Ecology of Major Invasive Alien Plant Species (IAPs) Across Various Habitats in Pakke Tiger Reserve, Arunachal Pradesh

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DEEMED UNIVERSITY, UTTARAKHAND

For

THE AWARD OF THE DEGREE OF
DOCTOR OF PHILOSOPHY IN FORESTRY
(WILDLIFE SCIENCE)

By

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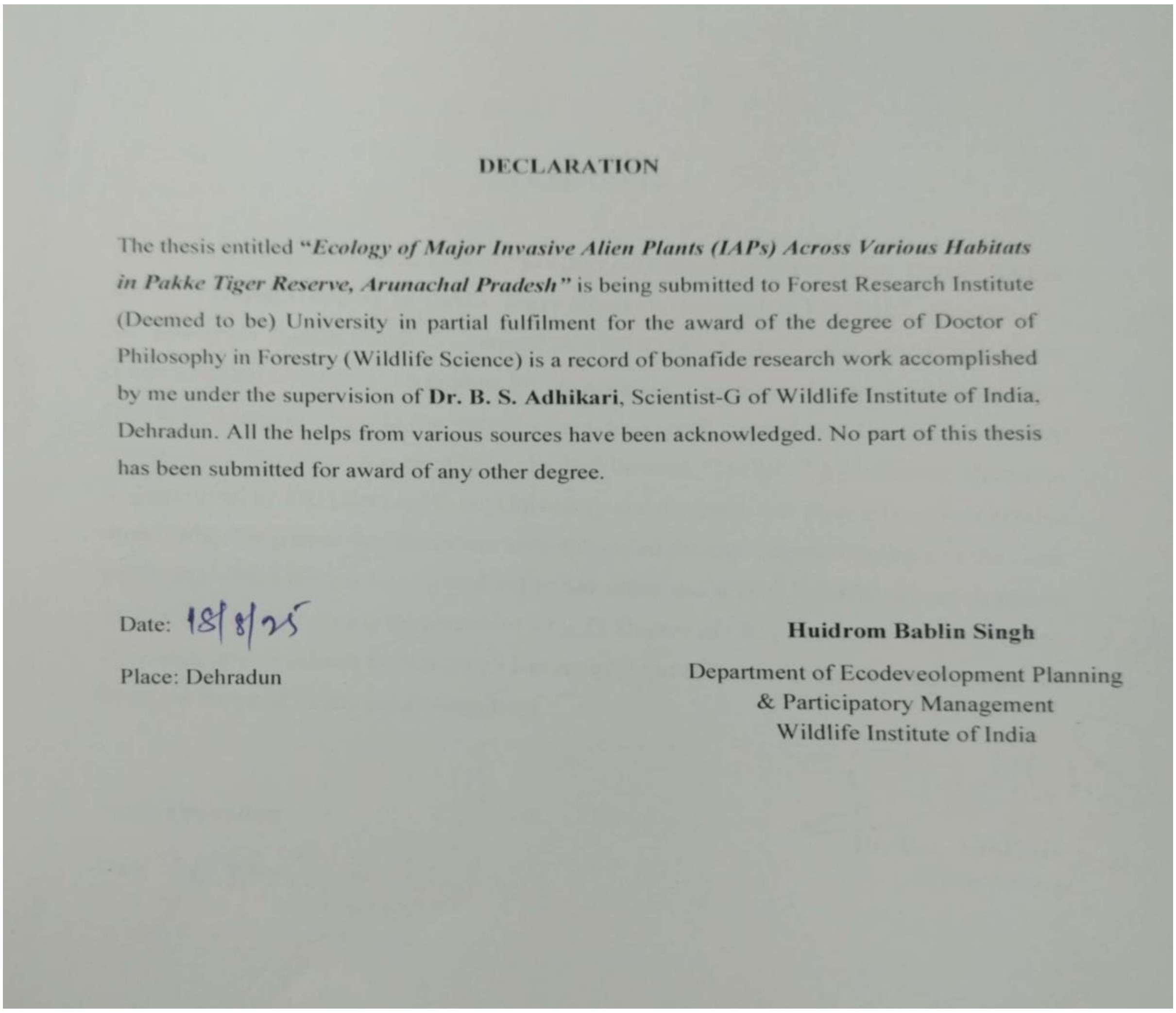
Abstract

Proliferation of invasive alien plant species poses a significant threat to the structure and function of ecosystems, making this a critical area of focus within the scientific communities. These species cause extensive damage to the biodiversity, as they are capable of altering the ecosystems and outcompete the native biodiversity. Globally, their abundance has increased dramatically in forested and other natural areas due to rapid economic growths and anthropogenic activities. To counter this alarming situation, understanding the distribution pattern, climatic influences on phenophases and ecological impact on soil by invasive alien plant species is a priority. In northeast India, studies on invasive alien plant species diversity and their impacts are very limited, especially in Arunachal Pradesh. Therefore, the present study aims to determine the patterns of distribution, phenophases and physiochemical impacts by the dominant invasive alien plant species in five forest types (Assam Alluvial Plain Semi Evergreen Forest-AAPSEF, East Himalayan Mixed Coniferous Forest-EHMCF, East Himalayan Subtropical Wet Hill Forest-EHSTWH, Non-Forest-NF and Riverine Forest-RF) of Pakke Tiger Reserve, Arunachal Pradesh. The altitudinal range of Pakke Tiger Reserve is 150- 2050 m above the sea level (490 to 6725 feet), with an average annual rainfall of 2500 mm. The current study reveals a total of 43 invasive alien plant species in the study area. However, their species richness in five forest types was in declined trend with the increase of elevation. Across the forest types, AAPSEF (30%) display the maximum level of invasion and least incursion in EHSTWHF (16%) in the current study. Among the nine species selected for the study, *Chromolaena odorata* (L.) R.M.King & H.Rob. was found dominant and distributed extensively in all forest types, with highest relative density (51%) in EHSTWHF and relative abundance (10%) in NF. The entire selected species shows contagious distribution pattern (<0.5) in all forest types except for *Urena lobata* in NF displaying random distribution (0.025 - 0.5) in study area. *Ageratum conyzoides* L., *Chromolaena odorata* (L.) R.M.King & H.Rob. and *Urena lobata* L. are the species with extensive distribution in all forest types in comparison to other selected species showing their dominance over the native vegetation. Moreover, maximum area cover percentage was exhibit by *Chromolaena odorata* (L.) R.M.King & H.Rob. followed by *Ageratum conyzoides* L., *Mikania micrantha* Kunth and *Urena lobata* L. across the forest types in Pakke Tiger Reserve. Fores types such as AAPSEF, NF and RF display high level of disturbances while the EHMCF and EHSTWHF have fewer incursions by invasive alien plant species in the study area.

