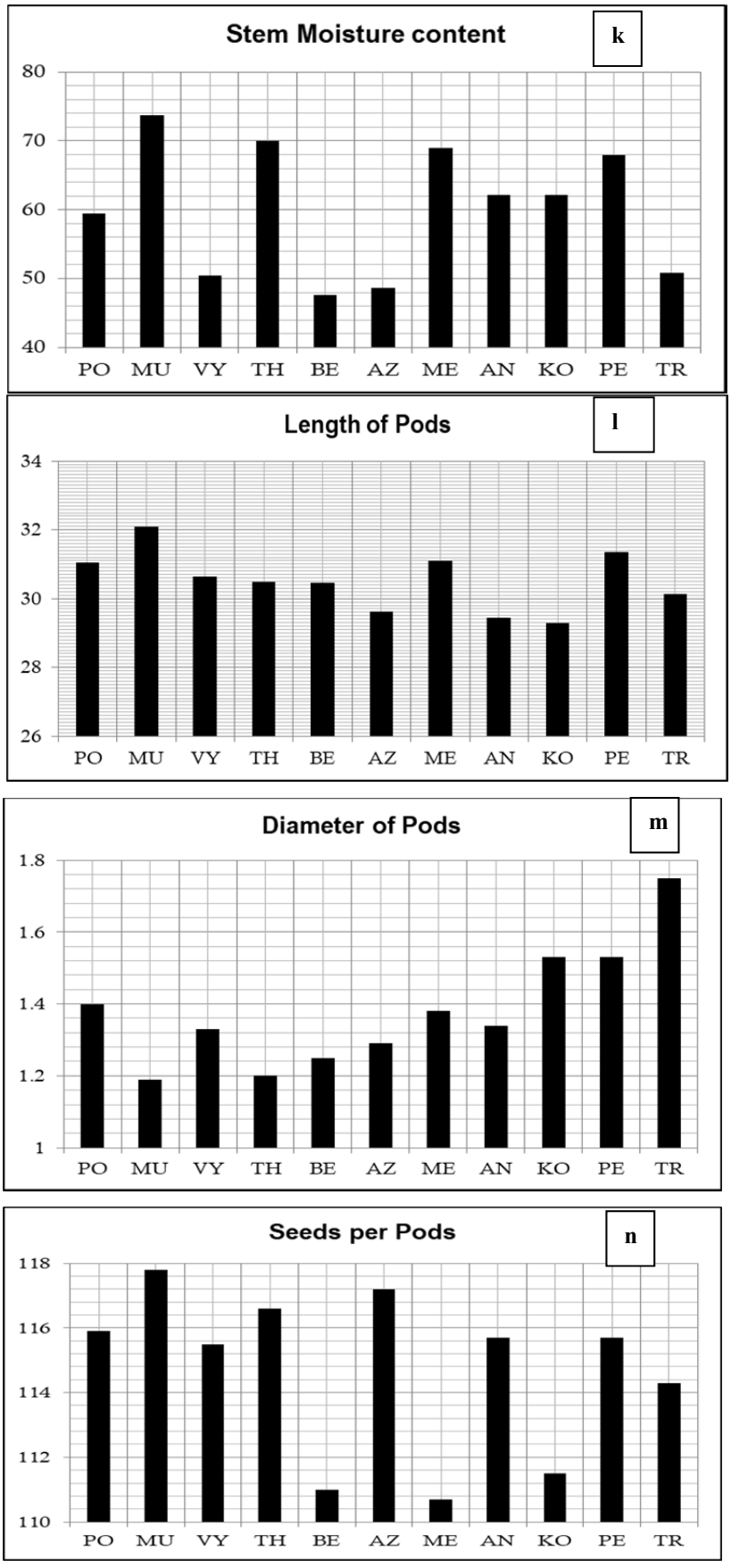
PO MU VY TH BE AZ ME AN KO PE TR

Fig. 4.33.k-p. Variations of *S.spectabilis* growth parameters in different locations of Kerala



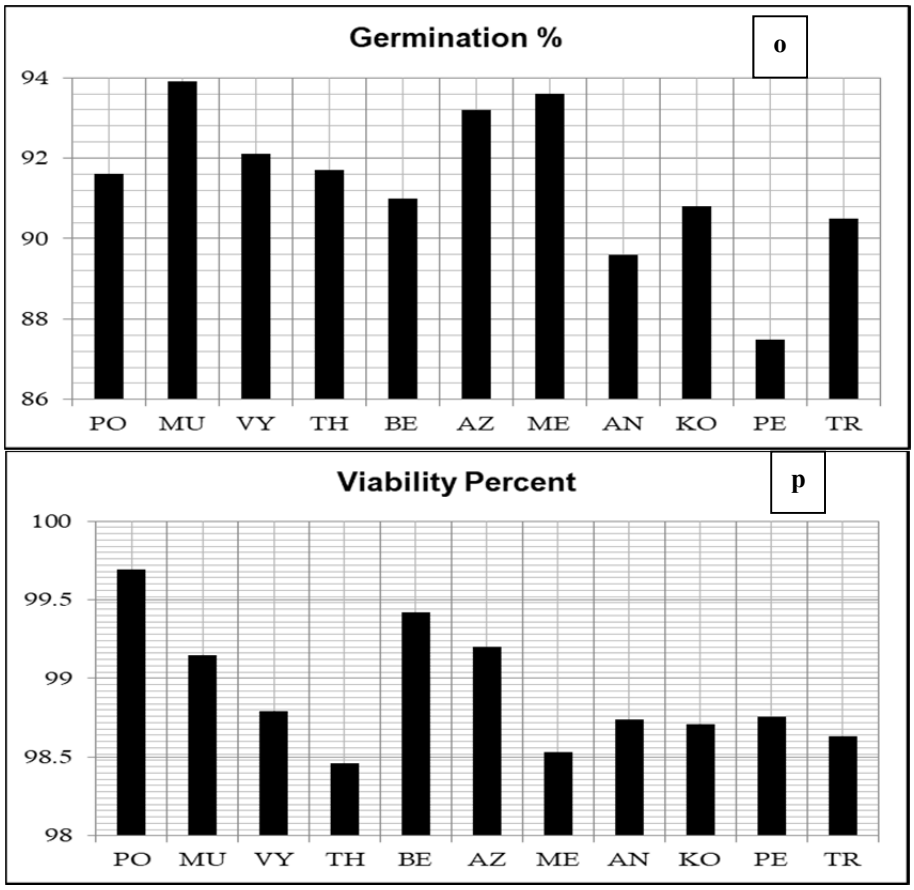


Table 4.27 Population-wise tree characters of *Senna spectabilis* in Kerala

GBH	PO	MU	VY	TH	BE	AZ	ME	AN	KO	PE	TR
N	10.00	10.00	10.00	10.00	10.00	10.00	10.00	10.00	10.00	10.00	10.00
Min	42.50	39.50	39.50	38.60	36.70	41.00	38.00	33.20	35.00	35.60	32.50
Max	55.00	54.00	62.00	57.20	57.00	73.00	90.00	46.20	48.00	62.00	55.00
Sum	490.50	481.20	505.90	499.60	461.20	548.30	583.40	399.90	404.70	468.40	458.00
Mean	49.05	48.12	50.59	49.96	46.12	54.83	58.34	39.99	40.47	46.84	45.80
SE	1.22	1.31	2.36	2.10	2.26	3.45	4.53	1.33	1.27	2.77	2.25
VAR	14.80	17.16	55.69	44.00	50.88	118.80	205.51	17.66	16.16	76.93	50.48
SD	3.85	4.14	7.46	6.63	7.13	10.90	14.34	4.20	4.02	8.77	7.10
CV	7.84	8.61	14.75	13.28	15.47	19.88	24.57	10.51	9.93	18.73	15.51

HT	PO	MU	VY	TH	BE	AZ	ME	AN	KO	PE	TR
N	10	10	10	10	10	10	10	10	10	10	10
Min	9	7	8.5	9.5	8	9	8	9	6	9	6
Max	15	14	18	15	15	20	13	17	10	15	15
Sum	119.50	94.50	137.50	115.50	120.00	135.00	102.50	134.00	82.00	118.00	107.00
Mean	11.95	9.45	13.75	11.55	12.00	13.50	10.25	13.40	8.20	11.80	10.70
SE	0.59	0.72	0.86	0.58	0.70	0.97	0.55	0.72	0.44	0.68	0.84
VAR	3.47	5.14	7.40	3.36	4.89	9.39	3.07	5.16	1.90	4.62	7.12
SD	1.86	2.27	2.72	1.83	2.21	3.06	1.75	2.27	1.38	2.15	2.67
CV	15.59	23.98	19.79	15.87	18.43	22.70	17.09	16.94	16.81	18.22	24.94

CA	PO	MU	VY	TH	BE	AZ	ME	AN	KO	PE	TR
N	10	10	10	10	10	10	10	10	10	10	10
Min	35.77	16.61	38.46	31.65	50.24	47.15	17.71	19.63	12.56	47.15	17.71
Max	153.86	78.5	153.86	90.71	153.86	153.86	60.1	78.5	67.16	122.66	122.66
Sum	668.05	502.1	876.73	548.03	837.84	842.07	405.11	444.29	353.33	887.26	638.26
Mean	66.81	50.21	87.67	54.80	83.78	84.21	40.51	44.43	35.33	88.73	63.83
SE	12.31	6.42	11.78	5.94	10.18	10.31	4.71	5.68	5.38	7.76	12.97
VAR	1515.83	412.20	1387.55	352.38	1035.34	1062.98	221.48	322.51	289.95	601.80	1683.20
SD	38.93	20.30	37.25	18.77	32.18	32.60	14.88	17.96	17.03	24.53	41.03
CV	58.28	40.44	42.49	34.25	38.40	38.72	36.74	40.42	48.19	27.65	64.28

PB	PO	MU	VY	TH	BE	AZ	ME	AN	KO	PE	TR
N	10.00	10.00	10.00	10.00	10.00	10.00	9.00	10.00	10.00	10.00	10.00
Min	4.00	2.00	4.00	5.00	5.00	2.00	4.00	2.00	2.00	2.00	2.00
Max	9.00	7.00	8.00	9.00	9.00	5.00	7.00	9.00	5.00	8.00	8.00
Sum	63.00	47.00	53.00	71.00	72.00	35.00	46.00	53.00	31.00	46.00	51.00
Mean	6.30	4.70	5.30	7.10	7.20	3.50	5.11	5.30	3.10	4.60	5.10
SE	0.47	0.47	0.42	0.41	0.42	0.37	0.39	0.79	0.38	0.60	0.57
VAR	2.23	2.23	1.79	1.66	1.73	1.39	1.36	6.23	1.43	3.60	3.21
SD	1.49	1.49	1.34	1.29	1.32	1.18	1.17	2.50	1.20	1.90	1.79
CV	23.72	31.80	25.24	18.12	18.29	33.67	22.83	47.11	38.62	41.25	35.14

Table 4.27. Population wise tree characters of *Senna spectabilis* in Kerala

SB	PO	MU	VY	TH	BE	AZ	ME	AN	KO	PE	TR
N	10	10	10	10	10	10	10	10	10	10	10
Min	12	14	15	19	16	5	4	6	6	6	9
Max	21	22	25	31	32	16	21	26	16	22	22
Sum	162	179	197	254	233	83	120	176	110	137	168
Mean	16.20	17.90	19.70	25.40	23.30	8.30	12.00	17.60	11.00	13.70	16.80
SE	0.90	0.86	1.05	1.01	1.71	1.07	1.84	2.11	1.01	1.60	1.20
VAR	8.18	7.43	11.12	10.27	29.12	11.34	34.00	44.49	10.22	25.57	14.40
SD	2.86	2.73	3.34	3.20	5.40	3.37	5.83	6.67	3.20	5.06	3.79
CV	17.65	15.23	16.93	12.61	23.16	40.58	48.59	37.90	29.07	36.91	22.59

BT	PO	MU	VY	TH	BE	AZ	ME	AN	KO	PE	TR
N	10	10	10	10	10	10	10	10	10	10	10
Min	0.23	0.63	0.85	0.52	0.64	0.82	0.47	0.49	0.49	0.54	0.49
Max	1.02	1.21	1.34	0.81	1.05	1.26	0.86	0.97	0.76	0.89	0.87
Sum	6.05	8.33	10.4	6.65	7.84	10.53	6.16	6.74	6.41	7.19	7.26
Mean	0.61	0.83	1.04	0.67	0.78	1.05	0.62	0.67	0.64	0.72	0.73
SE	0.08	0.06	0.04	0.03	0.04	0.04	0.04	0.05	0.03	0.03	0.03
VAR	0.07	0.04	0.02	0.01	0.02	0.02	0.02	0.02	0.01	0.01	0.01
SD	0.26	0.19	0.13	0.09	0.12	0.13	0.13	0.14	0.09	0.09	0.11
CV	42.67	22.79	12.44	14.06	15.79	11.96	21.39	21.17	14.08	12.94	15.16

LC	PO	MU	VY	TH	BE	AZ	ME	AN	KO	PE	TR
N	10	10	10	10	10	10	10	10	10	10	10
Min	19.68	19.52	18.69	18.74	18.65	19.85	18.69	18.36	18.72	19.26	17.49
Max	25.31	23.17	24.72	24.06	22.87	24.31	22.71	22.38	23.64	23.71	21.38
Sum	222.81	215.98	205.15	214.12	206.53	218.05	208.59	207.35	208.53	210.79	191.25
Mean	22.28	21.6	20.52	21.41	20.65	21.81	20.86	20.74	20.85	21.08	19.13
SE	0.56	0.43	0.56	0.51	0.44	0.45	0.39	0.44	0.61	0.48	0.37
VAR	3.16	1.84	3.11	2.59	1.9	2.01	1.51	1.89	3.76	2.33	1.34
SD	1.78	1.36	1.76	1.61	1.38	1.42	1.23	1.38	1.94	1.53	1.16
CV	7.97	6.28	8.59	7.52	6.68	6.5	5.88	6.64	9.3	7.24	6.05

SM	PO	MU	VY	TH	BE	AZ	ME	AN	KO	PE	TR
N	10	10	10	10	10	10	10	10	10	10	10
Min	45.31	51.36	36.41	57.28	39.54	39.54	59.42	49.52	47.06	49.05	40.52
Max	71.23	84.78	71.25	80.55	57.21	61.85	80.02	70.39	78.23	77.03	69.35
Sum	593.78	737.26	503.84	699.9	475.85	487.04	690.15	620.62	622.27	679.07	508.02
Mean	59.38	73.73	50.38	69.99	47.59	48.7	69.02	62.06	62.23	67.91	50.8
SE	3.16	3.19	3.11	2.48	2.17	2.08	2.56	2.65	3.18	2.66	2.62
VAR	99.62	101.51	96.66	61.52	47.02	43.09	65.49	70.13	100.84	70.76	68.38
SD	9.98	10.08	9.83	7.84	6.86	6.56	8.09	8.37	10.04	8.41	8.27
CV	16.81	13.67	19.51	11.21	14.41	13.48	11.73	13.49	16.14	12.39	16.28

Table 4.27 Population wise tree characters of *Senna spectabilis* in Kerala

LA	PO	MU	VY	TH	BE	AZ	ME	AN	KO	PE	TR
N	10	10	10	10	10	10	10	10	10	10	10
Min	310.49	308.50	329.50	300.50	308.50	316.58	308.50	345.20	336.50	308.90	325.80
Max	353.20	360.20	371.50	371.40	340.20	370.60	362.40	374.60	374.80	326.50	370.20
Sum	3305.89	3352.05	3562.80	3324.05	3238.30	3502.98	3412.60	3635.60	3625.80	3180.25	3500.50
Mean	330.59	335.21	356.28	332.41	323.83	350.30	341.26	363.56	362.58	318.03	350.05
SE	4.60	6.39	4.82	7.55	3.42	5.51	6.08	3.05	4.22	2.22	3.97
VAR	211.37	408.01	232.19	569.75	117.29	304.05	369.95	93.01	177.86	49.15	157.36
SD	14.54	20.20	15.24	23.87	10.83	17.44	19.23	9.64	13.34	7.01	12.54
CV	4.40	6.03	4.28	7.18	3.34	4.98	5.64	2.65	3.68	2.20	3.58

LF	PO	MU	VY	TH	BE	AZ	ME	AN	KO	PE	TR
N	10	10	10	10	10	10	10	10	10	10	10
Min	16.50	15.40	18.40	17.60	16.50	19.60	19.30	15.30	14.90	20.30	16.30
Max	19.60	21.60	22.60	25.30	20.70	25.10	24.10	19.20	17.40	25.30	21.30
Sum	185.40	183.70	207.90	208.00	189.80	216.30	215.60	172.10	160.30	228.18	190.97
Mean	18.54	18.37	20.79	20.80	18.98	21.63	21.56	17.21	16.03	22.82	19.10
SE	0.31	0.60	0.45	0.80	0.43	0.56	0.53	0.47	0.26	0.45	0.54
VAR	0.96	3.66	2.06	6.44	1.86	3.14	2.79	2.25	0.69	2.01	2.96
SD	0.98	1.91	1.44	2.54	1.36	1.77	1.67	1.50	0.83	1.42	1.72
CV	5.28	10.41	6.91	12.20	7.18	8.19	7.74	8.72	5.18	6.21	9.00