Phenological pattern on the selected species shows that *Bidens pilosa* L. has the longest flowering phenophases while *Chromolaena odorata* (L.) R.M.King & H.Rob. has the maximum fruiting duration in EHSTWHF. For leaf fall and initiation, *Urena lobata* L. display the longest duration in RF. Moreover, coefficient of variation shows that most of the species display asynchronous phenophases across the forest types of Pakke Tiger Reserve. Repeated measures of ANOVA also display that *Ageratum conyzoides* L. display difference in flowering in NF and EHSTWHF while *Chromolaena odorata* (L.) R.M.King & H.Rob. leaf initiation show difference in EHMCF. Correlation and linear regression display influences of climatic parameters i.e., precipitation, relative humidity, minimum and maximum temperature to the selected species in the study area. CCA also indicates that precipitation, minimum and maximum temperatures directly influence the phenophases of the selected species. However, relative humidity influences the phenophases of the selected species when in cohesion with other climatic parameters. Ecological impact on native species by invasive alien plant species was determined by collecting the soil sample of the most dominant species i.e., *Chromolaena odorata* (L.) R.M.King & H.Rob. across the forest types in Pakke Tiger Reserve. The result shows that clay and copper contents were high in all invaded sites while soil pH, sand, boron and water holding capacity were significantly lower in invaded plots. Out of 16 soil variables, T-test reveals that AAPSEF and NF have variation in 13 variables while EHMCF, EHSTWHF and RF have difference in 12 variables. ANOVA between the invaded and uninvaded sites also indicates significant differences in Boron, iron, copper, nitrogen, water holding capacity, clay, silt and sand. Moreover, correlation shows eight of the variables display relation between them. Furthermore, PCA indicates that INNF soil properties were different from the remaining sites. The results indicate that *Chromolaena odorata* (L.) R.M.King & H.Rob. alter the soil properties of invaded areas by improving the fertility, making it ideal for the proliferation process in future scenarios.

Keywords: Arunachal Pradesh, Biological invasion, Climate changes, Ecosystem threat, Invasive alien plant species, Northeast India, Pakke Tiger Reserve



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The thesis entitled "***Ecology of Major Invasive Alien Plants (IAPs) Across Various Habitats in Pakke Tiger Reserve, Arunachal Pradesh***" is being submitted to Forest Research Institute (Deemed to be) University in partial fulfilment for the award of the degree of Doctor of Philosophy in Forestry (Wildlife Science) is a record of bonafide research work accomplished by me under the supervision of **Dr. B. S. Adhikari**, Scientist-G of Wildlife Institute of India, Dehradun. All the helps from various sources have been acknowledged. No part of this thesis has been submitted for award of any other degree.

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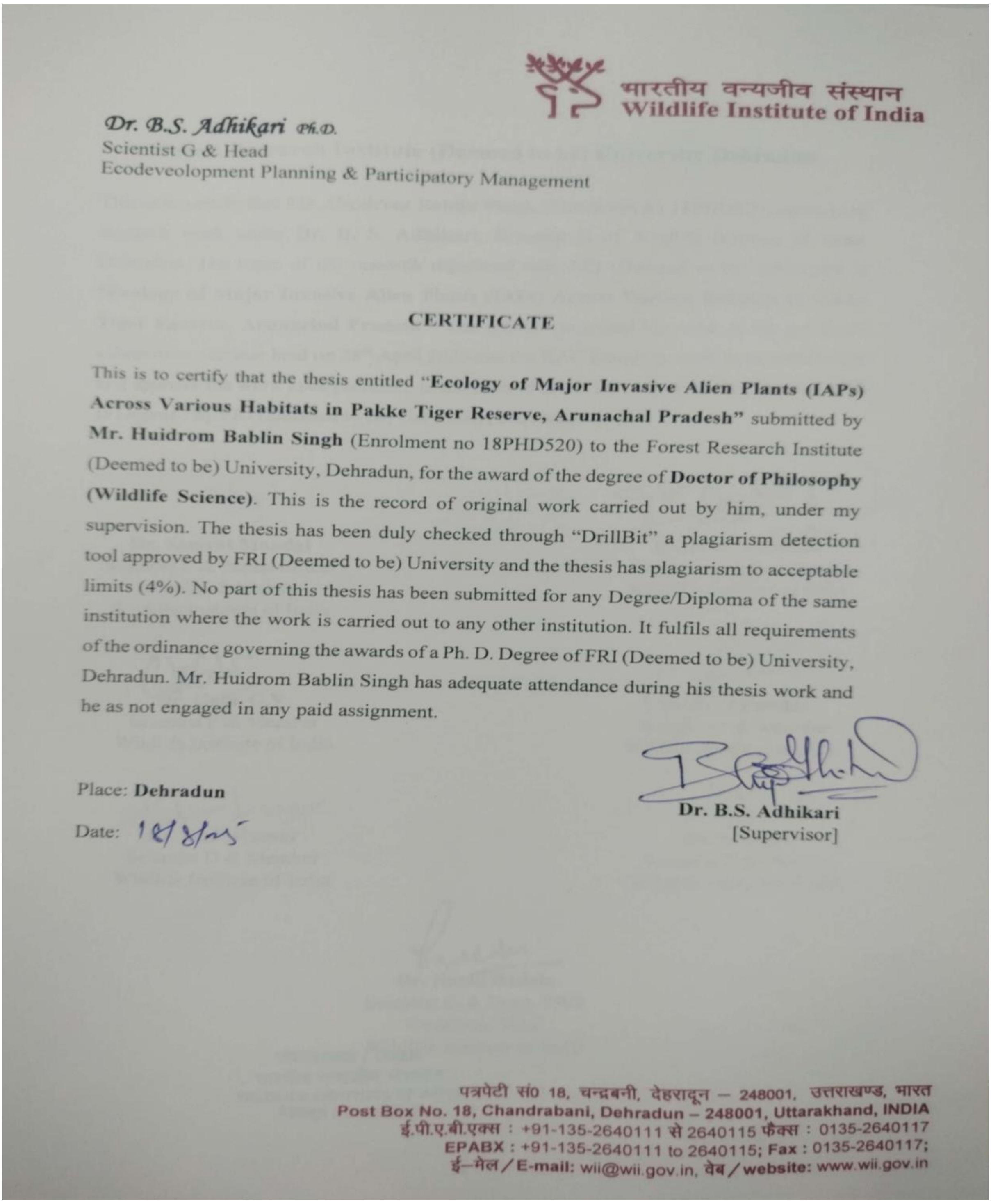

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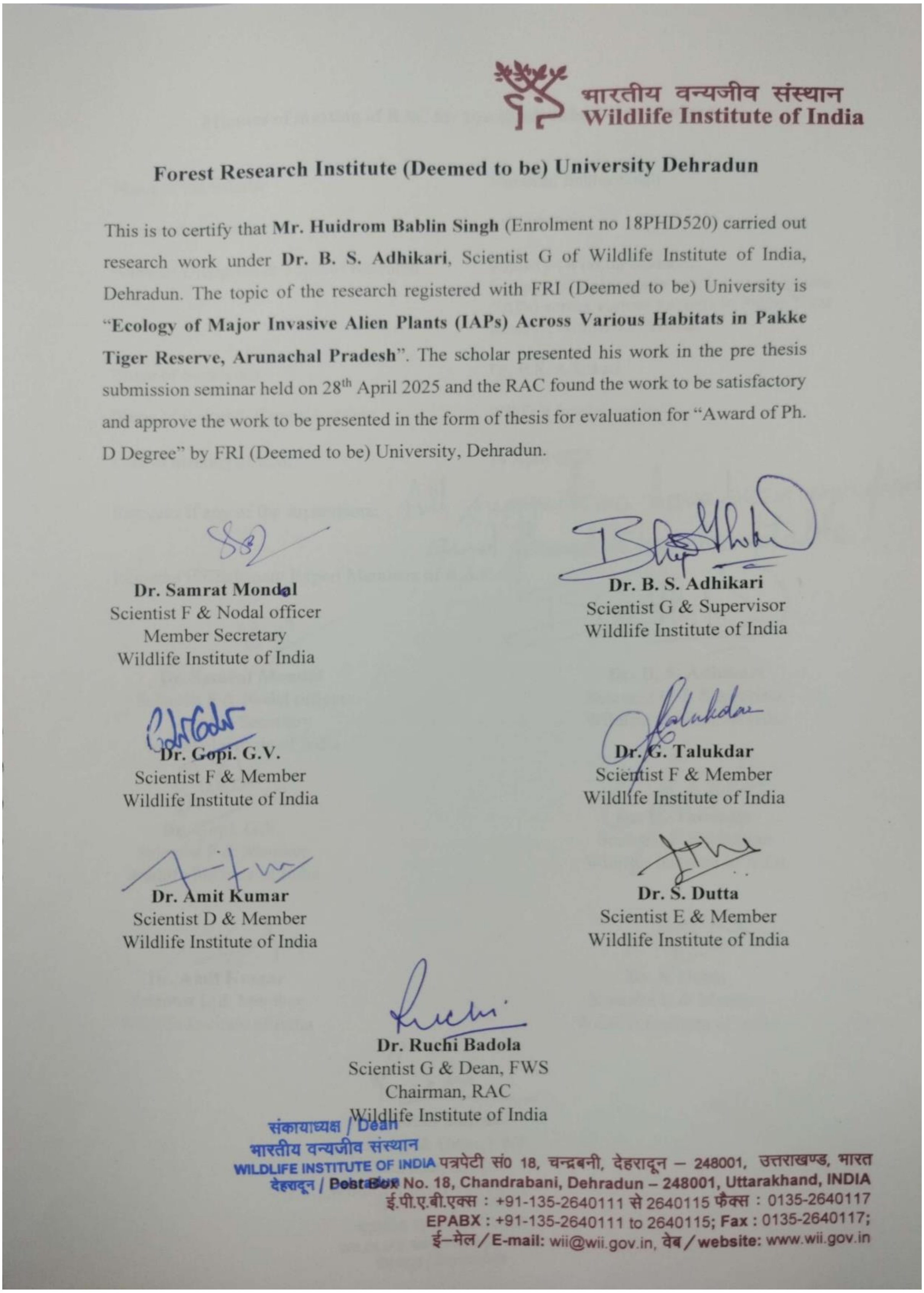

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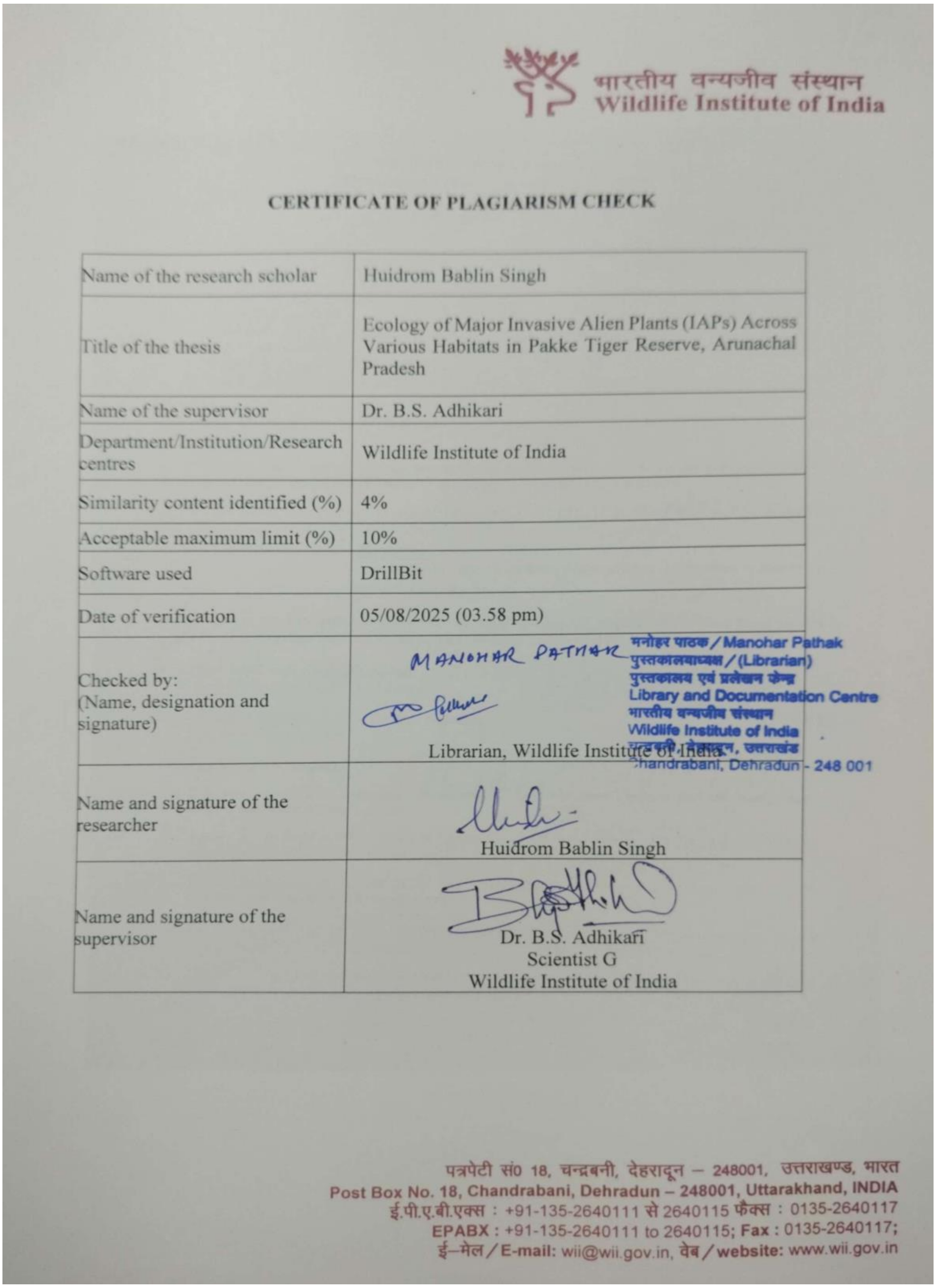