LD	PO	MU	VY	TH	BE	AZ	ME	AN	KO	PE	TR
N	10.00	10.00	10.00	10.00	10.00	10.00	10.00	10.00	10.00	10.00	10.00
Min	3.11	2.93	3.67	3.08	3.14	3.65	3.24	2.84	2.85	4.17	3.34
Max	5.38	4.36	5.34	5.31	4.25	5.36	5.60	4.27	4.08	5.30	4.28
Sum	37.32	35.58	44.42	43.98	37.29	46.58	41.11	35.32	33.43	48.56	38.94
Mean	3.73	3.56	4.44	4.40	3.73	4.66	4.11	3.53	3.34	4.86	3.89
SE	0.23	0.15	0.19	0.21	0.11	0.19	0.21	0.16	0.13	0.13	0.11
VAR	0.54	0.23	0.35	0.46	0.12	0.34	0.44	0.26	0.17	0.17	0.13
SD	0.73	0.48	0.59	0.68	0.35	0.59	0.67	0.51	0.41	0.41	0.36
CV	19.67	13.61	13.23	15.43	9.37	12.60	16.18	14.54	12.18	8.43	9.33

LP	PO	MU	VY	TH	BE	AZ	ME	AN	KO	PE	TR
N	10	10	10	10	10	10	10	10	10	10	10
Min	28.1	28.6	28.5	28.1	28.3	27.6	28.7	28	27	30	27
Max	34	34	33.5	33.1	32.6	32.4	33.6	31	32.3	33.2	32.4
Sum	310.6	321	306.54	304.85	304.7	296.3	311.1	294.51	293	314	301.3
Mean	31.06	32.1	30.654	30.485	30.47	29.63	31.11	29.451	29.3	31.4	30.13
SE	0.6	0.68	0.44	0.47	0.43	0.47	0.45	0.36	0.52	0.32	0.58
VAR	3.55	4.63	1.98	2.24	1.84	2.19	2.02	1.29	2.68	1.05	3.36
SD	1.88	2.15	1.41	1.5	1.35	1.48	1.42	1.13	1.64	1.03	1.83
CV	6.06	6.7	4.59	4.91	4.45	5	4.57	3.85	5.59	3.27	6.08

Table 4.27 Population wise tree characters of *Senna spectabilis* in Kerala

DP	PO	MU	VY	TH	BE	AZ	ME	AN	KO	PE	TR
N	10	10	10	10	10	10	10	10	10	10	10
Min	1.12	1.12	1.13	1.02	1.11	1.23	1.11	1.03	1.26	1.33	1.57
Max	1.59	1.34	1.93	1.56	1.38	1.47	1.58	1.54	1.94	1.75	1.91
Sum	14.04	11.94	13.34	12.02	12.52	12.9	13.76	13.4	15.25	15.27	17.51
Mean	1.404	1.194	1.334	1.202	1.252	1.29	1.376	1.34	1.525	1.527	1.751
SE	0.05	0.02	0.10	0.05	0.03	0.03	0.06	0.05	0.07	0.04	0.04
VAR	0.03	0.00	0.09	0.02	0.01	0.01	0.04	0.02	0.05	0.02	0.01
SD	0.16	0.07	0.30	0.16	0.08	0.09	0.20	0.15	0.22	0.14	0.12
CV	11.68	5.49	22.80	12.90	6.73	6.60	14.80	11.25	14.72	9.09	6.79
SP	PO	MU	VY	TH	BE	AZ	ME	AN	KO	PE	TR
N	10	10	10	10	10	10	10	10	10	10	10
Min	110	114	108	112	106	110	106	110	108	104	108
Max	122	122	123	123	118	124	114	120	116	124	119
Sum	1159	1178	1155	1166	1110	1172	1107	1157	1115	1157	1143
Mean	115.9	117.8	115.5	116.6	111	117.2	110.7	115.7	111.5	115.7	114.3
SE	1.37	0.90	1.55	0.98	1.24	1.46	0.79	0.90	0.89	1.92	0.97
VAR	18.77	8.18	24.06	9.60	15.33	21.29	6.23	8.01	7.83	36.90	9.34
SD	4.33	2.86	4.90	3.10	3.92	4.61	2.50	2.83	2.80	6.07	3.06
CV	3.74	2.43	4.25	2.66	3.53	3.94	2.26	2.45	2.51	5.25	2.67

GM%	PO	MU	VY	TH	BE	AZ	ME	AN	KO	PE	TR
N	10	10	10	10	10	10	10	10	10	10	10
Min	82	90	90	86	89	90	90	85	86	82	87
Max	98	98	96	96	95	98	96	96	93	91	94
Sum	916	939	921	917	910	932	936	896	908	875	905
Mean	91.60	93.90	92.10	91.70	91.00	93.20	93.60	89.60	90.80	87.50	90.50
SE	1.69	0.74	0.71	1.00	0.60	0.81	0.56	1.11	0.65	1.11	0.83
VAR	28.71	5.43	4.99	10.01	3.56	6.62	3.16	12.27	4.18	12.28	6.94
SD	5.36	2.33	2.23	3.16	1.89	2.57	1.78	3.50	2.04	3.50	2.64
CV	5.85	2.48	2.43	3.45	2.07	2.76	1.90	3.91	2.25	4.00	2.91

VP	PO	MU	VY	TH	BE	AZ	ME	AN	KO	PE	TR
N	10	10	10	10	10	10	10	10	10	10	10
Min	97.4	97.4	97.6	96.6	98.4	98.2	97.5	97.2	97.1	97.4	93.2
Max	106.4	100	99.2	100	100	100	99.1	100	100	100	100
Sum	997.00	991.60	988.00	984.60	994.20	992.00	985.10	987.30	987.00	987.50	986.40
Mean	99.7	99.16	98.8	98.46	99.42	99.2	98.51	98.73	98.7	98.75	98.64
SE	0.80	0.25	0.18	0.37	0.17	0.25	0.18	0.28	0.30	0.27	0.66
VAR	6.33	0.64	0.33	1.36	0.30	0.62	0.31	0.79	0.89	0.75	4.33
SD	2.52	0.80	0.58	1.17	0.55	0.79	0.56	0.89	0.95	0.86	2.08
CV	2.52	0.81	0.58	1.19	0.55	0.79	0.57	0.90	0.96	0.88	2.11

Analysis of variance

The analysis of variance (ANOVA) revealed that there was significant variation ($p<0.05$) observed between the populations in all 16 characters used for morphological variation analysis of 10 trees from 11 populations. The ANOVA results indicated a range of p-values for all the characters, ranging from 0.000 to 0.003, except for the seed viability percentage ($p=0.414$). The p-values for the remaining characters ranged between 0.001 and 0.004. Therefore, the ANOVA confirms that significant variation was observed between the populations of the fifteen characters subjected to variation analysis.

According to Tesfay *et al.,* (2023), the success of invasive species can be attributed to two factors pertaining to the response of their fitness characteristics to environmental changes. Firstly, invasive species demonstrate high adaptability in maintaining their fitness across diverse environments. Secondly, they possess the ability to enhance their fitness in favorable environments. The evolutionary development of traits in invasive plant species holds significant importance as it can influence crucial demographic parameters that contribute to their successful invasion of new habitats (Hodgins *et al.,* 2018). Numerous studies have provided evidence of phenotypic variations in invasive plant species along geographic gradients (Hodgins *et al.,* 2018). For instance, variations in reproductive capacity and growth-related traits have been observed in invasive species such as *Impatiens glandulifera* (Balsaminaceae) (Kollman and Bañuelos, 2004), *Eschscholzia californica* (Papaveraceae) (Leger and Rice, 2007), *Lythrum salicaria* (Lythraceae) (Montague *et*

al., 2008), *Ambrosia artemisiifolia* (Asteraceae) (van Boheemen *et al.,* 2019), and *Alternanthera philoxeroides* (Amaranthaceae) (Yang *et al.*, 2021).

In a recent investigation conducted by Du *et al.,* (2023), notable differentiation in growth and reproductive traits, as well as chlorophyll content, was observed among populations of the invasive species *Solidago Canadensis* in China. This differentiation was evident when comparing populations across a wide geographic range. The study involved cultivating individuals from diverse populations under controlled and uniform conditions in a typical garden setting, suggesting that the observed intraspecific variation is likely attributable to genetic differences in these traits.

The capacity of an organism to enhance its fitness in response to a changing environment confers a competitive advantage over species lacking such adaptability (Pérez-Ramos *et al.,* 2019; Mroue *et al.,* 2018). Notably, plants, as a group, have demonstrated their ability to adapt to rapid climate change in order to optimize their fitness in variable environments (Anderson and Song, 2020).

Invasive plants have demonstrated their ability to adapt to new environmental conditions, suggesting they may also possess a greater capacity to adapt to environmental conditions caused by climate change than native species. This adaptability contributes to their competitive advantage; observed that the phenotypic changes observed in invasive plants within their introduced ranges indicate their ability to adapt to novel environmental conditions, which enables them to tolerate and invade diverse geographic areas successfully (Schultheis and MacGuigan, 2018; Ghalambor *et al*., 2007).

Understanding the ecological and evolutionary mechanisms that underlie the phenotypic and genetic diversity observed among invasive plant populations and their ability to successfully colonize diverse environments is a crucial objective within the field of ecology. The divergence in phenotypic and genetic traits among natural populations can be attributed to the processes of natural selection and may confer adaptive advantages (Linhart and Grant, 1996). If populations possess a sufficient amount of heritable variation in their phenotypic traits and are subjected to strong natural selection pressures, they can evolve adaptations that are specifically suited to the local environmental conditions (Hall and Willis, 2006). Numerous studies have demonstrated that invasive plant species exhibit local adaptation across a wide range of

environmental conditions (Oduor *et al.,* 2016). The findings of these studies also underscore the significant influence of adaptive environmental and genetic factors on the phenotypic characteristics of invasive plants, thereby emphasizing the necessity for a comprehensive investigation into the population genetics of these species within this particular context.

4.2.5. Study on Karyotypic variations of *Senna spectabilis* among different populations of Kerala

The present study reports the results of karyotype analysis conducted on six populations of *Senna spectabilis*, namely Anakkatty, Muthanga, Vythiri, Azhinjilam, Munnar, and Thiruvananthapuram. The root tips of *S. spectabilis* were utilized for the analysis, and the findings are presented in Plate 4.9.a and 4.9.b. The mitotic metaphase cells of all six populations exhibited a stable chromosome count of 2n= 28, x=14 (Table 4.28). Comparison with the karyotype analysis of a native *S. spectabilis* specimen collected from Royal Botanic Gardens, Kew by Mohanty and Das (2006) revealed that the chromosome number remained unchanged at 2n =28(Table 4.28). The chromosome length of the native specimen was 1.33 µm, while the populations collected from Kerala did not exhibit any significant variation.. The results revealed that the sample collected from Muthanga exhibited the highest chromosome length (1.35 µm), whereas Vythiri and Thiruvananthapuram had the lowest chromosome length (1.33 µm). The arm ratio of the mitotic chromosome was found to be 1.26 for the five populations, except for the Anakkatty population, which exhibited a slightly lower value of 1.25 (Table 4.28). Notably, these values were lower than the native specimen records of 1.32, although no significant differences were observed among the Kerala populations.

However, the statistical analysis indicated a significant difference between the rest of the characters studied, including chromosome length, total genomic chromosome length, total chromosome volume, and total form percentage, which exhibited statistically significant variations among the populations. These findings suggest that the chromosome characteristics of the studied populations may be influenced by various factors, such as environmental conditions, genetic variations, and evolutionary processes. Further research is warranted to explore the underlying mechanisms and implications of these variations in the context of population genetics and evolutionary biology.

Table 4.28 The Karyotype details of the characters studied

Population	Chromosome number (2n)	Chromosome length (µm)	Total genomic Chromosome length (µm)	Total chromosome volume(µm^3)	Total form%	Arm ratio
Anaikkatty	28	1.34	28.45 ±0.04	7.88 ± 0.06	36.53 ± 0.01	1.25 ± 0.01
Muthanga	28	1.35	26.33 ±0.21	8.44 ± 0.02	37.46 ± 0.01	1.26 ± 0.00
Vythiri	28	1.33	27.08 ±0.01	8.73 ± 0.01	37.53 ± 0.01	1.26 ± 0.00
Thiruvanthapuram	28	1.33	27.60 ±0.02	8.93 ± 0.01	37.62 ± 0.02	1.26 ± 0.00
Azhinjilam	28	1.34	26.92 ±0.02	8.77 ± 0.00	37.54 ± 0.00	1.26 ± 0.00
Munnar	28	1.34	27.86 ±0.02	8.42 ± 0.03	36.94 ± 0.04	1.26 ± 0.00
SEM	0.09*	0.03*	0.02*	NS		
Mohanty et al, 2006	**28**	**1.33**	**22.85**	**9.50**	**38.55**	**1.32**

* indicates significant differences at 5 % of significance, and NS indicates non-significant differences.

The results indicated that the total genomic chromosome length of *S. spectabilis* ranged from 26.33µm to 28.45 µm, with the sample from Anaikkatty exhibiting the highest total genomic chromosome length (28.45 µm) and Muthanga displaying the lowest (26.33µm). The present values for total genomic chromosome length ranged from 25.85 µm to 28.41µm, while the plants from the native region had a genomic chromosome length of 22.85 µm. The total chromosome volume of *S. spectabilis* varied from 7.88 µm3 (Anaikkatty) to 8.93 µm3 (Thiruvananthapuram), with the total chromosome volume of the native specimen being 9.5 µm3, which was higher than the values recorded from Kerala. The total form percentage of *S. spectabilis* collected from six locations ranged from 36.53 % (Anaikkatty) to 37.62% (Thiruvananthapuram), which was lower than the observation from the native specimen (38.55).

Statistical analysis of the observations provided evidence of chromosome variations in the populations in Kerala. These differences in measurements could be attributed to variations in condensation levels during processing. Additionally, the

genus *S. spectabilis* is known to be highly unstable, and these differences could indicate the need for a more extensive and in-depth study.

The taxonomic classification of the plant genus Senna has been a subject of much debate and disagreement. Initially, it was considered a part of the larger genus Cassia, known for its extensive diversity, consisting of nearly 500 species. However, the delimitation and taxonomic position of this plant group been the cause of considerable confusion.

The genus Cassia was divided into three subgenera: fistula, senna, and lasiorhegma (Bentham, 1871). Subsequently Britton and Rose (1930) proposed splitting Cassia into twenty-eight separate genera. However, Irwin and Barneby, (1981) recognized Cassia, Senna, and Chamaecrita as three subgenera of the genus Cassia. Later, in 1982, they further elevated both Senna and Chamaecrita to separate genera (Irwin and Barneby, 1982).

The taxonomic status of the genus Cassia remains perplexing due to the blurred distinctions between and within the three genera caused by factors such as polyploidy, hybridization, and apomixes (Lewis *et al*., 1980). For instance, Isley (1975) recognized *Cassia excelsa* and *C. spectabilis* as distinct species, while Irwin and Barneby (1982), considering the overlap in measurements, identified them as two varieties of the species *S. spectabilis*. The native Brazilian population was designated as var.*excelsa*, while the more commonly cultivated variety was recognized as

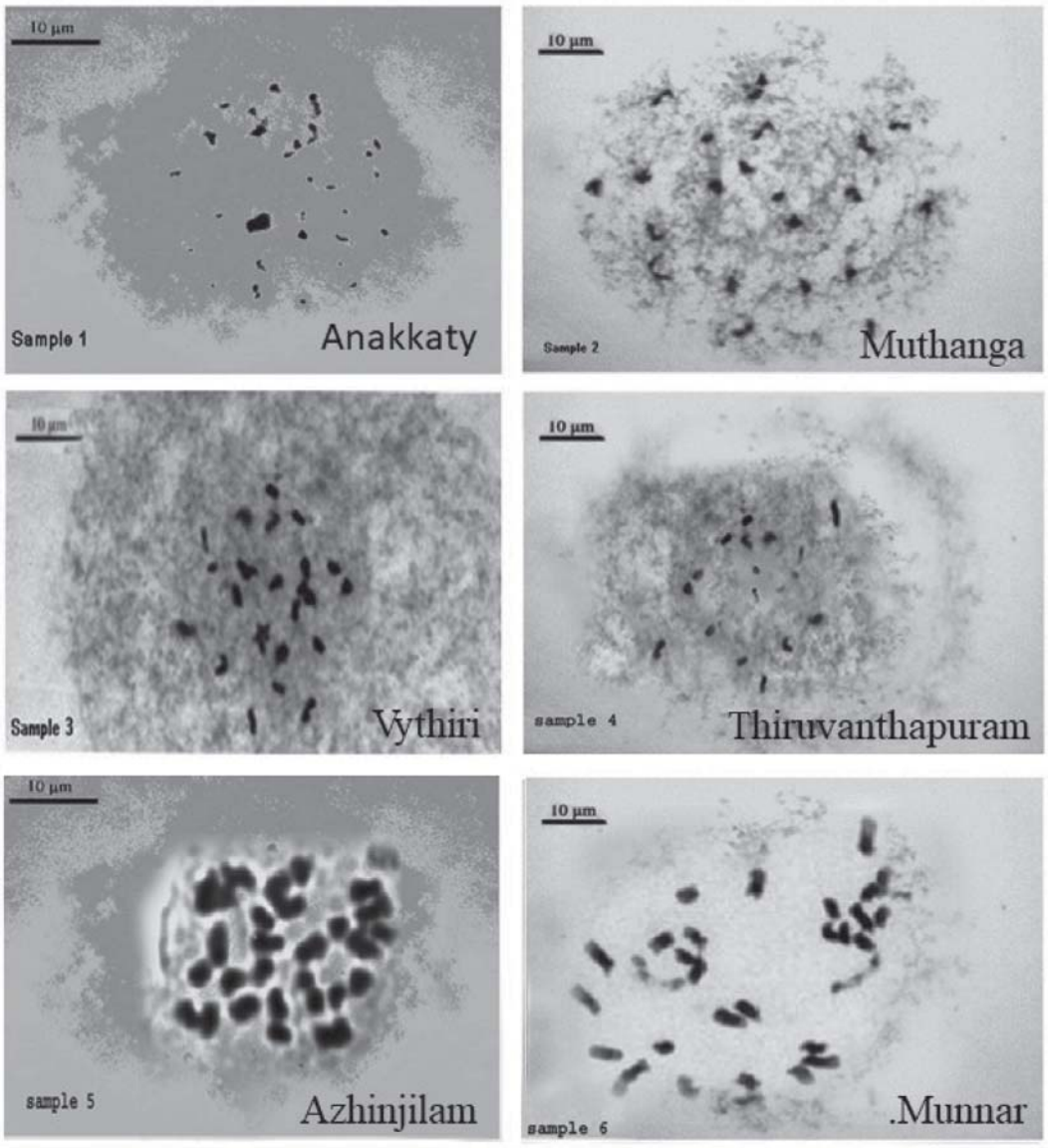


Plate.4.9.a Somatic chromosome plates in the species of Senna spectabilis (2n = 28)

Plate.4.9.b. Germinated seeds of *Senna spectabilis* with root tips

var. *spectabilis* (PIER, 2014). In conclusion, the taxonomic classification of the genus Cassia, particularly its relationship with the genus Senna, has been a subject of ongoing debate and confusion. The distinctions between and within these genera have been complicated by various factors, making it challenging to establish a clear taxonomic position for this plant group. One of the most essential features of an invasive species is its broad adaptation to a range of biotic and abiotic factors. This adaptation could be through genetic or plastic changes. Considerable work needs to be done on this invasive population in South India to arrive at conclusive evidence on taxonomic relationships and evolutionary changes in detail.

4.2.6 To study the genetic diversity of *Senna spectabilis* from different locations of Kerala.

This study aimed to assess the genetic variability of eleven accessions of *Senna spectabilis* collected from different locations of Kerala using Inter Simple Sequence Repeat (ISSR) markers. The ISSR primer was used to evaluate the degree of polymorphism. Genetic diversity studies are essential for understanding the mechanism of a successful invasion.

DNA isolation

In this study, eleven locations were identified to collect *S. spectabilis*, to conduct the study on genetic variation. The modified CTAB protocol yielded a high concentration of good-quality DNA fit for PCR amplification (Table 4.29).

Table 4.29 DNA yield and purity from young leaves of *Senna spectabilis* from eleven locations.

Sl No.	Code	Location of samples	Leaf type	DNA (µg/ml)	A_{260}/A_{280}
1	Ss-1	Ponkuzhy	Young	1484.8	1.62
2	Ss-2	Vythiri	Young	498.3	1.77
3	Ss-3	Muthanga	Young	1090.9	1.85
4	Ss-4	Tholpetty	Young	924.6	1.52
5	Ss-5	Meppadi	Young	408.1	1.65
6	Ss-6	Azhinjilam	Young	473.7	1.73
7	Ss-7	Begur	Young	392.4	1.65
8	Ss-8	Anaikatty	Young	450	1.60
9	Ss-9	Kottathara	Young	398.6	1.64
10	Ss-10	Periyar	Young	1044.1	1.78
11	Ss-11	Thiruvananthapuram	Young	1199.7	1.84

ISSR PCR amplification

Optimization of annealing temperature and time is crucial for achieving successful PCR amplification. The highest amplification of DNA was observed at an annealing temperature of 60°C for 45 seconds. The 5' CAGCAGCAGCAGCAG 3' primer [$(CAG)_5$]demonstrated good amplification in ISSR PCR among the eight primers tested. These conditions resulted in well-defined bands in the amplification products (Plate.4.10.a and 4.10.b). The size of each fragment was determined using a 100bp DNA ladder as a standard.

This study also aimed to identify a DNA extraction protocol suitable for PCR applications from *Senna spectabilis* leaves. A modified CTAB extraction method (Southern, 1975; Tai *et al*., 1990; Sanghai *et al.,* 1984; Williams *et al.,* 1990) was employed to remove polyphenols, polysaccharides, and other secondary metabolites. Polysaccharides are known to inhibit PCR (Kotchoni *et al*., 2003). This protocol yielded high high-quality DNA concentrations, successfully amplified using ISSR primers.

A previous study by Mohanty *et al.,* (2010) investigated the genetic diversity of cassia species and identified 11 ISSR primers that produced 111 polymorphic bands. Our study analyzed eight ISSR primers and found that the $(CAG)_5$ primer produced well-defined PCR bands—the (CAG)5 primer generated scorable bands ranging from 162 to 1487 bp.

Genetic variability of *Senna spectabilis* in populations of Kerala

Data analysis was conducted using a binary matrix, where the presence of bands was represented by one and the absence by zero. Only the robust and unambiguous bands were considered in the analysis. The genetic diversity was assessed using several parameters, including the number of bands, polymorphic bands, private bands (bands found in only one population), and the number of distinct banding patterns (different combinations of bands). The presence of individuals with identical banding patterns significantly contributed to the observed low genetic diversity indices among the sampled populations in Kerala. The gel images of DNA amplifications displayed variations in the banding patterns (see Plate. 4.10.b). Among the evaluated sampled locations, Periyar (Ss-10), Thiruvananthapuram (Ss-11), and Tholpetty (Ss-4) exhibited higher genetic diversity and the most significant number of distinct banding patterns, followed by Ponkuzhy (Ss-1), Vythiri (Ss-2), and Muthanga (Ss-3) (see plate 4.10,a and 4.10.b).

A dendrogram based on binary matrix data was constructed to assess the genetic variations among all populations using the NTSYSpc.2.02 software with the UPGMA algorithm (table.4.30 and 4.31). This cluster analysis aimed to determine the similarities and dissimilarities between the populations. Although the results were obtained from a single primer, the genetic grouping of the studied populations in Kerala was determined based on the binary matrix data. This investigation will contribute to understanding the genetic diversity among *Senna spectabilis* accessions in Kerala using ISSR markers.

Two dendrograms were constructed based on similarity using the SIMQUAL method (Fig.4.34) and the dissimilarity table using the SIMGEND method (Fig.4.35), both implemented in NTSYSpc 2.02i. The binary data obtained from bands' presence (1) or absence (0) was utilized. The genetic grouping in the similarity dendrogram revealed that the populations formed two clusters initially. The first cluster was divided into two sub-clusters, with the first sub-cluster separating into two clades. The first clade consisted of the Ponkuzhy (Ss1) population, while the second clade included the Vythiri (Ss-2) and Muthanga (Ss-3) populations. In the second sub-cluster, the grouped populations were Tholpetty (Ss-4), Periyar (Ss-10), and Thiruvananthapuram (Ss-11). Regarding the second cluster, it was also divided into two sub-clusters. In the first sub-cluster, it was further divided into two, with Meppadi

(Ss-5) in one clade and Azhinjilam (Ss-6), Begur (Ss-7), and Anikkaty (Ss-8) in another clade. In the second sub-cluster, the Kottathara (Ss-9) population was grouped (Fig.4.34; table 4.30). The dissimilarity dendrogram yielded the same results (Fig.4.35; table-4.31).

The distribution of these populations based on this genetic grouping does not follow a similar distribution. It may be due to the conscious planting of the forest department throughout Kerala, as indicated by the history of the introduction. It has been found that the populations also have a similar age group, excluding the invaded range. The seedlings and saplings are continuously growing, and new individuals are joining the populations. Based on this observation, the likelihood of multiple introductions into the invaded areas of Kerala is low. The phenomenon of a species becoming invasive following multiple introductions often accounts for its initial establishment. Multiple introductions can lead to populations in the new range that exhibit a higher degree of genetic variability compared to populations in the native range (Kolbe *et al*., 2004; Maron *et al*., 2004).The prevention of founder effects, which result in genetic bottlenecks, is ensured through the occurrence of multiple introductions.

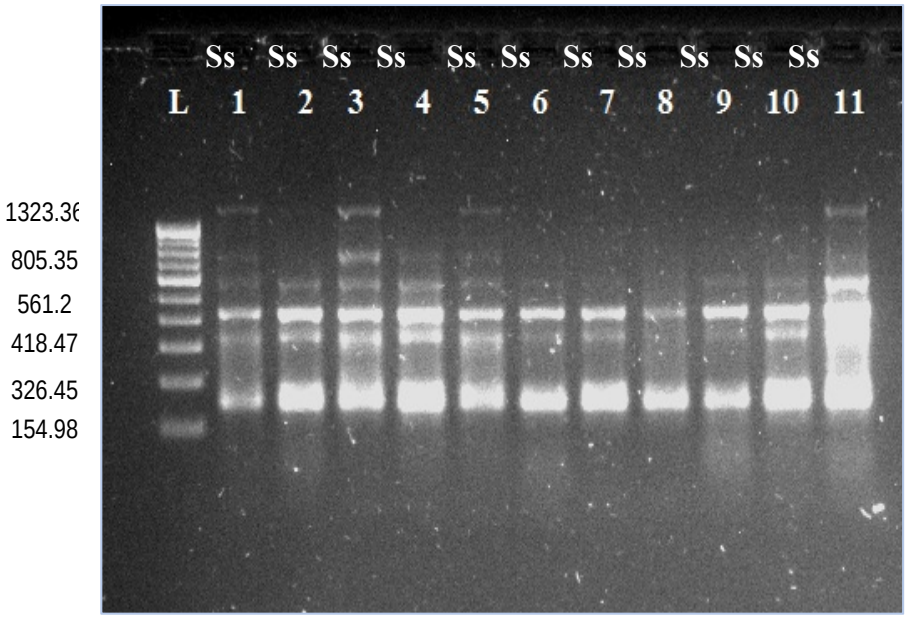


Plate .4.10.a. ISSR agarose gel profile of 11 *S.spectabilis* accessions

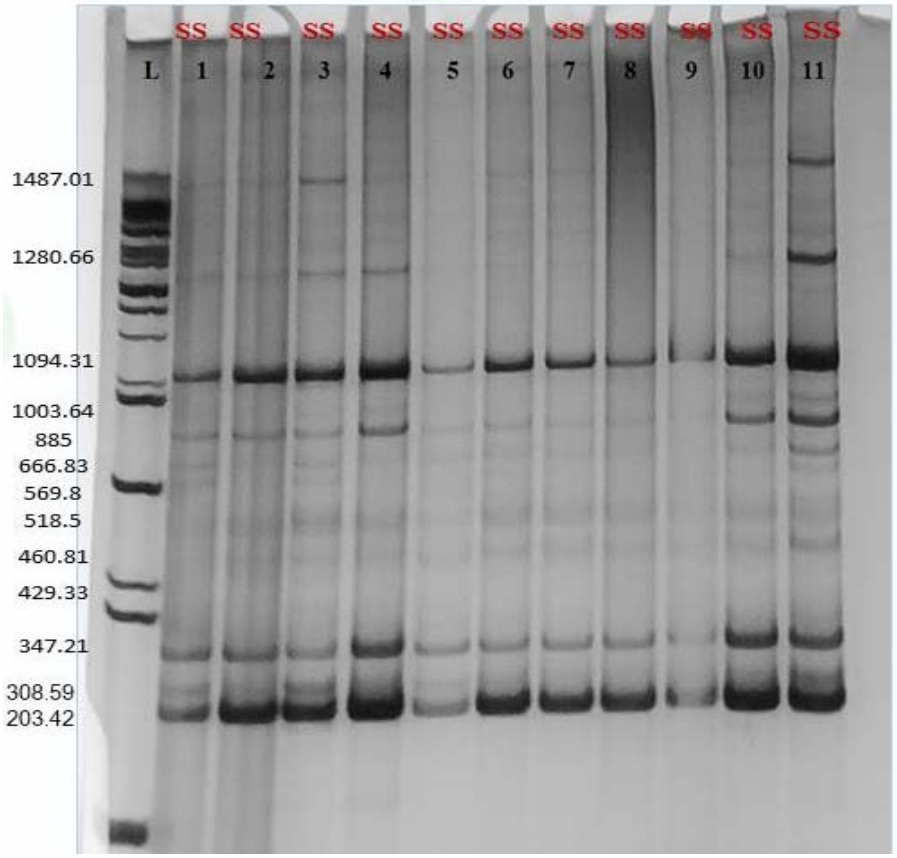


Plate.4.10.b. DNA amplification of ISSR – PAGE profile of 11 *S.spectabilis* accessions using $(CAG)_5$ primer

	Ss-1	Ss-2	Ss-3	Ss-4	Ss-5	Ss-6	Ss-7	Ss-8	Ss-9	Ss-10	Ss-11
Ss-1	1.0000										
Ss-2	0.7000	1.0000									
Ss-3	0.7273	0.8000	1.0000								
Ss-4	0.5833	0.6364	0.6667	1.0000							
Ss-5	0.3636	0.5556	0.6000	0.6000	1.0000						
Ss-6	0.4545	0.6667	0.7000	0.5455	0.8571	1.0000					
Ss-7	0.4545	0.6667	0.7000	0.5455	0.8571	1.0000	1.0000				
Ss-8	0.4545	0.6667	0.7000	0.5455	0.8571	1.0000	1.0000	1.0000			
Ss-9	0.3000	0.5000	0.4000	0.4000	0.6667	0.5714	0.5714	0.5714	1.0000		
Ss-10	0.5833	0.5000	0.5385	0.6667	0.4545	0.5455	0.5455	0.5455	0.2727	1.0000	
Ss-11	0.6000	0.5333	0.6667	0.6667	0.4000	0.4667	0.4667	0.4667	0.2667	0.6667	1.0000

Table 4.30 Similarity table of different populations based on SIMQUAL method using NTSYSpc 2.02i

	Ss-1	Ss-2	Ss-3	Ss4	Ss-5	Ss-6	Ss-7	Ss-8	Ss-9	Ss-10	Ss-11
Ss-1	0.0000										
Ss-2	0.1924	0.0000									
Ss-3	0.1705	0.1116	0.0000								
Ss-4	0.3040	0.2451	0.2231	0.0000							
Ss-5	0.6082	0.3262	0.2554	0.2554	0.0000						
Ss-6	0.4621	0.2209	0.1783	0.3325	0.0771	0.0000					
Ss-7	0.4621	0.2209	0.1783	0.3325	0.0771	0.0000	0.0000				
Ss-8	0.4621	0.2209	0.1783	0.3325	0.0771	0.0000	0.0000	0.0000			
Ss-9	0.6931	0.3466	0.4581	0.4581	0.2027	0.2798	0.2798	0.2798	0.0000		
Ss-10	0.3040	0.3993	0.3567	0.2231	0.4377	0.3325	0.3325	0.3325	0.7458	0.0000	
Ss-11	0.2554	0.3143	0.2027	0.2027	0.4581	0.3811	0.3811	0.3811	0.6609	0.2027	0.0000

Table 4.31 Dissimilarity table of different populations based on SIMGEND method using NTSYSpc 2.02i

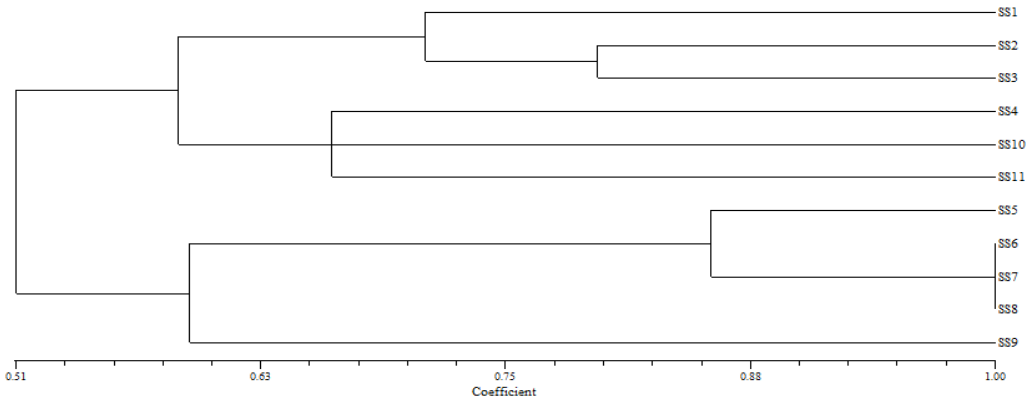


Fig. 4.34 Dendrogram based on Similarity of 11 *S. spectabilis accessions* by SIMQUAL method using NTSYSpc 2.02i**.**

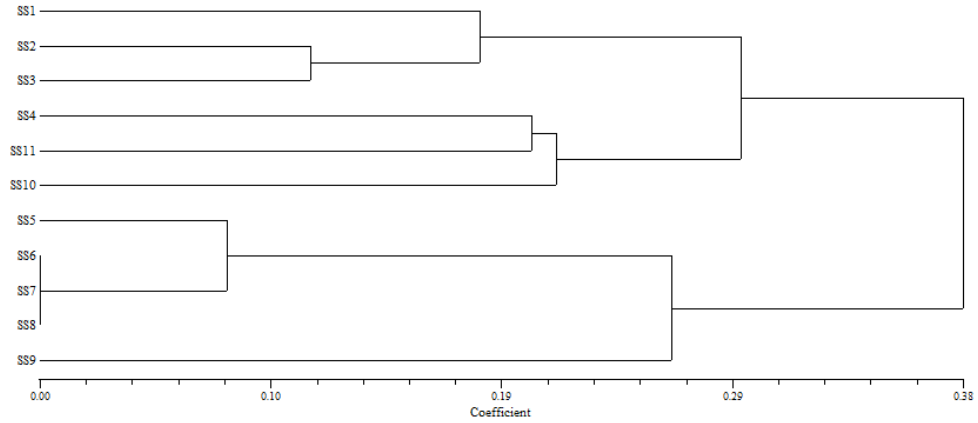


Fig. 4.35 Dendrogram based on dissimilarity of 11 *S. spectabilis* accessions by SIMGEND method using NTSYSpc 2.02i .Ss-1-Ponkuzhy; Ss-2- Vythiri; Ss-3-Muthanga; Ss-4-Tholpetty; Ss-5-Meppadi ;Ss-6-Azhinjilam; Ss-7-Begur; Ss-8-Anaikatty; Ss-9-Kottathara;Ss-10-Periyar;Ss-11-Thiruvananthapuram

This also allows for the retention of the necessary genetic diversity required to effectively respond to new selective challenges in the new range. Although our initial hypothesis suggested that the recent invasive spread of *Senna spectabilis* could be attributed to the recombination of genotypes from multiple introductions, our findings indicate that this may not be the case. Further clarification and a comprehensive genetic study are necessary to confirm this'

As per the findings of Santos *et al*., (2013), the determination of genetic diversity among accessions of *Senna spectabilis* was conducted by utilising RAPD markers. The study outcomes indicate that genetic variability exists in the accessions of *S. spectabilis* accessible in the germplasm collection of Embrapa Meio-Norte, located in

Northern Brazil. However, it is noteworthy that the genetic variability observed in these accessions is relatively limited. The analysis of genetic diversity and structure using ISSR molecular markers has revealed that the populations of *Senna spectabilis* under study exhibit low levels of intra-population genetic diversity and significant divergence. Factors such as self-fertilization, drift events, colonization by a limited number of individuals, varying selection pressures even within small geographic areas, and multiple introductions may have influenced the genetic diversity and distribution of *S. spectabilis* populations. It is important to note that this study represents a preliminary investigation into this invasive tree species, and it is strongly recommended that further research be conducted, encompassing a broader sample from different locations across the invaded populations.