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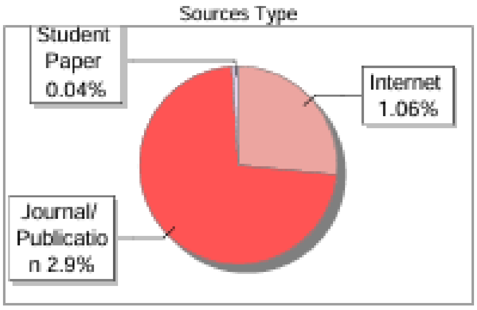

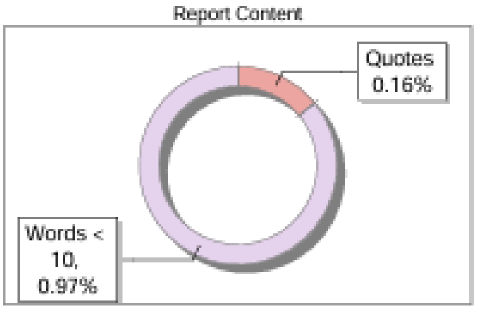

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Date: **Huidrom Bablin Singh**

Place:

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ABBREVIATIONS

IAPs	Invasive Alien Plant Species
AAPSEF	Assam Alluvial Plain Semi Evergreen Forest
EHMCF	East Himalayan Mixed Coniferous Forest
EHSTWHF	East Himalayan Subtropical Wet Hill Forest
NF	Non-Forest
RF	Riverine Forest
PTR	Pakke Tiger Reserve
ANOVA	Analysis of Variance
PCA	Principal Component Analysis
PCs	Principal Components
CCA	Canonical Correspondence Analysis
CV	Coefficient of variation
USA-NPN	USA National Phenology Network
RD	Relative Density
RA	Relative Abundance
UPGMA	Unweighted Pair Group Method with Arithmetic Mean
GPS	Global Positioning System
IHR	Indian Himalayan Region
IUCN	The International Union for Conservation of Nature
NEFA	The North East Frontier Agency
IPCC	Intergovernmental Panel on Climate Change
LI	Leaf Initiation
FL	Flowering
FR	Fruiting
LF	Leaf Fall
PR	Precipitation
MIT	Minimum Temperature
MAT	Maximum Temperature

Chapter 1

Introduction

1.0 Background

The stories of biological invasions have been known and recognized for millennia. Naturalists have been observing and contemplating biological invasions in some capacity for a long time. In fact, invasive species were mentioned in the works of Charles Lyell, Frank Egler, and Charles Darwin (Richardson and Pyšek, 2008) but the true beginning point was the book of Charles Elton, (1958) “The ecology of animal and plant invasions", which subsequently became known as invasion ecology (Rejmánek et al., 2005). The field of invasion biology is a challenging, multidisciplinary one that includes both basic and application study. A unique opportunity to study fundamental population processes is provided by biological invasions. A biological invasion, also known as a bio-invasion, is the movement of organisms into areas where they had not previously existed. This encompasses both naturally occurring slow, progressive dispersion and naturally occurring fast growth brought on by extraordinary geological or climatic occurrences (Olenin et al., 2017).

Biological invasion is a natural process, the current increased rate of invasions is unquestionably a human-instigated occurrence and one of the most significant effects that people have had on the globe and the pace of invasion had accelerated, especially over the past century, as a result of the fast alteration of natural ecosystems (Vitousek et al., 1997). Regular or sporadic disturbance is a characteristic of dynamic ecosystems by nature (Sousa, 1984; Gurevitch and Padilla, 2004), but it also makes invasions easier to carry out. But it also makes the invading process easier. According to Simberloff et al. (2013), biological invasions pose a threat to biodiversity in terrestrial, freshwater, and marine environments and one of the primary causes of biodiversity loss and is allegedly the second leading cause of species extinction after habitat loss (Jose et al., 2009). Ecologists have taken an interest in invasion research due to its major ecological effects and global economic implications (Liu et al., 2005). According to Mack and D'Antonio (1998) and Sax (2002), human activities have a significant impact on the dissemination of alien plants and invasive ecology had been linked to human-mediated activities that lead to spread outside of their potential range and biogeographic boundaries (Richardson and Pyšek, 2006). Rapid pace of globalization had made easier infiltration of invasive species from one geographic region to another. The threat of invasive plants had spread globally and is mostly due to human intervention (Vitousek et al., 1997). Biological invasion and climate change pose a threat to both the environment and the economy worldwide (Thuiller et al., 2007: Tripathi, 2009: Walther et al., 2009).

Climate change is a significant factor in invasion since it affects invasive plants flowering and creates competition with native habitat flora, where invading species typically have a modest advantage over native species (Lowe et al., 2000). The amount and intensity of invading species' effects varies. Some invaders merely affect communities and have little to no effect on ecosystems, whereas other invasions change the ecosystem's fundamental structure (Jeschke et al., 2014) and induce regime transitions (Gaertner et al., 2014). Ecosystems, communities, and native biodiversity may all be negatively impacted by rising temperatures, brisk economic development, and alien species invasion (Sanders et al., 2003; Lin et al., 2007; Thuiller et al., 2007; Kelly and Goulden, 2008; Walther et al., 2009). These changes may be brought about by invasive plant species using excessive amounts of water, light, and oxygen, adding resources, promoting or suppressing fire, retaining sand or assisting in erosion, accumulating litter, and sand accumulation or redistribution that alters nutrition flows (Richardson and van Wilgen, 2004).

The impact of invasive plants on global biodiversity is second only to habitat fragmentation and is a serious worldwide concern. Furthermore, the economic costs of invasive species are significant, albeit this had not been thoroughly explored (Pimentel, 2005). The biodiversity and environment may be seriously impacted by climate change and plant invasion (Mainka and Howard, 2010). If the climate becomes unfavorable, changes in the climate may also aid in preventing the biological invasion (Bradley et al., 2009). According to the Convention on Biological Diversity (2002), biological invasion is the second worst threat after human destruction and one of the most serious threats to Protected Areas (PAs) after habitat loss, fragmentation, and more than tourism (Pyšek et al., 2013). While the impacts of invasive alien species are widely recognized around the world (Mooney and Hobbs, 2000), basic information on naturalized plant taxa and plant invasion is still lacking in many developing countries. Many of the irreversible losses of native biodiversity that occurred in the past went unreported, but today there is a growing understanding of the ecological implications of biological invasion (Sharma and Raghubanshi, 2012).