Chapter 4.3

Development of Management Protocol for Controlling *Senna spectabilis* and Restoration Protocol using native species.

4.3.1. A comprehensive management strategy of *Senna spectabilis* tree invasion

The conducted management approaches in the experimental plots show significant results. Both mechanical and chemical methods were practised to contain *Senna spectabilis* at Muthanga and Tholpetty of Wayanad Wildlife Sanctuary in Wayanad district. One hector plot with the high infestation of *Senna spectabilis* was taken as a model site. This plot was divided into twenty-five subplots having 20m x 20m. Thirty-three experiments were conducted for pilot observations (Table 4.32). Among those, 23 successful treatments were selected, and experiments were replicated in Muthanga and Tholpetty regions. The observations are recorded at regular intervals.

Table 4.32 Different treatments applied on *Senna spectabilis* invaded area for pilot observation.

Description	Mechanical methods	Chemical methods
Seedlings/Saplings	1: Cut from ground level 2: Hand pulling	1. Foliar spraying of Herbicide 2. Slash weeding and apply chemicals
Medium-sized and multi-branched trees	3:Uprooting the whole tree using an excavator 4. Uprooting the whole tree manually. 5. Cut from ground level 6. Cut from 30 cm above ground 7. Ring barking 9. Cut from ground level every two months 10. Cut from the base and cover with soil 11. Cut the tree and bark stripping 12. Cut the tree from the base and split the scion portion	3. Hatch and squirt -1 ft wide above the ground and apply Herbicide 4. Root feeding (Petrol +Diesel+ Neem) 5. Cut from ground level & Apply Glyphosate 6. Cut from ground level & Apply $Cuso_4$ 7. Application of rock salt 8. Cut the tree and bark stripping –apply chemicals 9. Cut from the base cover with soil and apply chemicals
Large trees	13. Debarking at 1m around the tree (including the collar region and cutting the buttresses around the tree trunk) 14. Completely cut the tree at 50cm above the ground level and strip the bark, including the collar	10. Drill fill herbicide 11. Cut from ground level & Apply Glyphosate 12.Debarking & Apply $Cuso_4$ 13. Completely cut the tree at 50cm above the ground level, strip the bark, including the

	region 15. Uprooting the whole tree using JCB 16. Uprooting the whole tree by manually 17. Complete cut and fill soil 18. Cut the tree 1m above ground	collar region and apply Glyphosate 14. Cut the root and apply chemicals 15. Cut the branches and apply chemicals

The efficacy of controlling treatments was evaluated by assessing the mortality of trees and saplings and the cessation of regeneration or the formation of sprouts and barks in *Senna spectabilis*. Indicators of tree death include lack of foliage, flaking and peeling bark, and insufficient nutrient uptake, as evidenced by the absence of a green layer under the bark when scratched.

In the case of *S. spectabilis*, treatments such as debarking and ring barking typically result in foliage loss and dryness symptoms within a month, but the tree can regenerate its bark and sprouts after a month or with the onset of rain. To confirm the weather the tree dried, sapwood portions were sampled from the treated wood and subjected to anatomical studies to determine the presence of living tissues.

In experiment M1T (Mechanical Treatment in medium-sized and multi-branched trees), compared with other treatments, mechanical treatment 4(M1T4) and Mechanical treatment 3(M1T3) possess more mean value which indicates that these treatments are more promising, followed by M1T2 (Table. 4.33, Fig. 4.36). In experiment M2T (Mechanical treatments in large trees), treatments such as M2T2 and M2T1 were evident, followed by M2T3 (Table 4.33, Fig 4.37). M2T5 also indicated dryness of trees.

In experiment C1T (Chemical treatment in medium-sized and multi-branched trees), C1T3 is more effective, followed by C1T2 and C1T4 (Table 4.33, Fig. 4.36). In chemical methods for large trees (C2T) experiments, compared to other treatments, C2T1 is an effective control method, and C2T3 is also indicated as an effective killing method (Table 4.33, Fig 4.37). Experiments for seedlings and saplings combined with mechanical and chemical methods (ST) indicate that ST3 and ST2 were effective.

Experiments indicate that in mechanical methods, Uprooting the whole tree manually, Completely cutting the tree at 50cm above the ground level and stripping

the bark, including the collar region, completely cutting at ground level and filling soil and leaves are more effective for medium-sized and multi-branched trees.

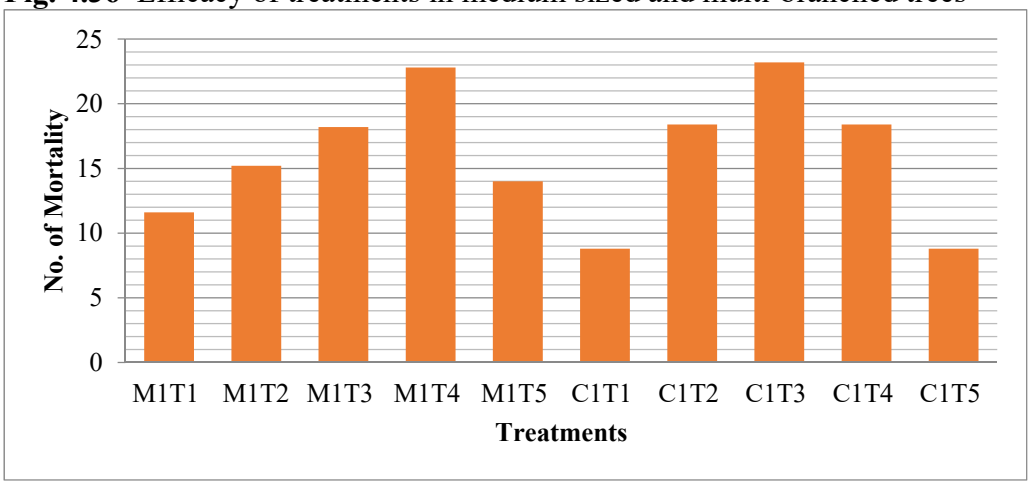
In large trees, uprooting the whole tree manually and debarking at 90cm around the tree (including the collar region and cutting the buttresses around the tree trunk) were effective. In chemical methods for medium-sized and multi-branched trees, Hatch and Squirt one foot wide above the ground and applying herbicide was very effective. In chemical methods for large trees, the hatch and squirt method and complete cut from ground level and application of Glyphosate are adequate to control measures compared to other treatments. It is a safe method when we take proper prerequisites to apply on a large scale. For seedlings and saplings, hand pulling/uprooting and foliar spray of 2- 4D + metribution 5:3/ litter effective strategies. A comparison of the treatments is presented in figures 4.36 and 4.37. *Senna spectabilis* aggressively invaded regions can be easily controlled by these methods.

Table 4.33 Comparison of different treatments applied on *Senna spectabilis* invaded area

Sl. No.	**Treatments**		**Mean ± Sd**
1	**$M1T_1$**	Debarking at 90cm around the tree (including collar region &cut the buttresses around the tree trunk)	11.60 ±1.41
2	**$M1T_2$**	Completely cut the tree at 50cm above the ground level and strip the bark including collar region	15.20 ±5.11
3	**$M1T_3$**	Uprooting the whole tree using excavator	18.20 ±3.03
4	**$M1T_4$**	Uprooting the whole tree by manually	22.80 ±4.32
5	**$M1T_5$**	Complete cut @ ground level and fill soil & leaves	14.00 ±3.80
6	**$M2T_1$**	Uprooting the whole tree using excavator	18.60 ±5.68
7	**$M2T_2$**	Uprooting the whole tree by manually	26.00 ±5.61
8	**$M2T_3$**	Cut from ground level and fill with soil	15.60 ±4.33
9	**$M2T_4$**	Cut from 30 cm above ground	3.20 ±3.63
10	**$M2T_5$**	Debarking at 90cm around the tree (including collar region &cut the buttresses around the tree trunk)	10.80 ±1.41
11	**ST_1**	Cut from ground level	223.60±5.3
12	**ST_2**	Hand pulling/Uprooting	421.60±7.2
13	**ST3**	Foliar spray of 2- 4D + Metribution 5:3/ litter	480±5.14
14	**$C1T_1$**	Drill fill herbicide	8.80 ±1.48
15	**$C1T_2$**	Cut from ground level & Apply Glyphosate	18.40 ±1.94

16	**$C1T_3$**	Hatch and squirt -1 ft wide above the ground and apply Herbicide	23.20 ±0.83
17	**$C1T_4$**	Completely cut the tree at 50cm above the ground level and strip the bark including collar region and apply Glyphosate	18.40 ±1.51
18	**$C1T_5$**	Root feeding (Petrol +Diesel+ Neem Oil)	8.80 ±1.49
19	**$C2T_1$**	Hatch and squirt -1 ft wide above the ground and apply Herbicide	24.80 ±1.78
20	**$C2T_2$**	Root feeding (Petrol +Diesel+Neem Oil)	14.40 ±1.82
21	**$C2T_3$**	Cut from ground level & Apply Glyphosate	19.20 ±1.64
22	**$C2T_4$**	Cut from ground level & Apply Cuso4	5.40 ±1.67
23	**$C2T_5$**	Drill fill herbicide	12.40±1.68

Fig. 4.36 Efficacy of treatments in medium sized and multi-branched trees



$M1T_1$ Debarking at 90cm around the tree (including collar region &cut the buttresses around the tree trunk)

$M1T_2$ Completely cut the tree at 50cm above the ground level and strip the bark including collar region

$M1T_3$ Uprooting the whole tree using excavator

$M1T_4$ Uprooting the whole tree by manually

$M1T_5$ Complete cut @ ground level and fill soil & leaves

$C1T_1$ Drill fill herbicide

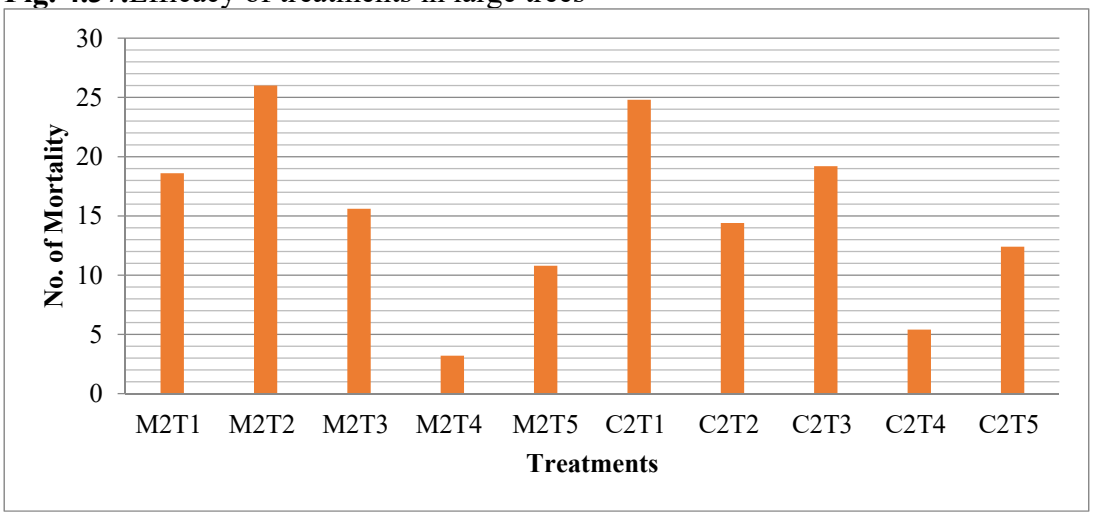
$C1T_2$ Cut from ground level & Apply Glyphosate

$C1T_3$ Hatch and squirt -1 ft wide above the ground and apply Herbicide

$C1T_4$ Completely cut the tree at 50cm above the ground level and strip the C1T5bark including collar region and apply Glyphosate

$C1T_5$ Root feeding (Petrol +Diesel+ Neem Oil)

Fig. 4.37. Efficacy of treatments in large trees



$M2T_1$ Uprooting the whole tree using excavator

$M2T_2$ Uprooting the whole tree by manually

$M2T_3$ Cut from ground level and fill with soil

$M2T_4$ Cut from 30 cm above ground

$M2T_5$ Debarking at 90cm around the tree (including collar region & cut the buttresses around the tree trunk)

$C2T_1$ Hatch and squirt -1 ft wide above the gr

$C2T_2$ Root feeding (Petrol +Diesel+Neem Oil

$C2T_3$ Cut from ground level & Apply Glypho

$C2T_4$ Cut from ground level & Apply Cuso4

$C2T_5$ Drill fill herbicide

4.3.1.1. An Integrated Management Protocol for *Senna spectabilis:* a workable Strategy

An urgent need exists for a practical containment protocol to control and manage *Senna spectabilis* in forest areas. However, a complete and viable strategy has yet to be developed. Developing a practical and pragmatic management strategy for controlling *S spectabilis* requires addressing issues related to the invasive species' habitat, management constraints, potential reinvasion, seed bank in the soil, regular monitoring, and restoration. This protocol includes the eradication of *Senna spectabilis*. Eradication, defined as removing every potentially reproducing individual of a species from an area of its population density below sustainable levels, represents the optimal management option. However, success is likely to be limited to the earliest stages of an invasion, in a manageable size, primarily during the lag phase. For ecologists to investigate the functions of species within communities, the effects of Invasive Alien Species, and the behaviour and population dynamics of exotic species targeted for eradication, it is imperative that this knowledge serves as the fundamental basis for their research design. Each treatment should be regarded as an ecological experiment, with future and ongoing programs presenting exceptional research opportunities. Through a continuous exchange of ideas and the integration of expertise between scientists and managers, novel and more suitable strategies can be developed to restore the ecosystem status of the *Senna spectabilis* that invaded the region.

A three-tier eradication strategy protocol has been developed to address the threat of invasive tree species. This protocol aims to assess the impact of the invader on forest productivity and ecological damage, as well as implement, monitor, and evaluate a management program. The effectiveness of these treatments relies on the physiological characteristics and ecological behaviour of the target species.

The management protocol has been tailored to the age and growth forms of *Senna spectabilis*, as outlined in Table 4.34 and 4.35. For protected areas, such as ecologically fragile or forest lands, protocol one recommends using mechanical methods. Protocol two suggests combining mechanical and chemical methods in non-protected or recommended areas.

The methods employed in this protocol include (i) the implementation of appropriate pre-requisite strategies to contain the species, (ii) the adoption of effective strategies for containment and eradication, including the mechanical removal of *S spectabilis* from forest areas and the reduction of their population size through the use of herbicides, and (iii) a post-containment protocol involving habitat management through ongoing monitoring and evaluation, followed by eco-restoration using indigenous species.

The eradication program for the species in forest areas/protected areas commences at the fundamental forest management unit, namely the beat level. The systematic removal of all individuals through selected treatment from a chosen beat is preferred over a broad approach. The initial step is to identify isolated low-density regions, as this makes it easier to prevent further spread. The eradication program comprises treatments suitable for the selected patch based on its girth class. The general approach for controlling and restoring the invaded region is presented in Tables 4.34 and 4.35.

The ensuing techniques have been chosen for the management protocol to regulate the invasion of *S. spectabilis*.

(i) Eradication of Seedlings and saplings of *Senna spectabilis*

Hand pulling/Uprooting the seedlings: The seedlings of herbaceous nature, up to approximately one inch in diameter, must be removed either by pulling or uprooting by hand. In the case of saplings (<10 cm), removing as much of the root system as possible by hand or using a digging fork, crow bar, or similar tool is imperative. Even a small portion of the root system can lead to the re-growth of the infestation. (Plate.4.16) If the roots are entirely removed, eradication is possible. However, hand pulling may disturb the soil, creating a seedbed for other invasive plants or the same *S. spectabilis* seeds that may be established in the area. This issue can be mitigated by firming the soil with tools or boots, covering the area with leaf litter, and, if possible, broadcasting seeds of native fast-growing plants. In areas invaded by *S. spectabilis*, a high rate of seed bank is present in the soil, and the seeds will germinate from the first summer rain shower onwards. To eradicate this species, it is necessary to quickly detect and prevent the growth of newly germinated seedlings using the hand-pulling method and to prevent their spread. A continuous three-week monitoring period is

necessary for detecting newly germinated seedlings. For complete eradication of this species, a three-year monthly observation for detecting seedlings is required based on this protocol. Frequent monitoring is necessary during the post-monsoon period, and continuous, intermittent observation is needed throughout the rainy season. Regular monitoring should begin from the first summer shower onwards.

Uprooted plant material should be disposed of safely (Plate.4.16.b) because this species produces allelochemicals that prevent the growth of other indigenous species. The uprooted stem material will not re-sprout, but proper disposal must be ensured.

Table 4.34 Three tier management Protocol – I : Protected areas:-Mechanical methods

Growth form	**Pre-requisites for eradication programme**	**Workable strategies for containment & eradication of the species**	**Post-containment protocol**
Seedlings & Saplings	1. Mark the designated area prior to commencing eradication efforts. 2. Ensure that the necessary tools are readily available and that hand gloves are worn. 3. The eradication program is scheduled to take place from July to November, prior to the post-monsoon period	1.Uproot the seedlings (Digging fork/crow bar/weed puller) 2.Hand pulling the seedlings ≤10cm	1. Compact the soil using appropriate tools such as boots or forks. 2. Safely dispose of uprooted materials to prevent any potential regrowth. 3. Conduct intermittent monitoring for three weeks.
Medium sized(<30) & Multi branched trees	1. Choose and designate the area according to the density classification, giving priority to areas with low density. 2.Implementthe treatments during the period from February to May, which corresponds to the pre-monsoon season.	1.Uprooting the whole tree by manually. 2.Completely cut the tree at 50cm above the ground level and strip the bark including collar region. 3.Complete cut at ground level and fill with soil &leaves.	1. The uprooted area should be thoroughly filled with soil and firmly compacted using a spade, ensuring that the cut ends of the roots are not exposed. 2. Regular monitoring for three weeks, followed by recurrent monitoring for one year.

Large trees>30 cm	1. Select the specific region for the implementation of treatments. 2. The application of treatments should be carried out during the pre-monsoon period.	1. Debarking at 90cm around the tree (including the collar region and cut the buttresses around the tree trunk and firm with soil) 2. Complete cut at ground level and fill with soil and leaves. 3. Uprooting the whole tree by manually &fill with soil	1.Four weeks of consistent evaluation, followed by 18 months of recurrent monitoring, is recommended to prevent the reformation and re-sprouting of bark in the process of Debarking. 2. To ensure the complete uprooting and cutting of a tree, it is advised to conduct three weeks of regular monitoring, followed by one year of recurrent monitoring.

Table 4.35 Three tier management Protocol – II : Non-Protected areas(road sides abandon lands& private lands)

Growth form	**Pre-requisites for eradication programme**	**Workable strategies for containment & eradication of the species**	**Post-containment protocol**
Seedlings & Saplings	1. Prior to eradication, it is imperative to demarcate the designated area. 2. It is mandatory to wear protective hand gloves and a mask during the eradication process. 3.The eradication program has been scheduled for implementation between October and December.	1.Foliar spray to the seedlings using 2-4D + Metribuzn (5:3) at 1 liter (Knapsack Sprayer/Squirt bottle) 2.Hand pulling the herbaceous seedlings $\leq$10cm	1. It is recommended to prevent any spillage and ensure that actions are specific to the intended target. 2. Disposal of uprooted materials should be carried out safely. 3. It is advisable to evaluate the area after one week.

Medium sized(<30) & Multi branched trees	1. Choose the area according to the density classification, giving preference to those with a low density class. 2. Implement eradication measures between February and May, specifically during the pre-monsoon period, while avoiding days with rain forecasts.	1.Hatch and squirt -1 ft wide above the ground and apply glyphosate 45% /Paraquat dichloride (5ml In1-liter water). 2. Completely cut the tree at 50cm above the ground level and strip the bark, including the collar region.	1. Utilizing a syringe to prevent spillage and ensure precision in application is recommended. 2. The targeted trees should be assessed after four weeks, followed by recurrent monitoring at intervals of 24 weeks.
Large trees>30cm	1. Mark the designated area for the implementation of treatments. 2. Administer the prescribed treatments prior to the onset of the monsoon season and refrain from applying herbicides on days when rainfall is predicted.	1. Uprooting the whole tree manually and fill it with soil. 2. Complete cut at ground level and fill with soil and leaves. 3.Hatch and squirt -1 ft wide above the ground and apply glyphosate 45% / Paraquat dichloride (5ml In1liter %urea). 4..Drill fill 3ml of 45% glycil around the basal portion with a syringe and driller through the bark	1. Assess the designated trees after four weeks, followed by an additional ten weeks of recurring monitoring after the herbicide treatments. 2. Conduct regular monitoring for three weeks, and subsequently implement a recurrent monitoring plan for uprooting and removing the tree over one year.

b) Foliar spray to the seedlings using herbicides

During the process of manual weeding, the roots of the seedlings remain in the soil, increasing the likelihood of regrowth and leading to the failure of eradication programs. Herbicides are useful when dealing with larger-scale operations and have proven effective in managing weed populations (Tuinstra *et al*., 2009; Rajcan and Swanton, 2001; Khaliq *et al.,* 2011). After applying several herbicides, it was determined that most harm the ecosystem. However, some herbicides were found to be comparatively less harmful and more effective, and these were selected for foliar spray treatments. These treatments involved a combination of two herbicides, namely 2,4-D (2,4-D amine salt) and Metribuzin, in a 5:3 ratio mixed with one litre of water.

The variations in weed densities observed among the different treatments can be attributed to the specific properties of each herbicide, including solubility in the soil, volatilization, photodegradation, microbial breakdown, persistence, and weed tolerance. The post-emergence herbicide 2,4-D amine salt was found to be particularly effective in reducing weed density compared to manual weeding, and its use can be considered based on economic advantages and convenience (Chepkoech *et al.,* 2021). On the other hand, Metribuzin has a high solubility of 1165 mg/L (GRDC, 2015). Metribuzin, on the other hand, is a systemic herbicide used as a pre-and early post-emergence treatment. It has a broad-spectrum effect on annual broad-leaved and grass weeds and has been successfully used to control weeds in crops such as maize, potato, tomato, and soybean (Volova *et al.,* 2020). 2,4-D is known for its effectiveness in eliminating broad-leaved weeds and is one of the oldest herbicides widely used for controlling such weeds in cereal crops (Peterson and Hulting, 2004).

The application of herbicides for weed control has resulted in a significant reduction in the density of *S. spectabilis* and the biomass of seedlings. It is essential to strictly adhere to the prerequisites for herbicide application to ensure the effectiveness of these treatments and safety to the application regions.

(ii) Methods of management of medium-sized (< 30cm GBH) and multi - branched trees

a) Uprooting the whole tree manually: Initially, selecting a severely infested patch of medium-sized and multi-branched trees is imperative. Observations of the present study indicate that *S. spectabilis* trees are typically multi-branched, which

makes controlling them through debarking and manual cutting a tedious task. Once the selection process is complete, the trees should be cut using a chain saw, hand saw, loppers, or a cutting knife, and the plants should be lopped off at ground level. (Plate.4.12.a).

Subsequently, the soil around the tree trunk (1.3 cm ground line radios from the tree trunk) should be dug up, and the root zone should be uprooted as much as possible, with the root fragments detached. This process is necessary to damage the underground carbohydrate storage structures, including the taproot zone of the tree. After complete removal, the cleared portion should be filled with soil. Since *S. spectabilis* possesses large roots, it is nearly impossible to deplete root energy reserves with repeated cutting, so strict monitoring is required for the uprooted patch for up to two years.

Regular evaluation is necessary during the first rainy season to prevent re-sprouting from any remaining root fragments exposed to sunlight. This option can be adopted if adequate financial support and labour are available, as it is labour-intensive, expensive, and requires strict monitoring. If this option is not feasible, the next method should be pursued.

a) Cut from ground level and cover with soil and leaves: Select the patch of *S. spectabilis* invaded area. Machetes, axes, or chainsaws are recommended for cutting down trees. The cuts should be made as close to the ground as possible to remove most stem buds effectively. If feasible, these tools should also cut the root buttresses or crown. Afterwards, the cut end of the tree should be covered with nearby soil up to a distance of 30 cm from the above ground level (Plate..4.12b). The soil should be firmly packed down using tools and any surrounding leaves. This layer of soil will act as a barrier, preventing sunlight exposure to the cut end of the trees. It has been observed that elephants tend to approach and disturb the treated area in forested areas, removing the soil. It is advisable to place some dried leaves on top to conceal the treated area.

Typically, prolific sprouting occurs within five days in the case of *S. spectabilis*. However, our experiment has shown that this barrier method reduces the sprouting rate and gradually kills the tree. It is important to continue cutting when re-sprouts appear, repeating this process until no further regrowth occurs. Strict monitoring is necessary for the success of this treatment. The most effective time for cutting is during the hot summer months when the root reserves are at their lowest, and the soil

is dry. Mechanical treatments alone are insufficient in cases of severe and widespread invasion of *S. spectabilis* in a larger area. These treatments require intense labour and significant funds, and there is also a risk of re-invasion by this species. Applying biocides such as herbicides in a controlled and specific manner for this species is recommended for effective eradication. The following treatments have proven to be significant in managing this species.

c) Hatch and squirt: In this method, an initial incision is made in the lower portions of the tree trunk. For medium-sized *S. spectabilis* trees, three or four evenly spaced cuts around the woody stem are sufficient. These cuts can be made using a hatchet, axe, machete, or cane knife. Once the cuts are made, the concentrated herbicide (Glyphosate) should be immediately applied to the exposed area of the cut end within 15 seconds (Plate..4.14).The cuts should be approximately 1-1.5 inches deep, primarily affecting the cambial layer. It is important to avoid making multiple cuts directly above or below each other, as this can hinder the movement of the herbicide within the stem. Instead, make an incision around the stem. To apply the herbicide, use a handheld, chemical-resistant 10 ml syringe and apply 2 ml of concentrated herbicide to each incision. The amount of glyphosate used will depend on the size of the cut and how much it can hold without spilling onto the bark, but the minimum amount should be 2 ml. using a syringe helps minimize spillage of the herbicide onto the soil or other surrounding plant species. The glyphosate should remain within the incision cut. The cuts should be made on the stem up to 1 foot above ground level. It is advisable to avoid using this method if rainfall is predicted within 48 hours, as prolonged exposure to water can hinder herbicide absorption. Similarly, prolonged cold temperatures can cause the herbicide to freeze in the cut, resulting in poor absorption. Heavy spring sap flow can wash away the herbicide from the incision cuts, leading to poor control and potential transfer to non-target plants in the soil. Additionally, prolonged and severe drought conditions are also ineffective for this method. However, this method is highly selective in managing the targeted Senna tree species with minimal damage to the ecosystem. It is also a fast and cost-effective approach. After applying glyphosate to the targeted Senna trees, the herbicide's activity can be observed by disrupting typical phenology, such as yellowing foliage and gradually drying off. The timeframe for these effects can range from one to three months, depending on environmental conditions.

iii) Methods of removing large trees (> 30cm GBH)

a) Debarking

It is an effective technique for large *Senna spectabilis* trees with a single trunk. In comparison to other methods of tree removal, it causes minimal disruption to the surrounding environment and does not disturb the soil (Plate. 4.15). The process involves cutting a strip of bark 90 cm wide along the entire length of the tree trunk. This strip must be cut deep enough to remove the vascular cambium, or inner bark responsible for transporting sugars and carbohydrates between different parts of the tree. The inner cambium layer is also responsible for producing new wood and bark. The main challenge in debarking is maintaining the connectivity of the bark. After one week of debarking, the tree may exhibit wilting symptoms during the summer days. However, within a month, bridge-like bark connectivity(Plate.4.18.e) or re-sprouts can be observed from the collar region of the trees. Sapwood samples were collected from above and between the debarked areas to confirm the presence of living tissues or a functioning conducting system in the treated tree. Anatomical studies were conducted, and Plate.4.17 shows that the inner layer persists in the debarked trees even when wilting symptoms are present.

In order to remove the bark from a tree, it is recommended to make parallel cuts around the tree's circumference, with a distance of approximately 3 inches or more between each cut. These cuts can be made using a knife, axe, or saw and should be slightly deeper than the cambium layer. For debarking, the trunk should be struck sharply between the cuts using the back of an axe or other blunt object. This will cause the bark to come off in large pieces, effectively preventing any further growth of the tree.

It is important to exercise caution when making these cuts, as cutting too deeply into the trunk may cause the tree to snap and fall during high winds. To determine the depth of the cambium layer, it is recommended to make two short test cuts and strike the bark between them. After several strikes, the bark should come off intact, exposing the cambium and wood (xylem) below.

When debarking a tree, it is important to extend the cuts to include the collar region of the tree and the buttresses around the trunk. Failure to do so may result in

the accumulation of food materials at the exposed bark ends, leading to the formation of several new sprouts.

b) Completely cut the tree at 50cm above the ground level and strip the bark, including collar region

In comparison to the process of debarking, this method is significantly faster and more effective. Typically used for tree eradication, we cut down and remove the tree. (Plate.4.13)However, in the case of *Senna spectabilis*, after being cut down, the tree regrows vigorously from the cut end within a week, forming a multi-branched tree within three months. To address this issue, we have developed a mechanical method of tree extermination by cutting the tree and stripping the bark. The bark is stripped from the tree up to the basal portion or collar region, 50 cm above ground level. Soil is then filled at the cut ends of the collar region to prevent exposure to sunlight and reduce regrowth. When stripping the bark, we follow the same procedures as in debarking, which involve removing the vascular cambium or inner bark, a thin layer of living tissue. After completely removing the bark, we make slanting cuts around the tree trunk or stock to ensure and prevent further sprouting.

c) Drill fill

Please choose the trees to be eliminated. Utilize a drilling technique to create holes with a 5-10 mm diameter through the bark and cambium of the trunk (Plate. 4.11.a). These holes should be spaced approximately 5-6 cm apart around the trunk, forming a 300-degree angle of slant. The drilling should be done below the lowest living branch and 1 foot above ground level. Fill these holes with undiluted glyphosate using a syringe. For large trees measuring over 30 cm, creating 10-15 holes is necessary. The drilling can be performed using a battery-operated or manual drill bit. The drilling method offers the advantage of minimal herbicide leakage into the soil.

d) Hatch and squirt for large trees

The technique of using a hatchet or axe to cut the lower sections of the tree trunk is employed in the same manner for medium-sized trees. However, there is a distinction regarding large trees, as many cut ends are required (Plate.4.14). It is necessary to disrupt the translocation of food materials and allow the active ingredient to spread throughout the tree, ultimately destroying leaves and the root system.

Consequently, the tree is unable to regenerate. A concentrated herbicide, measuring 2 ml, is administered through a syringe into the incision. This application is carefully timed to coincide with the active growing season of *S. spectasbilis*. Following this treatment, a strict monitoring process is implemented for three months.

4.3.1.2. Prerequisites, impacts, and application methodology of herbicides.

When managing areas invaded by trees, several essential factors must be considered, including efficacy, cost-effectiveness, environmental disruption, and time. The most effective approach for the treatment of woody stems is to apply treatments during the late summer and autumn, specifically between late August and November. Cut the stems as close to the ground as possible and promptly apply herbicide directly to the cut surface. Delaying the application will result in reduced effectiveness of the treatment (Table 4.34 and 4.35).

Various methods can be used to apply the herbicide, such as a sponge, paintbrush, syringe, or Knapsack Sprayer/squirt bottle. In this case, we have standardized Glyphosate as the herbicide to control the tree forms of *S. spectabilis*. Glyphosate is a broad-spectrum herbicide that has been extensively utilized since its commercialization in 1974 (Franz *et al.,* 1997). It is commonly employed in the agricultural sector (Dill *et al.,* 2010). The chemical name of this herbicide is N-(Phosphonomethyl) glycine, which is derived from the amino acid glycine and phosphonic acid.

Herbicides are comprised of three primary constituents, specifically the active ingredient glyphosate, water, and a surfactant blend possessing soap-like properties. This surfactant blend serves to facilitate the adherence and penetration of the active ingredient onto the leaves or the area of application. Following the application to plant foliage and subsequent absorption by the leaves, glyphosate is transported through the phloem to actively growing tissues, such as roots and storage organs (Duke and Powles, 2008). Glyphosate is absorbed and translocated throughout the various tissues of the plant, with the surfactant augmenting its delivery. Once within the plant, glyphosate hampers the activity of EPSP synthase, an enzyme that hinders the plant's ability to produce specific essential amino acids necessary for growth and survival (Franz *et al.,* 1997).Notably, humans and animals do not utilize EPSP synthase, which

mitigates any unreasonable toxicity concerns associated with glyphosate use when applied according to label directions.

In order to ensure the safety of an herbicide, the United States Environmental Protection Agency conducts a comprehensive risk assessment. Additionally, various regulatory bodies and scientific organizations, such as the European Commission, Health Canada, and the World Health Organization, have thoroughly examined data on glyphosate.