The competitive impacts of invasive alien plants can affect native plant composition, abundance, and structure, as well as ecological processes. However, the success of introduced species outside their native range had been attributed to environmental, biotic, and dispersion factors (Theoharides and Dukes, 2007). Functional characteristic analysis, demographic analyses, spread rate models, and ecosystem invasibility methodologies have all been utilized in attempts to comprehend (Rejmánek and Richardson, 1996) or forecast the success of plant invasions to date (Burns et al., 2013). These methods have demonstrated that some settings are more vulnerable to invasion and were more likely to be colonized by a certain plant life type (Jelbert, 2018). Because of the evolutionary similarity of invading and native species, prediction of invasion success is simple, but prediction of invasion effect is difficult (Gallien and Carboni, 2017). There is a large risk of plant invasion due to the speed of globalization, however understanding risk assessment and the geographic distribution of IAPs may act as an early warning system in future research (Feng and Zhu, 2010).

As a result, an essential component that must be recognized in order to prevent invasion is to guide the effect of people's perception in future research and control measures (Shackleton et al., 2019). However, there had been a paucity of data on extinction that is based on plant invasion, despite the fact that measures must be undertaken before the real crisis of extinction (Downey and Richardson, 2016). According to Van Wilgen and Wilson, (2018), the pathway-area-species framework regulation may be extremely beneficial for effective control methods. Since invasion biology is a constantly evolving field, its terminology likewise changes quickly and is affected by a number of sometimes opposing academic, cultural, and even political viewpoints (Elliott, 2003; Carlton, 2009; Olenin et al., 2010).

1.1 Invasive Alien Plant Species

According to Richardson et al. (2000), Invasive Alien Plant Species (IAPs) are the species that were introduced into recipient ecosystem from a new area and later prohibited the growth of native species or replacing by their dominance. They were considered as the growing issue in the global scenario (Vitousek et al., 1996). IAPs were responsible for species as well as biodiversity losses and disturbances in functioning of environments (Drake et al., 1989; Mack et al., 2000; Mooney, 2005; Pejchar and Mooney, 2009). Although many authors define invasive alien plant species (IAPs) differently, the definition provided by Macdonald et al., (2003) to be the most relevant to the current discussion: “the non-native species which have the ability to harm the ecosystem including economy and human beings”. Potentially detrimental financial effects of invasive alien species might affect both human well-being and environmental services (Vilà and Hulme, 2017).

Invasive alien species can alter community composition, induce local extinctions and the loss of native genotypes, alter habitats, and have an impact on ecosystems (Vilà et al., 2010; Bellard et al., 2016). Therefore, these species' establishment and expansion put ecosystems, habitats, or species that negatively impact the environment or the economy at jeopardy (GISP, 2000). There is widespread recognition of the detrimental effects that invasive alien plant species (IAPs) have on the world's biodiversity, which have led to the loss of several plant and animal species in various the world's ecosystems (Baillie et al., 2004; Early et al., 2016; Sandilyan, 2015). Moreover, entire ecosystems come under certain risk that results in the irreversible loss of endemic species (Levine et al., 2003). Because of their capacity to change ecological processes like carbon and nitrogen cycling, hydrological cycles, the frequency and/or intensity of fire, and the alteration of the typical disturbance regimes in the native communities, invasive alien plants have drastically changed many ecosystems by outcompeting native species (Calder and Dye, 2001; D’antonio and Meyerson, 2002; Brooks et al., 2004; Liao et al., 2008; Ehrenfeld, 2010; Werner et al., 2010; Lankau, 2010) and were, thus, justifiably recognized as one of the biggest dangers to biodiversity on earth (Cronk and Fuller, 1995; Chapin et al., 2000; Koch, 2004).

It is well known that invasive species directly compete with native species for resources (Werner et al., 2010), interfere with interspecific interactions, community structure, and ecosystem processes in the native communities (Lonsdale, 1999) and disrupt co-evolved interactions among native species that have been long associated (Callaway and Aschehoug, 2000; Callaway et al., 2008; Zhang et al., 2010). According to Williams et al., (2010), managing invasive populations is estimated to cost the global economy more than £300 billion annually. Among the primary factors contributing to the loss of biodiversity and the extinction of species in the world's major plant and animal taxa were biological invasions (Duenas et al., 2021). Therefore, identifying characteristics of invasive plant species and the processes associated with invasion success had been a major goal for both invasion biologists and ecologists (Agrawal et al., 2005; Prentis et al., 2008; Castro-Dez et al., 2011; Davidson et al., 2011; Hovick et al., 2012; Burns et al., 2013).

1.2 Invasive Alien Plant Species on Biodiversity

According to Early et al. (2016), 17% of global biodiversity is believed to be very vulnerable to biotic invasions. While the "diversity resistance hypothesis" (Kennedy et al. 2002) suggests that the high natural biodiversity serves as a barrier against IAP infection, on the other hand, it is generally accepted that the landscape expansion of IAPs poses a constant danger to biodiversity (El-Barougy et al., 2021). Nonetheless, indigenous biodiversity and carbon sequestration may be adversely affected by the proliferation of invasive alien species in novel ecosystems (Leclere et al., 2020). Moreover, biotic invasions pose a serious danger to the spatial landscapes of global biodiversity hotspots and low-human development index (HDI) nations (Early et al., 2016). Furthermore, when combined with climate change and a high rate of deforestation i.e., 13 million hectares worldwide, the negative impacts of IAPs on indigenous or native biodiversity were further aggravated into the loss of an estimated 500 million hectares of tropical forests (Chazdon, 2008; Pysek et al., 2020). Multiple anthropogenic disturbances encouraged the spread of interspecific plant-animal phenotypes in forested landscapes, which in turn negatively impacted the forest succession, resulting in invasion (Chazdon, 2008). The invasion windows created by anthropogenic disturbances allow IAPs to act as passengers along with potential plant traits (Sakachep and Rai, 2021).

IAPs may infiltrate any ecosystem in every bioregion, with varying degrees of intensity, anywhere in the globe; yet, the management dilemma remains an unexcogitable problem (Gentili et al., 2021). By altering the composition and functioning of ecosystems (Zhang and Ye, 2002), leading to enormous financial losses (Xu and Ding, 2003) and accelerating the extinction of species and genetic biodiversity (Qiang and Cao, 2001). Despite their capacity to penetrate any ecosystem in every bioregion with various degrees of potency anywhere in the world, IAP management continues to be an unsolvable problem (Gentili et al., 2021). IAPs disrupt both the environment and people when they arise in unfamiliar settings (Seebens et al., 2017). One of the elements influencing the distribution of IAPs is the number of agricultural sectors used in the food security (Mack et al., 2000; Gilbert and Levine, 2013; Rai et al., 2018).

The introduction of IAPs modifies the physio-chemical environment, increases the likelihood of disturbances, and diminishes native biodiversity, among other consequences (Hulme, 2015; Bellard et al., 2016). Additionally, in high-resource conflict circumstances, IAPs cause the native population to decline or become extinct (Alyokhin and Sewell, 2004; Gallardo et al., 2019). They also have an effect on the environment, the economy, and the welfare of people (Barney et al., 2013; Diagne et al., 2020). Due to environmental changes brought about by human activity over the last several decades, the invasion of IAPs had escalated (Keller et al., 2011). According to Richardson et al. (2000), Hulme, (2007) and Merceron et al. (2017), certain IAPs have the capacity to alter habitat and ecosystem functioning to the point where they endanger ecosystem services and human well-being. IAPS may significantly diminish native biodiversity via a variety of secondary methods, including herbivory, hybridization, ecological change, competition, hunting, and other processes (Gaertner et al., 2009; Waser et al., 2015; Manchester and Bullock, 2000).

With their violent and contentious ability to proliferate in more disturbed and edge habitats or soil with high levels of light and nitrogen (Saunders et al., 1991; Turner, 2005), it is evident that IAPs can eradicate rare native species (Houlahan and Findlay, 2004) and have a major negative impact on ecological processes (Allison and Vitousek, 2004). Furthermore, rather than wilderness areas, IAPs were more likely to penetrate regular ecosystems, which were collapsing due to human dominance (Lindenmayer and Mccarthy, 2001; Guirado et al., 2006; Leyva et al., 2006; McKinney, 2006). Many researchers have conducted studies on the impact of the ecosystem (Liao et al., 2008; Gaertner et al., 2009; Powell et al., 2011; Vilà et al., 2011), but there have yet to be studies of qualitative synthesis in relation to the recipient ecosystem and invading flora attributes (Parker et al., 1999; Thiele et al., 2010; Hulme, 2012). Even though IAPs were recognized to inflict damage globally, many parts of the globe still lack knowledge about IAPs or were in the dark phases of plant invasion (Corlett, 1988; Enmoto, 1999; Meyer, 2000). In order to identify and monitor IAPs in future situations, the creation of an occurrence database is a necessary initial step in the development of invasion biology (Wu et al., 2004; Latombe et al., 2017).

1.3 Invasive Alien Plant Species on Phenology Pattern

Phenology is the study of how the weather and climate affect the recurring life cycles of plants and animals (Schwartz, 2003; Cleland et al., 2007; Ovaskainen et al., 2013; Cara-Donna et al., 2014). The word is derived from the Greek word "phaino", which implies to appear or show. Ecologists have been captivated by phenological research for many years due to their consistent occurrences that cause changes in the community structure when new alien species arrive (Wolkovich and Cleland, 2011). Based on seasonal knowledge of phenophases, phenology may be utilized to investigate the interactions between groups and species in ecological adaptation (Desai and Patel, 2010). Phenology refers to the effective timing of a species' development and reproduction when biotic and environmental components spontaneously choose one another (van Schaik et al., 1993). Both biotic and climatic characteristics were composed of a number of different elements, according to Kumbhani (2021). In contrast to climatic elements such as temperature, photoperiod, and rainfall, which have been studied by Opler et al. (1976), Arroyo et al. (1981), and Borchert (1983), biotic have modified their plant physiology and morphology. There were consequences for ecological evolution in the way that the timing of abiotic effects had impacted phenology phenomena (Forrest and Miller-Rising, 2010). Phenology is associated with many characteristics that were essential to a plant species' or community's ability to compete, including availability and resource acquisition (Driess, 2016). Phenology is often thought of as a temporal niche and is a major predictor of species succession, community assembly, and climate change (Fitter & Fitter, 2002; Cleland et al., 2007; Sherry et al., 2007; Driess, 2016). According to Chuine and Beaubien (2001), phenology restricts the ranges of species, promotes species cohabitation (Fargione and Tilman, 2005), and is adaptive (Volis, 2007). More people were beginning to recognize that IAPs' distinct phenology or phenological sensitivities could be advantageous for them in recently introduced environments (Harrington et al., 1989; Wolkovich and Cleland, 2011; Fridley, 2012).

In the basic ecology of invasion biology, phenology had proven critical to the invasion of IAPs and the ensuing climatic change (Field et al., 2007). The success of IAPs' invasion may have been attributed to their clear advantages over the native flora (Cadotte and Lovett-Doust, 2002). In particular, phenology may be linked to a number of characteristics that were essential to plant competitiveness (Wolkovich and Cleland, 2011). Based on theories like priority effects and vacant niches, some IAPs with different phenological patterns than the native community might benefit from occupying areas with resources where the later species were dormant (Xu et al., 2007; Wolkovich and Cleland, 2011; Durham et al., 2017; Gioria and Osborne, 2018). According to Willis et al. (2010) and Wolkovich et al. (2013), invading species tend to exhibit different phenological responses in hotter climates than native species.

Seasonal changes seem to be connected with the positive results of IAPs (Laube, 2015). An extended blooming season was the outcome of several IAPs using the autumn niche significantly more effectively than native species (Harrington et al., 1989; Pysek and Richardson, 2007; Wolkovich and Cleland, 2011; Knapp and Kühn, 2012; Fridley, 2012). Furthermore, IAPs were more adaptable to temperature changes than native species (Willis et al., 2010; Hulme, 2011) and sprout sooner in the spring (Cleland et al., 2015; Wilsey et al., 2015). It causes a rise in abundance and dispersion (Willis et al., 2010; Hulme, 2011) and permits the IAPs to have full exposure to light (Polgar et al., 2014). Climate-wise, IAPs flower differently from indigenous ones, with more frequent and profuse flowering in invaded areas (Godoy et al., 2009) and afterward flowering profusely (Celesti-Grapow et al., 2003; Lloret et al., 2005). Relocating to a new area may cause changes in the blooming characteristics (Rathcke and Lacey, 1985; Dlugosch and Parker, 2008). Because of evolutionary restrictions, species from the tropical area were known to flower (Wright and Calderon, 1995; Singh and Kushwaha, 2006). The climate of a given area may sometimes serve as a corrective filter for the effective invasion of IAPs (Richardson et al., 2000). This breathable filter therefore serves as the last link in the chain of IAPs that leads to the area that had been invaded (Thuiller et al., 2005). Therefore, climatic variables with seasonal variations in phenophase events need to be examined in order to understand the phenology of IAPS migration towards the extinction of native populations in a new habitat.