Over the course of the past four decades, glyphosate has undergone extensive toxicology testing. Multiple registering companies from around the world have generated at least six distinct sets of toxicology data, resulting in over 300 individual toxicology studies. These data consistently indicate no cause for concern regarding developmental and reproductive toxicity, carcinogenicity, genotoxicity, endocrine disruption potential, neurotoxicity, and immunotoxicity. Based on these findings, the European Commission's Health and Consumer Protection Directorate-General recently re-registered glyphosate in Europe in 2014.

Glyphosate has been demonstrated to lack the ability to induce acute or chronic toxicity in higher organisms, encompassing wild mammals, birds, fish, aquatic invertebrates, as well as terrestrial invertebrates like earthworms and honey bees, even when exposed to environmentally plausible levels. Correspondingly, when utilized appropriately and within realistic thresholds, glyphosate-based formulations do not engender undesirable detrimental consequences on wildlife. Nevertheless, notwithstanding these established conclusions, certain accusations persist, necessitating a cautious approach towards the application of glyphosate in delicate ecosystems, such as wildlife sanctuaries.

Glyphosate only affects non-target plants when there is direct contact during application. Target-specific methods such as hatch, squirt, and drill fill are employed to mitigate this risk. For applicator safety, proper hand gloves and respiratory masks must be worn during these treatments. The dosage of glyphosate typically depends on various factors. During hatch and squirt treatment, we used different dilutions of the herbicide, such as 1:1, 3:1, and concentrated solutions.

Concentrated glyphosate followed by a 1:1 dilution proved to be more effective. Some studies suggest that applying a concentrated dose of herbicide to frilled regions at the beginning of the growing season can kill developing cambium cells, hindering the translocation of the poison and reducing its effectiveness (Geldenhuys, 1982). However, the concentrated solution demonstrated significant effects in our treatments for *S. spectabilis* trees and led to mortality.

4.3.1.3. The potential for re-invasion.

The management of tree invasion is an arduous task that requires strict monitoring. In the case of *S. spectabilis*, the situation is particularly critical as eradicating and managing its invasion and regrowth poses significant challenges. In the Wayanad wildlife sanctuary, where the Senna invasion is rampant, we have attempted over 50 eradication and management treatments in collaboration with forest department officials over the past decade.

However, despite the treatment, the *S. spectabilis* continues to re-sprout and regrow. While some treatments have shown initial success and sudden mortality rates, new shoots and sprouts emerge from the remnants of cut-down or treated trees (Plate.4.18.f & g). For instance, uprooting the entire tree using an excavator has proven to be a more effective and efficient eradication method. However, in our study area, where selected patches of Senna invasion were uprooted in this manner without scientific study or background knowledge, reinvasion was detected after the first summer shower. New sprouts emerged from the exposed areas of the root portion or collar region, proliferating and spreading more aggressively than before.

Additionally, uprooting by excavators severely disrupts the soil condition and the ecosystem, making it an impractical method in a fragile ecosystem (Plate 4.11.e). Similarly, traditional methods such as debarking and ring barking, which typically result in drying ordinary trees, do not have the same effect on *S spectabilis*. In this case, a connecting bridge forms from the remaining cambial or vascular layer between the ring-barked portions, allowing the tree to regenerate and become multi-branched (Plate.4.18 a-b)

The successful debarking of trees relies on properly removing bark, including the cambial layer (as depicted in Plate 4.15.b). In the case of multi-branched trees, when using glyphosate for cut sump treatment, it is observed that the tree appears dried

within the first three months. However, new sprouts emerge abnormally from the collar region after being showered (Plate.4.11.f &g). Additionally, cut stump treatment has a higher risk of glyphosate spillage into the surrounding area. While cutting the tree at ground level and applying glyphosate proves to be a successful method, the spillage issue remains a concern, making it impractical for field application. Complete cutting of the tree is not feasible due to the rapid growth of prolific sprouts within a week, forming a sizeable multi-branched tree with a broad canopy within six months. Traditional methods, such as salt application to the cut end of the tree, have been attempted for complete tree eradication (Plate.4.11.b). However, these efforts have yielded negative results in the study area, as it is part of a wildlife foraging area. Elephants, in particular, have wiped out the salt and even dug up portions of the tree at its collar region. Numerous challenges have been encountered while eradicating the *S.spectabilis* tree invasions.

4.3.1.4. Monitoring and evaluation of control treatments in *Senna spectabilis* invaded area

In the case of the *Senna spectabilis* invasion eradication protocol, potential methods for eradication and control were only successful after a strict three-year monitoring period. Following the application of each treatment, regular monitoring is required for three weeks, followed by weekly monitoring for one year. Subsequently, monthly observations are necessary for three years. It is crucial to carefully observe the treated area that has been invaded by the tree, particularly for any re-sprouting from the treated trees. If any signs of re-sprouting are observed, immediate removal is necessary. Additionally, any new seedlings in the invaded area should be uprooted immediately.

To ensure the eradication programme's success, it involves forest watchers, social groups such as natural clubs, tourists, trekking groups, school student groups like NCC and NSS volunteers, participants in natural camps, social forestry groups, and tribal participatory programs. These groups should be incorporated into the campaign through skill training, as their support is crucial. However, before their involvement, they must clearly understand the tree's nature, its invasive behaviour, and the possibility of re-invasion. Implementing the treatments outlined in the management protocol, conducting regular monitoring, and executing a comprehensive eradication

campaign can effectively manage the invasion of Senna spectabilis within a specified timeframe.

4.3.2. Restoration techniques for areas affected by the invasion of *Senna spectabilis*.

Thirteen indigenous species were carefully chosen for their suitability in the respective regions and subsequently planted for restoration experiments at Muthanga within the WWL sanctuary, where the *S. spectabilis* had been eradicated. Different models of planting methods were tested (Table 4.37. Plate.4.19). Nine months after planting, the survival percentage of thirteen indigenous species at the restoration area is being assessed.

Table 4.36 Three tier management Protocol – I : Protected areas:-Mechanical methods

Growth form	**Pre-requisites for eradication programme**	**Workable strategies for containment & eradication of the species**	**Post-containment protocol**
Seedlings & Saplings	1. Mark the designated area prior to commencing eradication efforts. 2. Ensure that the necessary tools are readily available and that hand gloves are worn. 3. The eradication program is scheduled to take place from July to November, prior to the post-monsoon period	1.Uproot the seedlings (Digging fork/crow bar/weed puller) 2.Hand pulling the seedlings ≤10cm	1. Compact the soil using appropriate tools such as boots or forks. 2. Safely dispose of uprooted materials to prevent any potential regrowth. 3. Conduct intermittent monitoring for three weeks.
Medium sized(<30) & Multi branched trees	1. Choose and designate the area according to the density classification, giving priority to areas with low density. 2.Implementthe treatments during the period from February to May, which corresponds to the pre-monsoon season.	1.Uprooting the whole tree by manually. 2.Completely cut the tree at 50cm above the ground level and strip the bark including collar region. 3.Complete cut at ground level and fill with soil &leaves.	1. The uprooted area should be thoroughly filled with soil and firmly compacted using a spade, ensuring that the cut ends of the roots are not exposed. 2. Regular monitoring for three weeks, followed by recurrent monitoring for one year.
Large trees>30cm	1. Select the specific region for the implementation of treatments. 2. The application of treatments should be carried out during the pre-monsoon period.	1. Debarking at 90cm around the tree (including the collar region and cut the buttresses around the tree trunk and firm with soil) 2. Complete cut at ground level and fill with soil and leaves. 3. Uprooting the whole tree by manually &fill with soil	1.Four weeks of consistent evaluation, followed by 18 months of recurrent monitoring, is recommended to prevent the reformation and re-sprouting of bark in the process of Debarking. 2. To ensure the complete uprooting and cutting of a tree, it is advised to conduct three weeks of regular monitoring, followed by one year of recurrent monitoring.

Table 4.37 Three tier management Protocol – II : Non-Protected areas(road sides abandon lands& private lands)

Growth form	**Pre-requisites for eradication programme**	**Workable strategies for containment & eradication of the species**	**Post-containment protocol**
Seedlings & Saplings	1. Prior to eradication, it is imperative to demarcate the designated area. 2. It is mandatory to wear protective hand gloves and a mask during the eradication process. 3.The eradication program has been scheduled for implementation between October and December.	1.Foliar spray to the seedlings using 2-4D + Metribuzn (5:3) at 1 liter (Knapsack Sprayer/Squirt bottle) 2.Hand pulling the herbaceous seedlings ≤10cm	1. It is recommended to prevent any spillage and ensure that actions are specific to the intended target. 2. Disposal of uprooted materials should be carried out safely. 3. It is advisable to evaluate the area after one week.
Medium sized (<30) & Multi branched trees	1. Choose the area according to the density classification, giving preference to those with a low density class. 2. Implement eradication measures between February and May, specifically during the pre-monsoon period, while avoiding days with rain forecasts.	1.Hatch and squirt -1 ft wide above the ground and apply glyphosate 45% /Paraquat dichloride(5 ml in 1-litre water). 2. Completely cut the tree at 50cm above the ground level and strip the bark, including the collar region.	1. Utilizing a syringe to prevent spillage and ensure precision in application is recommended. 2. The targeted trees should be assessed after four weeks, followed by recurrent monitoring at intervals of 24 weeks.

Large trees>30cm	1. Mark the designated area for the implementation of treatments. 2. Administer the prescribed treatments prior to the onset of the monsoon season and refrain from applying herbicides on days when rainfall is predicted.	1. Uprooting the whole tree manually and fill it with soil. 2. Complete cut at ground level and fill with soil and leaves. 3.Hatch and squirt -1 ft wide above the ground and apply glyphosate 45% / Paraquat dichloride (5 ml in 1 liter % urea). 4..Drill fill 3ml of 45% glycil around the basal portion with a syringe and driller through the bark	1. Assess the designated trees after four weeks, followed by an additional ten weeks of recurring monitoring after the herbicide treatments. 2. Conduct regular monitoring for three weeks, and subsequently implement a recurrent monitoring plan for uprooting and removing the tree over one year.

b) Foliar spray to the seedlings using herbicides

During the process of manual weeding, the roots of the seedlings remain in the soil, increasing the likelihood of regrowth and leading to the failure of eradication programs. Herbicides are useful when dealing with larger-scale operations and have proven effective in managing weed populations (Tuinstra *et al.,* 2009; Rajcan and Swanton, 2001; Khaliq *et al.,* 2011). After applying several herbicides, it was determined that most harm the ecosystem. However, some herbicides were found to be comparatively less harmful and more effective, and these were selected for foliar spray treatments. These treatments involved a combination of two herbicides, namely 2,4-D (2,4-D amine salt) and Metribuzin, in a 5:3 ratio mixed with one litre of water.

The variations in weed densities observed among the different treatments can be attributed to the specific properties of each herbicide, including solubility in the soil, volatilization, photodegradation, microbial breakdown, persistence, and weed tolerance. The post-emergence herbicide 2,4-D amine salt was found to be particularly

effective in reducing weed density compared to manual weeding, and its use can be considered based on economic advantages and convenience (Chepkoech *et al.,* 2021). On the other hand, Metribuzin has a high solubility of 1165 mg/L (GRDC, 2015). Metribuzin, on the other hand, is a systemic herbicide used as a pre-and early post-emergence treatment. It has a broad-spectrum effect on annual broad-leaved and grass weeds and has been successfully used to control weeds in crops such as maize, potato, tomato, and soybean (Volova *et al.,* 2020). 2,4-D is known for its effectiveness in eliminating broad-leaved weeds and is one of the oldest herbicides widely used for controlling such weeds in cereal crops (Peterson and Hulting, 2004).

The application of herbicides for weed control has resulted in a significant reduction in the density of *S. spectabilis* and the biomass of seedlings. It is essential to strictly adhere to the prerequisites for herbicide application to ensure the effectiveness of these treatments and safety to the application regions.

(ii) Methods of management of medium-sized (< 30cm GBH) and multi - branched trees

b) Uprooting the whole tree manually: Initially, selecting a severely infested patch of medium-sized and multi-branched trees is imperative. Observations of the present study indicate that *S. spectabilis* trees are typically multi-branched, which makes controlling them through debarking and manual cutting a tedious task. Once the selection process is complete, the trees should be cut using a chain saw, hand saw, loppers, or a cutting knife, and the plants should be lopped off at ground level (Plate 4.12).

Subsequently, the soil around the tree trunk (1.3 cm ground line radios from the tree trunk) should be dug up, and the root zone should be uprooted as much as possible, with the root fragments detached. This process is necessary to damage the underground carbohydrate storage structures, including the taproot zone of the tree. After complete removal, the cleared portion should be filled with soil. Since *S. spectabilis* possesses large roots, it is nearly impossible to deplete root energy reserves with repeated cutting, so strict monitoring is required for the uprooted patch for up to two years.

Regular evaluation is necessary during the first rainy season to prevent re-sprouting from any remaining root fragments exposed to sunlight. This option can be adopted if adequate financial support and labour are available, as it is labour-

intensive, expensive, and requires strict monitoring. If this option is not feasible, the next method should be pursued.

b) Cut from ground level and cover with soil and leaves: Select the patch of *S. spectabilis* invaded area. Machetes, axes, or chainsaws are recommended for cutting down trees. The cuts should be made as close to the ground as possible to remove most stem buds effectively. If feasible, these tools should also cut the root buttresses or crown. Afterwards, the cut end of the tree should be covered with nearby soil up to a distance of 30 cm from the above ground level (Plate 4.12.c). The soil should be firmly packed down using tools and any surrounding leaves. This layer of soil will act as a barrier, preventing sunlight exposure to the cut end of the trees. It has been observed that elephants tend to approach and disturb the treated area in forested areas, removing the soil. It is advisable to place some dried leaves on top to conceal the treated area.

Typically, prolific sprouting occurs within five days in the case of *S. spectabilis*. However, our experiment has shown that this barrier method reduces the sprouting rate and gradually kills the tree. It is important to continue cutting when re-sprouts appear, repeating this process until no further regrowth occurs. Strict monitoring is necessary for the success of this treatment. The most effective time for cutting is during the hot summer months when the root reserves are at their lowest, and the soil is dry. Mechanical treatments alone are insufficient in cases of severe and widespread invasion of S. spectabilis in a larger area. These treatments require intense labour and significant funds, and there is also a risk of re-invasion by this species. Applying biocides such as herbicides in a controlled and specific manner for this species is recommended for effective eradication. The following treatments have proven to be significant in managing this species.

c) Hatch and squirt: In this method, an initial incision is made in the lower portions of the tree trunk. For medium-sized *Senna spectabilis* trees, three or four evenly spaced cuts around the woody stem are sufficient. These cuts can be made using a hatchet, axe, machete, or cane knife. Once the cuts are made, the concentrated herbicide (Glyphosate) should be immediately applied to the exposed area of the cut end within 15 seconds (Plate.4.14).The cuts should be approximately 1-1.5 inches deep, primarily affecting the cambial layer. It is important to avoid making multiple cuts directly above or below each other, as this can hinder the movement of the

herbicide within the stem. Instead, make an incision around the stem. To apply the herbicide, use a handheld, chemical-resistant 10 ml syringe and apply 2 ml of concentrated herbicide to each incision. The amount of glyphosate used will depend on the size of the cut and how much it can hold without spilling onto the bark, but the minimum amount should be 2 ml. using a syringe helps minimize spillage of the herbicide onto the soil or other surrounding plant species. The glyphosate should remain within the incision cut. The cuts should be made on the stem up to 1 foot above ground level. It is advisable to avoid using this method if rainfall is predicted within 48 hours, as prolonged exposure to water can hinder herbicide absorption. Similarly, prolonged cold temperatures can cause the herbicide to freeze in the cut, resulting in poor absorption. Heavy spring sap flow can wash away the herbicide from the incision cuts, leading to poor control and potential transfer to non-target plants in the soil. Additionally, prolonged and severe drought conditions are also ineffective for this method. However, this method is highly selective in managing the targeted Senna tree species with minimal damage to the ecosystem. It is also a fast and cost-effective approach. After applying glyphosate to the targeted Senna trees, the herbicide's activity can be observed by disrupting typical phenology, such as yellowing foliage and gradually drying off. The timeframe for these effects can range from one to three months, depending on environmental conditions.

iv) Methods of removing large trees (> 30cm GBH)

e) Debarking

It is an effective technique for large *Senna spectabilis* trees with a single trunk. In comparison to other methods of tree removal, it causes minimal disruption to the surrounding environment and does not disturb the soil (Plate 4.15). The process involves cutting a strip of bark 90 cm wide along the entire length of the tree trunk. This strip must be cut deep enough to remove the vascular cambium, or inner bark responsible for transporting sugars and carbohydrates between different parts of the tree. The inner cambium layer is also responsible for producing new wood and bark. The main challenge in debarking is maintaining the connectivity of the bark. After one week of debarking, the tree may exhibit wilting symptoms during the summer days. However, within a month, bridge-like bark connectivity or re-sprouts can be observed from the collar region of the trees. Sapwood samples were collected from above and

between the debarked areas to confirm the presence of living tissues or a functioning conducting system in the treated tree. Anatomical studies were conducted and plate 4.17 shows that the inner layer persists in the debarked trees even when wilting symptoms are present.

In order to remove the bark from a tree, it is recommended to make parallel cuts around the tree's circumference, with a distance of approximately 3 inches or more between each cut. These cuts can be made using a knife, axe, or saw and should be slightly deeper than the cambium layer. For debarking, the trunk should be struck sharply between the cuts using the back of an axe or other blunt object. This will cause the bark to come off in large pieces, effectively preventing any further growth of the tree.

It is important to exercise caution when making these cuts, as cutting too deeply into the trunk may cause the tree to snap and fall during high winds. To determine the depth of the cambium layer, it is recommended to make two short test cuts and strike the bark between them. After several strikes, the bark should come off intact, exposing the cambium and wood (xylem) below.

When debark a tree, it is important to extend the cuts to include the collar region of the tree and the buttresses around the trunk. Failure to do so may result in the accumulation of food materials at the exposed bark ends, leading to the formation of several new sprouts.

f) Completely cut the tree at 50cm above the ground level and strip the bark, including collar region

In comparison to the process of debarking, this method is significantly faster and more effective. Typically used for tree eradication, we cut down and remove the tree. (Plate.4.13) However, in the case of *Senna spectabilis*, after being cut down, the tree regrows vigorously from the cut end within a week, forming a multi-branched tree within three months. To address this issue, we have developed a mechanical method of tree extermination by cutting the tree and stripping the bark. The bark is stripped from the tree up to the basal portion or collar region, 50 cm above ground level. Soil is then filled at the cut ends of the collar region to prevent exposure to sunlight and reduce regrowth. When stripping the bark, we follow the same procedures as in debarking, which involve removing the vascular cambium or inner bark, a thin layer

of living tissue. After completely removing the bark, we make slanting cuts around the tree trunk or stock to ensure and prevent further sprouting.

g) Drill fill

Please choose the trees to be eliminated. Utilize a drilling technique to create holes with a 5-10 mm diameter through the bark and cambium of the trunk. (Plate. 4.11.a) These holes should be spaced approximately 5-6 cm apart around the trunk, forming a 300-degree angle of slant. The drilling should be done below the lowest living branch and 1 foot above ground level. Fill these holes with undiluted glyphosate using a syringe. For large trees measuring over 30 cm, creating 10-15 holes is necessary. The drilling can be performed using a battery-operated or manual drill bit. The drilling method offers the advantage of minimal herbicide leakage into the soil.

h) Hatch and squirt for large trees

The technique of using a hatchet or axe to cut the lower sections of the tree trunk is employed in the same manner for medium-sized trees. However, there is a distinction regarding large trees, as many cut ends are required. It is necessary to disrupt the translocation of food materials and allow the active ingredient to spread throughout the tree, ultimately destroying leaves and the root system. Consequently, the tree is unable to regenerate. A concentrated herbicide, measuring 2 ml, is administered through a syringe into the incision. This application is carefully timed to coincide with the active growing season of *S. spectasbilis*. Following this treatment, a strict monitoring process is implemented for three months.

4.3.1.2. Prerequisites, impacts, and application methodology of herbicides.

When managing areas invaded by trees, several essential factors must be considered, including efficacy, cost-effectiveness, environmental disruption, and time. The most effective approach for the treatment of woody stems is to apply treatments during the late summer and autumn, specifically between late August and November. Cut the stems as close to the ground as possible and promptly apply herbicide directly to the cut surface. Delaying the application will result in reduced effectiveness of the treatment (Table 4.34 and 4.35).

Various methods can be used to apply the herbicide, such as a sponge, paintbrush, syringe, or Knapsack Sprayer/squirt bottle. In this case, we have standardized Glyphosate as the herbicide to control the tree forms of *S. spectabilis*. Glyphosate is a broad-spectrum herbicide that has been extensively utilized since its commercialization in 1974 (Franz *et al.,* 1997). It is commonly employed in the agricultural sector (Dill *et al.,* 2010). The chemical name of this herbicide is N-(Phosphonomethyl) glycine, which is derived from the amino acid glycine and phosphonic acid.

Herbicides are comprised of three primary constituents, specifically the active ingredient glyphosate, water, and a surfactant blend possessing soap-like properties. This surfactant blend serves to facilitate the adherence and penetration of the active ingredient onto the leaves or the area of application. Following the application to plant foliage and subsequent absorption by the leaves, glyphosate is transported through the phloem to actively growing tissues, such as roots and storage organs (Duke and Powles, 2008). Glyphosate is absorbed and translocated throughout the various tissues of the plant, with the surfactant augmenting its delivery. Once within the plant, glyphosate hampers the activity of EPSP synthase, an enzyme that hinders the plant's ability to produce specific essential amino acids necessary for growth and survival (Franz *et al.,* 1997). Notably, humans and animals do not utilize EPSP synthase, which mitigates any unreasonable toxicity concerns associated with glyphosate use when applied according to label directions.

In order to ensure the safety of an herbicide, the United States Environmental Protection Agency conducts a comprehensive risk assessment. Additionally, various regulatory bodies and scientific organizations, such as the European Commission, Health Canada, and the World Health Organization, have thoroughly examined data on glyphosate.

Over the course of the past four decades, glyphosate has undergone extensive toxicology testing. Multiple registering companies from around the world have generated at least six distinct sets of toxicology data, resulting in over 300 individual toxicology studies. These data consistently indicate no cause for concern regarding developmental and reproductive toxicity, carcinogenicity, genotoxicity, endocrine disruption potential, neurotoxicity, and immunotoxicity. Based on these findings, the

European Commission's Health and Consumer Protection Directorate-General recently re-registered glyphosate in Europe in 2014.

Glyphosate has been demonstrated to lack the ability to induce acute or chronic toxicity in higher organisms, encompassing wild mammals, birds, fish, aquatic invertebrates, as well as terrestrial invertebrates like earthworms and honey bees, even when exposed to environmentally plausible levels. Correspondingly, when utilized appropriately and within realistic thresholds, glyphosate-based formulations do not engender undesirable detrimental consequences on wildlife. Nevertheless, notwithstanding these established conclusions, certain accusations persist, necessitating a cautious approach towards the application of glyphosate in delicate ecosystems, such as wildlife sanctuaries.

Glyphosate only affects non-target plants when there is direct contact during application. Target-specific methods such as hatch, squirt, and drill fill are employed to mitigate this risk. For applicator safety, proper hand gloves and respiratory masks must be worn during these treatments. The dosage of glyphosate typically depends on various factors. During hatch and squirt treatment, we used different dilutions of the herbicide, such as 1:1, 3:1, and concentrated solutions.

Concentrated glyphosate followed by a 1:1 dilution proved to be more effective. Some studies suggest that applying a concentrated dose of herbicide to frilled regions at the beginning of the growing season can kill developing cambium cells, hindering the translocation of the poison and reducing its effectiveness (Geldenhuys, 1982). However, the concentrated solution demonstrated significant effects in our treatments for *S. spectabilis* trees and led to mortality.

4.3.1.3. The potential for re-invasion.

The management of tree invasion is an arduous task that requires strict monitoring. In the case of *S. spectabilis*, the situation is particularly critical as eradicating and managing its invasion and regrowth poses significant challenges. In the Wayanad wildlife sanctuary, where the Senna invasion is rampant, we have attempted over 50 eradication and management treatments in collaboration with forest department officials over the past decade.

However, despite the treatment, the *S. spectabilis* continues to re-sprout and regrow. While some treatments have shown initial success and sudden mortality rates,

new shoots and sprouts emerge from the remnants of cut-down or treated trees (Plate.4.18.f & g). For instance, uprooting the entire tree using an excavator has proven to be a more effective and efficient eradication method. However, in our study area, where selected patches of Senna invasion were uprooted in this manner without scientific study or background knowledge, reinvasion was detected after the first summer shower. New sprouts emerged from the exposed areas of the root portion or collar region, proliferating and spreading more aggressively than before.

Additionally, uprooting by excavators severely disrupts the soil condition and the ecosystem, making it an impractical method in a fragile ecosystem. Similarly, traditional methods such as debarking and ring barking, which typically result in drying ordinary trees, do not have the same effect on *S. spectabilis*. In this case, a connecting bridge forms from the remaining cambial or vascular layer between the ring-barked portions, allowing the tree to regenerate and become multi-branched.

The successful debarking of trees relies on properly removing bark, including the cambial layer (as depicted in Plate 4.15.b). In the case of multi-branched trees, when using glyphosate for cut sump treatment, it is observed that the tree appears dried within the first three months. However, new sprouts emerge abnormally from the collar region after being showered (Plate.4.11.f &g). Additionally, cut stump treatment has a higher risk of glyphosate spillage into the surrounding area. While cutting the tree at ground level and applying glyphosate proves to be a successful method, the spillage issue remains a concern, making it impractical for field application. Complete cutting of the tree is not feasible due to the rapid growth of prolific sprouts within a week, forming a sizeable multi-branched tree with a broad canopy within six months. Traditional methods, such as salt application to the cut end of the tree, have been attempted for complete tree eradication (Plate.4.11.b). However, these efforts have yielded negative results in the study area, as it is part of a wildlife foraging area. Elephants, in particular, have wiped out the salt and even dug up portions of the tree at its collar region. Numerous challenges have been encountered while eradicating the *S. spectabilis* tree invasions.

4.3.1.4. Monitoring and evaluation of control treatments in *Senna spectabilis* invaded area

In the case of the *S. spectabilis* invasion eradication protocol, potential methods for eradication and control were only successful after a strict three-year monitoring period. Following the application of each treatment, regular monitoring is required for three weeks, followed by weekly monitoring for one year. Subsequently, monthly observations are necessary for three years. It is crucial to carefully observe the treated area that has been invaded by the tree, particularly for any re-sprouting from the treated trees. If any signs of re-sprouting are observed, immediate removal is necessary. Additionally, any new seedlings in the invaded area should be uprooted immediately.

To ensure the eradication programme's success, it involves forest watchers, social groups such as natural clubs, tourists, trekking groups, school student groups like NCC and NSS volunteers, participants in natural camps, social forestry groups, and tribal participatory programs. These groups should be incorporated into the campaign through skill training, as their support is crucial. However, before their involvement, they must clearly understand the tree's nature, its invasive behaviour, and the possibility of re-invasion. Implementing the treatments outlined in the management protocol, conducting regular monitoring, and executing a comprehensive eradication campaign can effectively manage the invasion of *Senna spectabilis* within a specified timeframe.

4.3.2. Restoration techniques for areas affected by the invasion of *Senna spectabilis*.

Thirteen indigenous species were carefully chosen for their suitability in the respective regions and subsequently planted for restoration experiments at Muthanga within the WWL sanctuary, where the *S. spectabilis* had been eradicated. Different models of planting methods were tested (Table 4.37. Plate.4.19). Nine months after planting, the survival percentage of thirteen indigenous species at the restoration area is being assessed.

Table 4.38 List of treatments and methods of planting in the restoration area

Restoration Treatments	Method of planting	Planting densities plants or seeds /ha
RT-1	Seedlings of fast-growing species	2000/ha
RT-2	Seedlings of fast-growing species and slow-growing species	3000/ha
RT-3	Seedlings of fast-growing species and slow-growing species	4000/ha
RT-4	Seedlings of fast-growing species, slow-growing species, Seeds	3000/ha
RT-5	Seedlings of fast-growing species and slow-growing species, Seeds	4000/ha

Nine months after planting and seed broadcasting, the overall survival rate was recorded at 6.2% (Table 4.38, Plate.4.19). The treatments with the highest survival values involved a mixed planting model, incorporating seedlings of fast-growing and slow-growing species with seeds at higher densities. Notably, *Bambusa bamboo* and *Mangifera indica* demonstrated comparatively higher survival rates.

According to Knowles and Parrota (1995), survival rates exceeding 75% are considered high in plantings of seedlings in degraded areas. Durigan and Silveira (1999) consider survival rates above 60% satisfactory. Based on these standards, the restoration activities in the eradicated area of *S. spectabilis* did not yield positive results. Prajitha and Sudhabai (2022) studied the invaded area of *S. spectabilis* in Muthanga, analyzing the soil composition. Their findings revealed high acidity and water-holding capacity, which could contribute to the vigorous sprouting of seedlings.

Table 4.39 Survival % of 13 indigenous species at restoration area, nine months after planting

Name of the species	**RT-1**	**RT-2**	**RT-3**	**RT-4**	**RT-5**	**Mean**
Terminalia crenulata	----	----	----	----	---	0
Garcinia gummi-gutta	---	----	----	2.34	1.2	1.77
Terminalia arjuna	---	----	3.5	--	1.71	4.35
Aegle marmelos	---	---	2	--	1.3	2.65
Mangifera indica	8	11	12	21	20	14.4

Shorea roxburgii	---	---	---	5.7	4.73	5.21
Bambusa bamboo	9	10.4	15.5	12.4	24.4	14.34
Ficus racemosa	---	2.6	5.33	7.33	10.6	6.46
Syzygium cuminii	1.3	2.5	4.4	6.4	7.2	4.36
Artocarpus heterophyllus	----	---	5.5	9.2	12.4	9.03
Ficus relegiosa	5.33	5.5	8.4	---	8.8	7
Phyllanthus emblica	---	---	6	6.4	4.4	5.6
Bohinia verigata	4	4	10.8	8.4	8	7.04
Limonia acidissima	--	--	6	6.4	7.2	6.53
Mean	5.52	6	7.22	8.55	8.6	**6.82**

To attain a comprehensive understanding of the complexities involved in restoring the eradicated area of *S. spectabilis*, further research must be conducted to investigate the microbiological activities in the soil that may facilitate the rapid germination and growth of *S. spectabilis* seedlings in the sanctuary. Studies have shown that this species possesses allelopathic effects, and it is crucial to comprehend the impact of these effects on other native species, as they may cause the low survival percentage of restored species in the *S. spectabilis* eradicated area. Additionally, a detailed setup is required to prevent herbivores from foraging on the planted seedlings, which poses a significant challenge due to the large terrestrial mammals in the studied forest areas. The *S. spectabilis* eradicated area was covered with a thick layer of decaying leaves and bark peelings from *S. spectabilis*, which likely contributed to allelopathic activity in the soil. Further research is necessary to test this hypothesis.

Plate.4.11.a-g Different control treatments for eradicating *S.specatabilis* a)Drill fill,b)Rock salt application,c)Root feeding d) $CuSo_4$ Application,e)Uprooting using excavator f)Cut-stump treatments by glyphosate,g)Sprouting on the glyphosate treated stem

Plate 4.12.a. Uprooting manually and soil filling

Plate 4.12.b-c. Cut the tree at ground level and fill with soil and leaves

Plate.4.13. Completely cut the tree at 50cm above the ground level and strip the bark, including collar region

Plate.4.14. Methods of Hatch and squirt -1 ft wide above the ground and apply glyphosate. Dried trees after treatments.

Palte.4.15. a) Deabrking method, b) Inner layer of the bark, c)Dried trees after debarking treatment.

Plate.4.16. a) Uprooting the seedlings of *S.spectabilis*, b)Disposal method of *S.spectabilis* saplings

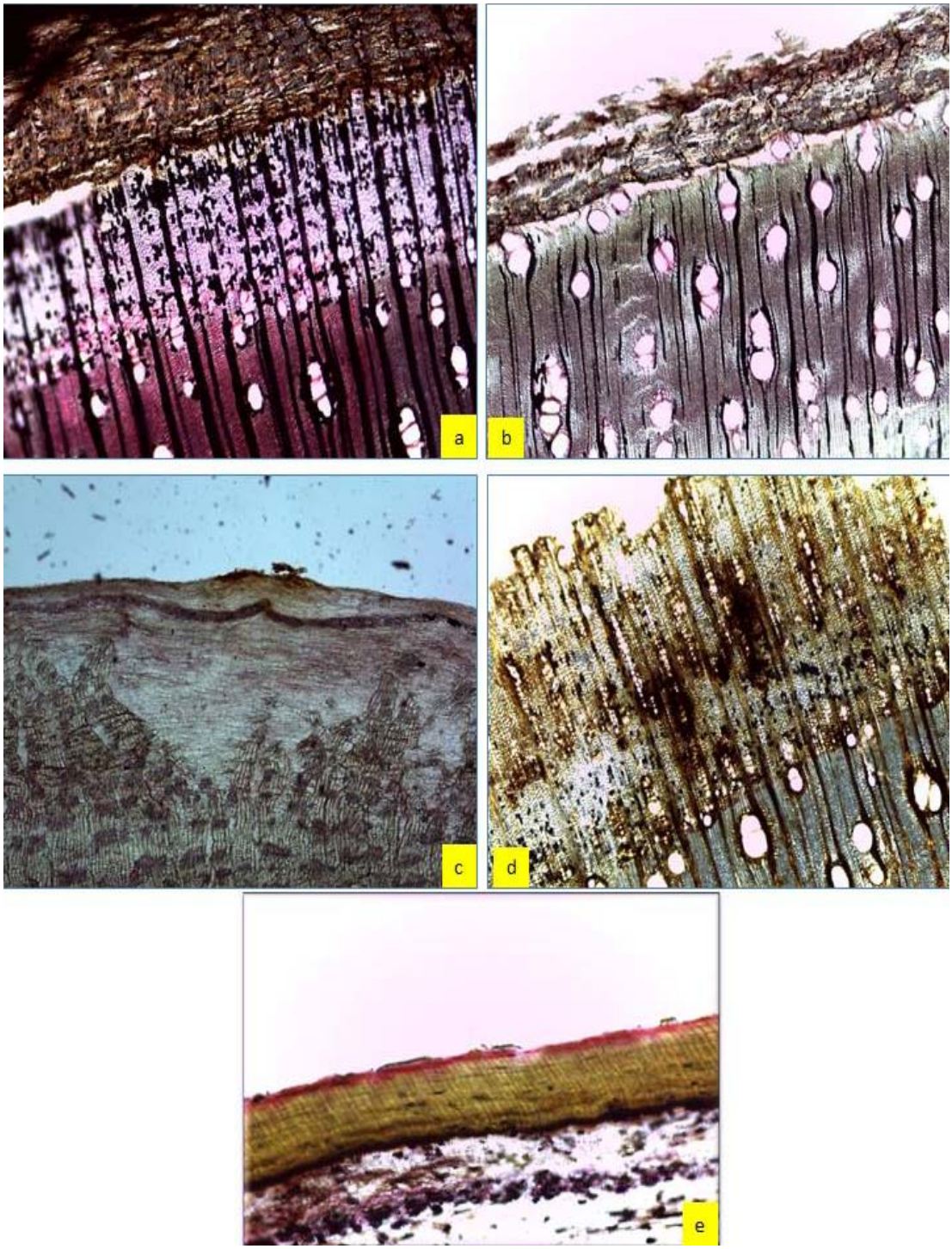


Plate.4.17.a) *S.spectabilis* debarking stem region c.s; b) Connecting bark Live C.S; c) Sapwood region including bark with inner cambial layer C.S; d) live sapwood TLS,e) Dry sapwood with bark C.S

Plate.4.18. Possibilities of re-establishment of *S.specatbilis* after control treatments. a)Ring barking method,Bark connectivity after ring barking, c)Debarked tree,d) coppicing from collar regions, e)Bridge like bark connectivity after debarking,f) Cut the tree g) re-sprouting.