1.4 Invasive Alien Plant Species on Soil

The majority of research on the consequences of plant invasions had focused on aboveground fauna and flora (Levine et al., 2003), nevertheless soil organisms play important roles in regulating ecosystem-level processes (Wardle et al., 2004) and soils contain a large portion of the biodiversity of terrestrial ecosystems (Wolfe and Klironomos, 2005). This was due to the fact that belowground diversity researches were subject to methodological limitations, whereas aboveground groups were often easier to see and measure (Belnap and Phillips, 2001). Some of the IAPs were able to change the topography as well as soil properties in the invaded regions by diminishing organic matter, minerals, and moisture retention capacity contributing to more environmental damage (Kumar et al., 2021; Qu et al. 2021; Rai, 2022). Competition of nutrients between native and IAPs also lead to severe changes in the soil properties (pH, calcium and Magnesium) which can be notice afterward (Stefanowicz et al., 2018; Kone et al., 2021; Shiferaw, 2021). The physical, chemical, and biological properties of the soil were also changed by prolonged colonization which might have positive or negative impact effects on soil (Jandova et al., 2014). Sometimes IAPs diverge the soil chemistry and composition of plant communities in their favor in comparison to uninvaded region (Dassonville et al., 2008). The correlation between plant characteristics and soil physical-chemical factors may influence IAP dynamics, resulting in alterations to the soil nutrient pool (Rai, 2022). The protracted colonization altered the physical, chemical, and biological aspects of the soil, potentially resulting in either positive or negative impacts (Jandova et al., 2014). In certain situations, IAPs invasion might help to bring soil conditions closer to uniformity (Qu et al., 2021). According to Ehrenfeld (2003), Daehler (2003) and Soti et al. (2019), once established, invasive species enhance the rhizosphere conditions in the recipient environments and maximize native species' performance relative to invaders by adjusting resource levels and disturbance regimes. The invasion of IAPs may significantly affect soil parameters, including dynamic content, enzymatic activity, phosphorus (P), soil carbon (C), and microbial diversity (Ni et al., 2020). Interactions between invasive exotic plants and their associated biota, both above and below the surface, may vary significantly between their native and recipient settings (Soti et al., 2020). Abiotic tolerance, biotic resistance, propagule pressure, or a mix of these three elements often determine an invasive species' capacity to infiltrate a new environment (Alpert et al., 2000; Levine et al., 2004; Simberloff, 2009; Byun et al., 2018). Plant characteristics including rapid growth, short lifespan, high resource allocation to reproduction, and adaptable utilization of available environmental resources were the main focus of most research on IAPs (Soti et al., 2020). Furthermore, colonization of IAPs often leads to change in soil physico-chemical parameters (Kumar et al., 2021).

Several researches on IAPs have shown that soil organisms affect plant performance as well as the microbiome of invasive species (Volin et al., 2010; Ramirez et al., 2019). Only few have demonstrated the performance of IAPs and the limitations they impose on soil properties- pH (Diekmann and Lawesson, 1999) and soil acidification (Foy, 1984). Thus, knowing how abiotic soil properties differ in IAPs native and invaded regions may help identify successful invasion strategies outside of their natural habitats.

1.5 Research Objectives and Questions

As of now, only two studies have been conducted on the IAPs in Arunachal Pradesh (Kosaka et al., 2010; Singh et al., 2021). Moreover, little is known about the ecological impact of IAPs on regional biodiversity. The state is situated in an area recognized as a biodiversity hotspot region; therefore, creating an occurrence database in each of the protected areas would require a comprehensive baseline study. The goal of the current study is to investigate the structural features of IAPs and how they affect the native flora in the Pakke Tiger Reserve (PTR), in Arunachal Pradesh. Additionally, it suggests solutions to lessen the risks and effects that now result from invasive alien plant species. In addition, a few selected invasive alien plant species will have their phenology, status, and distribution examined. This will help us understand how these plants interact with the surrounding vegetation and adapt to the changing climate. In addition to addressing the ecological effects of IAPs that alter the variety, abundance, and distribution of native flora in a system, this research will provide knowledge about IAPs in PTR to lessen the invasion and explore strategies to prevent it. At the present study site, no studies have been done on the distribution of IAPs and their ecological consequences on the native community. Consequently, the present investigation had been designed with the following objectives and questions in mind:

1.5.1. Research Objectives

The objectives of the current study are as follows:

- To assess the distribution and invasion patterns of selected Invasive Alien Plant species in Pakke Tiger Reserve.
- To study the phenology of selected Invasive Alien Plant species in Pakke Tiger Reserve.
- To assess the ecological impact on native flora in Pakke Tiger Reserve.

1.5.2. Research Questions

- Do distribution and invasion patterns of IAPs vary across different forest types in Pakke Tiger Reserve?
- Is the climatic parameters influence the phenology of Invasive Alien Plant species in Pakke Tiger Reserve?
- How the invaded soils of Invasive Alien Plant species differ from the soil of native flora in Pakke Tiger Reserve?

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Chapter 2

2.0 Review of Literature

Globally, Invasive Alien Plant Species (IAPs) had been widely recognized as a major threat to the environment (Davis et al., 2001; Richardson, 2004; Blackburn et al., 2019; Banerjee and Sankaran, 2023; Cristina-Maria et al., 2023a; Li et al., 2024). These species tend to escaped from its nativity to other natural area where it slowly invades the new habitat (Ricciardi, 2012; Kaufman and Kaufman, 2023), resulting in replacing of native vegetation in the introduced area (Bellini et al., 2024; Liu et al., 2024). The expansion of IAPs is mainly orchestrated by human activities (Barney, 2006) through migration and colonization (Kannan et al., 2013; Fig. 2.1). For example, the colonization of America, Africa and Asia by the European between 15th to 19th century made a gap in the geographic barriers (Essl et al., 2011), which immensely lead to overflow of many invasive species into their colonies (Kannan et al., 2014; Byrne and Warren, 2024).

Given the current state of globalization and rapid environmental change, it is especially important to comprehend how IAPs proliferate (Cristina-Maria et al., 2023b). The routine importation of dangerous non-native organisms - both unintentional and intentional - occurs every day and is projected to cost more than $100 billion annually worldwide (Baquero et al., 2023). Moreover, there is still debate regarding their successful infiltration and future expansion (Sakai et al., 2001; Barney, 2006). Understanding the details autecology of IAPs regarding their allocation in recipient environment and natural constraints restricting further expansion might lead to pinpoint location vulnerable to future invasion (Weber, 1998). According to Pysek and Prach (1995), phytogeographic historical data of IAPs distribution might help in detecting their establishment, dispersal pathway and process of invasion in the introduced region. But detailed census is often ceased to exist (Reddy, 2008), this led biologist to rely on herbaria and floral data of the region, which might be time consuming to rectify the details (Barney, 2006). In addition, their impact on native species, communities and ecosystems are widely known (Vitousek and Walker, 1989; Lodge, 1993; Simberloff, 1996; Mooney and Hobbs, 2000; Sala et al., 2000). Therefore, creating a detail repository of information on IAPs might be a proper solution in tackling this constraint (Wu et al., 2004; Reddy, 2008).

2.1 Genesis of Invasive Alien Plant Species in the World

The study of "Invasion Ecology or Biological Invasion" had a way back to 18th century, where many notable naturalists such as Augustin Pyramus De Candolle, Alphonse De Candolle, Charles Darwin, Joseph Hooker and Charles Lyell mentioned the phenomenon of invasive species (Richardson and Pysek, 2008; Gentili et al., 2021). These species were even described as possible threats to native population by De Candolle (1855) and Darwin (1859) during their time. However, the starting point of biological invasion comes during the first half 1900s when Charles Elton described the population explosion of species in his book "The ecology of invasions by animals and plants" in 1958 (Gentili et al., 2021). Their progression toward recognition was notably gradual before the publication of Charles Elton book because invasive species were not considered a threat at that time (Richardson and Pysek, 2007, 2008) but mostly valued as ornamental or crop (Bell, 2003).