Plate.4.19. Restoration activities at *S.spectablis* eradicated area, a)Restoration area at Muthanga, b) and c) Seeds and seedlings, d)-f) Planting of native seedlings at restoration area

Chapter-5

Summary and Conclusion

Chapter-5

Summary and Conclusion

The tree invasive species *Senna spectabilis*, previously classified as a 'moderate risk' invasive species, is currently exhibiting a rapid spread in some areas of the Western Ghats. The invasive presence of *S. spectabilis* is predominantly confined to Wayanad, Attappady, and Periyar in Kerala. This tree invader's density, abundance, and distribution was surveyed throughout Kerala. This invasive tree species has emerged as a matter of great concern among ecologists, professionals in biological conservation, forest authorities, and managers of natural resources.

The findings of this study revealed that the populations of *S. spectabilis* are dispersed across the districts of Wayanad, Idukki, and Palakkad, with additional isolated populations observed in Kozhikode, Thiruvananthapuram, and Kannur districts. Notably, Wayanad district exhibited the highest prevalence of infestation, with the potential for distribution in both natural forests of tropical moist deciduous and tropical dry deciduous, as well as road avenues. This trend was followed by the Periyar and Anaikkaty, with dense populations and significant regenerative abilities.

The vegetation analysis indicates that invasive alien plants, such as *S. spectabilis*, have had a significant detrimental impact on the ecosystem and the distribution patterns of plants within the Wayanad Wildlife Sanctuary. The vegetation characteristics and distribution patterns of the Wayanad Wildlife Sanctuary, Kerala, show that the Southern and Northwest ranges of the study area support many plant species, including deciduous trees. Phytosociological characteristics reveal that *S. spectabilis* is a dominant species in the studied sites, with the highest IVI value in the tree layer of the southern ranges and one of the dominant species in the tree layer of the Northwest ranges. In the two study sites, most of the plant population shows an aggregated or clumped pattern of species distribution.

This study unequivocally demonstrates that most of the dominant species observed in the shrub and herb layers were Invasive Alien Plants. *S. spectabilis* forms a part of the tree layer, as it has a higher frequency in this layer. While in the tree layer *S. spectabilis* dominates, the dominance of Lantana camara and Chromolaena odorata was observed in the shrub layer.

In both study locations, this prevailing dominance has led to the displacement of indigenous species within the region. Consequently, native species are currently being regarded as secondary species. *S. spectabilis* exhibits a substantial density and a higher Importance Value Index (IVI) value. Specifically, this species contributes to the decline of native vegetation within the examined area. This floristic investigation has identified 79 invasive plant species in the WWL Sanctuary.

The invasion of *S. spectabilis* in the Wayanad Wildlife Sanctuary has been identified as a significant predicament. By examining the phytosociological characteristics within the sanctuary, the populations of *S. spectabilis* have been categorised into areas of high density, medium density, and low density. The Muthanga region has exhibited the highest recorded density and abundance, followed by Tholpetty. The areas of high density for this invasive species have been found in the Vayal systems of the WWL Sanctuary. These Vayals, which are swamps or low-lying grasslands, represent the edaphic climax within the sanctuary. Unfortunately, the highly dense *S. spectabilis* is encroaching upon the Vayals, particularly in areas along the banks, such as Kakkapadam near Muthanga. This invasion poses a significant threat to preserving these pristine ecosystems and could lead to their complete eradication.

Based on these observations, the impact of *S. spectabilis* in the forests of the Wayanad Wildlife Sanctuary has been deemed highly significant, resulting in the rapid spread and distribution of the invasive species, followed by the decline of native species. The occurrence of these alien invasive species poses a significant threat and has a detrimental impact on the surveyed area.

The distribution of the invasive tree species *S. spectabilis* was studied to the changing climate. The widely used species distribution model Maximum entropy-based model (MaxEnt) was used. It gave robust output and high performance for a small set of presence data. Using these SDMs it was possible to delineate the invasion hotspot areas and thus devise better management plans.

The future projection of the distribution of the invasive species was determined by applying the maximum entropy probability distribution based on the current distribution analysis. The modelling process for predicting the future distribution of *S. spectabilis* utilised the same current environmental layers and future predictor layers

for different RCPs (RCP 4.5, RCP 6.0, and RCP 8.5) and the HadGEM2-ES model. The relationship between the environmental variables and the selected invasive species was analysed based on the model output.

Among the environmental variables, Isothermality (BIO3) had the highest percentage contribution in constructing the model for the distribution of *S. spectabilis*, followed by elevation and annual mean temperature (BIO1). Additionally, precipitation variables such as precipitation of the warmest quarter (BIO18), precipitation seasonality (BIO15), and precipitation of the driest quarter (BIO17) also made significant contributions. The influence of the isothermality variable varied, with an increased contribution in the RCP 2.6 scenario in the 2050s and the RCP 6 scenario in the 2070s. Non-climatic variables had less significance in the distribution model of *S. spectabilis.* However, the permutation importance analysis revealed that precipitation of the warmest quarter (BIO18) had the highest importance, while isothermality (BIO3) had the least.

The variables that positively influenced the distribution of *S. spectabilis* were temperature seasonality (BIO4) and precipitation seasonality (BIO15). A higher presence probability of the species was observed when the annual mean temperature was at 22.5^0C. The study results indicated that the protected areas in Wayanad are at high risk in every RCP scenario. The 2050s and 2070s showed an increase in range contraction, particularly in the RCP 8.5 scenario in the 2050s and the RCP 4.5 scenario in the 2070s.

The analysis of the distribution change of *S. Spectabilis* indicates that, compared to the current scenario, most areas in the Wayanad district, particularly the Wayanad Wildlife Sanctuary, experienced no change in distribution across all greenhouse gas pathways. Presently, it has been observed that 86% of the potential area within the wildlife sanctuary falls under the category of very high habitat suitability despite the absence of significant range expansion. In the future scenario, it is projected that over 90% of the potential area within the wildlife sanctuary will maintain its current level of habitat suitability. Conversely, the Wayanad district has experienced a notable contraction in range, particularly in the western region, where areas have no occupancy or low suitability for *S. Spectabilis*. However, it is important to note that

the distribution of *S. Spectabilis* could pose a risk of extinction for numerous flora and fauna species if appropriate measures are not taken promptly.

The no occupancy region and no change distribution areas remained unchanged, which will hamper the native flora and fauna community of the area. The high and very high suitability area for *S. spectabilis* showed a decreasing trend with the RCPs in both periods whereas good habitat suitability showed an increasing trend. Therefore, the study's observed habitat suitability of *S. spectabilis* calls for urgent action in managing areas where biodiversity is at a higher risk of danger.

The phenological patterns of the *S. spectabilis* were observed in Wayanad Wildlife Sanctuary and Anikkaty areas of Southern Western Ghats. The evaluated phenological events were vegetative events, such as leaf flushing, mature leaf and leaf abscission, and reproductive phenophases classified as flower bud initiation and anthesis or open flower for flowering and immature fruit and mature fruits for fruiting phenophases. The phenophases were correlated with climatic variables of maximum and minimum temperature and rainfall using Spearman's rank correlation. The vegetative phenophases were observed throughout the study; flowering predominated from September to January, and fruit maturation from January to May. The Fournier intensity index was used to analyse the phenophases. The significant correlation between phenophases and climatic variables was represented by the relation between rainfall and leaf flushing, leaf abscission, and mature fruit period with maximum temperatures. Linking phenological patterns to plant functional traits may help better understand plant community function and assembly during the invasion. The phenological events of the *S. spectabilis* were distinctly seasonal. To this day, no definitive documentation or research is available regarding the phenological occurrences of *S. spectabilis*. Satyanarayana *et al.,* (2013) noted that *S. spectabilis* flowers between August and March; http://www.flowersofindia.net mentions October to December. In Zambia, flowering is observed in January and February, with fruit ripening in September or October according to the Agroforestry Database 4.0 (Orwa *et al.,* 2009). In contrast, the flowering process is consistently observed throughout the year in the United States. However, comprehensive phenological records have not yet been reported from its native range and other regions. This study presents the phenological pattern of *S. spectabilis* within its invasive range.

Reproductive studies of *S. spectabilis* and pollen ovule ratio indicate this species is cross-pollinating. The species is self-compatible, owing to the simultaneous occurrence of xenogamy, geitonogamy and autogamy. This reproductive strategy helps the taxon to colonise degraded areas and invade the forest ecosystem. The anthesis process is diurnal and sometimes asynchronous. The peak insect visitors were observed from 9.00 to 12.30 hrs, with the major visitor being *Tetragonula iridipennis. Xylocopa violaceae* is also a regular visitor along with resident formicidae members, like *Oecophylla smaragdina* and *Myrmicaria brunnae*. They feed on the floral parts, like tender petals and sepals. The study revealed that the high rate of seed production in *S. Spectabilis* can be attributed to various factors, including the pollen viability and vigour of the pollen tube, the timing of anther dehiscence and stigma receptivity, the presence of multiple pollinators, and adequate pollen rewards. The pods of *S. Spectabilis* were observed to contain an average of 108.91 ± 9.69 seeds. Notably, the plant exhibited no sexual incompatibility or pollination difficulties. The reproductive syndrome of *S. Spectabilis* is conducive to achieving maximum fertilisation.

S. spectabilis is renowned for producing substantial quantities of seeds that possess an impressive longevity of up to three years and exhibit a notable resistance to mortality. The viability of these seeds was determined through rapid cutting tests, with results ranging between 79% and 87.2% and an average of 84.45% viable seeds recorded. In a controlled laboratory environment, it was observed that *S. spectabilis* exhibited a dormancy enforced by the seed coat, resulting in delayed germination. Consequently, the effects of various pre-treatment methods on the germination of *S. spectabilis* seeds were assessed. The pre-sowing treatments yielded a maximum germination rate of 96% when the seeds were immersed in 2M H_2SO_4.

Regarding the mechanisms of seed predation and seed dispersal, it has been observed that herbivores are responsible for the dispersal of seeds. The study results indicate that the Sambar deer is the most effective seed disperser of *S. spectabilis*, as it can move the seeds away from the tree crown. The successful germination of seedlings was found to occur in the distinct faecal pellets of Sambar deer and elephants, mainly when they were located away from the *S. spectabilis* tree crown. This phenomenon may have contributed to the widespread proliferation of *S. spectabilis* within the forest ecosystem. The observations also revealed that both barochory and terrestrial animals played a role in the dispersal mechanisms of *S.*

spectabilis in Wayanad Wildlife Sanctuary. Specifically, terrestrial mammals played a key role in the dispersal mechanisms of *S. spectabilis* in Wayanad Wildlife Sanctuary, and regeneration by barochory was observed.

An analysis was conducted to examine any adaptive variations in the invasive populations of *S. spectabilis* in Kerala. This analysis focused on population variations associated with morphological traits. The results revealed a significant variation in the seedling growth characteristics of the offspring from seeds collected from different populations. Additionally, correlation analysis of the characters of six-month and twelve-month-old *S. spectabilis* seedlings showed that few characters exhibited significant correlation.

These observed variations may be attributed to genetic contributions, the genotypes of the various populations, and the seedling growth traits among progenies of trees growing in different locations with varying environmental conditions.

Furthermore, the morphological variations among the different tree populations of *S. spectabilis* in Kerala were examined. The analysis revealed significant variation ($p<0.05$) between the populations in all 16 characters used for morphological variation analysis of 10 trees from 11 populations. The ANOVA results indicated a range of p-values for all the characters except for the seed viability percentage ($p=0.414$). The p-values for the remaining characters ranged between 0.001 and 0.004. Therefore, the ANOVA confirms that significant variation was observed between the populations for the fifteen characters subjected to variation analysis.

These observed variations highlight the influence of adaptive environmental and genetic factors on phenotypic characteristics. This emphasises the need for a comprehensive population genetic study to further understand these factors in this context.

The results of the karyotype analysis conducted on six populations, namely Anakkatty, Muthanga, Vythiri, Azhinjilam, Munnar, and Thiruvananthapuram, revealed a consistent chromosome count of $2n=28$, $x=14$ in mitotic metaphase cells of *S. spectabilis*. This finding agrees with the karyotype analysis of the native *S. spectabilis* specimen collected from the Royal Botanic Gardens, Kew by Mohanty and Das (2006), which also reported a chromosome count of $2n=28$.

However, statistical analysis of the other characters studied, including chromosome length, total genomic chromosome length, total chromosome volume, and total form percentage, showed significant variations among the populations. These differences suggest this species is highly unstable and may require a more comprehensive study to fully understand the underlying factors contributing to these variations.

The genetic variability of ten accessions of *S. spectabilis*, collected from various locations in Kerala, was examined using Inter Simple Sequence Repeat (ISSR) molecular markers. The findings of this study indicate that the populations of *S. spectabilis* under investigation exhibit low levels of intra-population genetic diversity and significant divergence among each other. Factors such as self-fertilisation, drift events, colonisation by a limited number of individuals, varying selection pressures even within small geographic areas, and multiple introductions have influenced the genetic diversity and distribution of *S. spectabilis* populations. It is important to note that this study represents a preliminary investigation into this invasive tree species, and it is strongly recommended that further research be conducted, encompassing a broader sample from different locations across the invaded populations using ISSR markers. Even though the results were obtained from a single primer, the genetic grouping of the studied populations in Kerala was determined based on the binary matrix data. Further clarification and a detailed genetic study are necessary to confirm this.

There is an urgent requirement for a practical containment protocol to manage and control *S. spectabilis* in forested regions. However, a comprehensive and feasible strategy has yet to be developed. Considering the management of *S. spectabilis* invasion area, efficacy, cost effective methods, environmental disruption, and time are all important factors.

This study has resulted in the development of a three-tier protocol that includes the eradication of *S. spectabilis*. According to the treatments that indicated the highest mortality rate of *S. spectabilis*, more effective strategies were adopted and compiled for controlling this species in an invaded ecosystem. The management protocol has been customised to the age and growth forms of *S. spectabilis*, as outlined in the protocols. For protected areas, such as ecologically fragile or forest lands, protocol

one recommends using mechanical methods. Protocol two suggests a combination of mechanical and chemical methods in non-protected or recommended areas.

The methods employed in this protocol include (i) the implementation of appropriate pre-requisite strategies to contain the species, (ii) the adoption of effective strategies for containment and eradication, including the mechanical removal of *S. spectabilis* from forest areas and the reduction of their population size using herbicides, and (iii) a post-containment protocol involving habitat management through ongoing monitoring and evaluation.

Successful restoration activities in *S. spectabilis* eradicated areas need further studies to discover the complexities of restoration in S. spectabilis invaded areas and understand the allelopathic effect of the species on other native species.

One of the most important features of an invasive species is its broad adaptation to a range of biotic and abiotic factors. This adaptation could be through genetic or plastic changes. Considerable work needs to be done on this invasive population in South India to arrive at conclusive evidence on taxonomic relationships and evolutionary changes in detail.

The profuse growth of the Invasive Alien Species in the Wayanad district, especially in the Wayanad wildlife sanctuary, was due to the lack of quarantine of the ornamentals as the species was accidentally introduced as ornamentals, which caused great havoc. The species' invasiveness is found in the wildlife sanctuary rather than the human-inhabited areas. Besides the distribution modelling, species traits, dispersal pathways and the mechanism of the natural filters should be better understood to prevent the profuse growth and colonisation of the invasive species. It was observed that the phenology of *S. spectabilis* exhibited a consistent pattern, thereby carrying significant implications for their management and regulation.

The results of this study can act as a precautionary note in a situation where there is a lack of information based on invasive species distribution and ecology. To tackle the aggressive growth of *S. spectabilis,* a short-term and long-term management action plan should be implemented. The legislative, scientists, and laymen should be involved in eradicating the species since it is an invasive species with high potential and competitiveness, which could survive in any conditions. Providing Awareness to the public is also a crucial step.

Over the past decade, there have been concerted efforts to compile comprehensive lists of invasive plant species in India and conduct studies on the impacts of these invasive species in various regions of the country. Managing this invasive tree species in natural habitats is a demanding endeavour that requires a significant investment of time and resources. However, this valuable knowledge can be utilised to effectively guide conservation initiatives through accurate prediction and efficient control of biological invasions.

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