Figure 2.1: Global expansion routes of IAPs after globalization based on literature reviews.

Afterward, naturalist and biologist started to realize the slow destruction of environments by the invasive species (Davis, 2009). By 2000s, several studies exposed the hidden truth of IAPs showing its effects on the global scenarios (Vitousek et al., 1996; Richardson et al., 2000; Maron et al., 2004; Strauss et al., 2006; Pysek and Richardson, 2007). Over the past two centuries and with a marked acceleration in the last five decades, the global proliferation of IAPs had reached a critical level (Seebens et al., 2021). With this revelation, many researchers begin to unearth both negative (Bhatta et al., 2024; Yin et al., 2024;

Mahmoodi et al., 2025) as well as positive impact of IAPs in global platform (Boadie-Ampong et al., 2024; Ullah et al., 2024; Vila et al., 2024). However, maximum studies were inclining toward negativity as they intend more harm than benefit (Sharp et al., 2011; Potgieter et al., 2019). Moreover, they were termed as worst threat after human destruction (CBD, 2002), capable of causing species loss (Lowe et al., 2000) and etiological agent in the diminution of global biodiversity (Butchart et al., 2010; Pysek et al., 2020).

According to Pysek et al. (2020), global estimation of IAPs is around 2500 species, which have gained the status of dangerous non-native plant, accounting for 25% of biological annihilation of plant species (Blackburn et al., 2019). Although these species influences were different from one region to another (Blackburn et al., 2014), yet causes serious damage to the environment in various way- disrupting native richness (Kumschick et al., 2015), alteration of biogeographic domains (Capinha et al., 2015), phylogenetic diversity (Renault et al., 2022), nutrient cycling (Shivers et al., 2023), productivity (Linders et al., 2019), hydrology (Ruhi et al., 2019), trophic structure (Calizza et al., 2021) and functioning of ecosystem services (Castro-Díez et al., 2019). They were also responsible for extinction of 39 species worldwide (Pysek et al., 2020) and a main predator of endemic species (Pysek et al., 2017). According to Schirmel et al. (2016), IAPs have significant impact on the faunal diversity by decreasing abundance, diversity and fitness, with strongest effect on herbivores. Similar results were observed by Fletcher et al. (2019), in which the IAPs reduce the native faunal abundance and impact increases over time, not depending whether exotic or native. However, there were also positive impacts of IAPs on forest fauna in ecosystem near developed areas (Hayes and Holzmueller, 2012). Globally, 63% of the studies concludes that IAPs is the major concern for native population (Pysek et al., 2012) and the leading cause of ecological modification, exceeding hunting, harvesting, and agriculture (Blackburn et al., 2019).

In addition, they can enhance the ecosystem by altering physicochemical properties of soil to their benefit in the invaded region (Weidenhamer and Callaway, 2010; Allen et al., 2011; Chen et al., 2015; Gibbons et al., 2017; Xu et al., 2022). Not only alteration, it modifies the soil seed banks (Lee et al., 2024), fire regimes (Fusco et al., 2022), microbial activities (Wang et al., 2022) and soil structure (Weidenhamer and Callaway, 2010), leading to more profuse invasion (Gaertner et al., 2012). Their degree of impact does not stop at environments but also to humanity (Early et al., 2016; Lazzaro et al., 2018). The effects of IAPs on human wellbeing can be categorized into direct and indirect pathways (Bacher et al., 2018; Shackleton et al., 2019). Direct impact on humans is mainly caused by diseases and infection (Hulme et al., 2014) while monetary losses and abandoning of farmland were indirect ones (Paini et al., 2016), thereby leading to disruption in socio-cultural relation (Pysek et al., 2020). Moreover, these species negatively impact the socio-cultural valuation of designated ecological and geographical areas (Ghermandi et al., 2015). However, the subjective nature of environmental perception poses a significant obstacle to the objective assessment of IAP impacts on socio-cultural valuation (Kueffer and Kull, 2017).

2.2 Distribution and Invasion Pattern

It is a known fact that invasive alien plant species (hereafter IAPs), affect the structure (García-Robledo and Murcia, 2005; Brown et al., 2006), diversity (Brown and Gurevitch, 2004), dynamics of the recipient native vegetation (Hejda et al., 2009; Mollot et al., 2017). Biological invasion by IAPs is often associated with loss of biodiversity (Kenis et al., 2009) and threat to functioning of ecosystem (D' Antonio and Vitousek, 1992; Richardson et al., 2000; Belnap and Philips, 2001; Vila et al., 2010), socioeconomic (Novoa, 2017) and human health (Roques et al., 2009). Over the last few decades, the severity of IAPS effects on biodiversity have grown dramatically large (Pimentel et al., 2001; Butchart et al., 2010) and become a concerning debate among the invasion biologist (Liao et al., 2008; Pereira et al., 2013). Invasion by IAPs have become a major environmental issue and the second largest threat to global biodiversity after loss of habitat and depletion of environment (Hulme, 2007; Peltzer et al., 2010; Simberloff et al., 2013). Moreover, the issues relating with the IAPs is increasing rapidly due to global trade and travels (Groves, 1991; Taylor et al., 2012) and ecosystem were becoming more susceptible to invasion by human mediated disturbance and propagules dissemination (Richardson et al., 2007; Holmes et al., 2008) jeopardizing the stability of the ecosystem and the provision of services (Richardson and van Wilgen, 2004; Vila et al., 2011).

Several studies shows that forest habitats exert strong pressure on invasion because of their natural character, competitive abilities of native species and low propagule pressure in normal or slightly fragmented forest areas (Gonzalez-Moreno et al., 2013), though this narrative is slightly differed in current scenario as they were increasing in forest region whether undisturbed or non-fragmented (Medvecka et al., 2018). The mountain forest system had a rapid change in climatic factors, wide range of different habitats and uneven propagule pressure of IAPs, thereby necessitating habiatat invasibility to understand the invasion success in the region (Giorgis et al., 2016). Distribution and invasion pattern of

IAPs differ strongly between different habitats due to increasing pressure from human interference, variation in disturbances and propagule pressure (Chytry et al., 2009). Forest differing in habitats might reveal plant invasion direction and stages in long term monitoring (Huebner, 2021). Invasion patterns of IAPs in mountain forest resulted in decreasing trends with the increased of elevation, indicating that their species richness might be different from native species (Zhang et al., 2015)**.** However, these IAPs at the lower altitude might filter through the elevation due to worsening climatic changes, and human interferences (Xie et al., 2024). Similarly, these results concurred with the study of Liu et al., (2005) suggesting human and climatic condition might play a role in IAPs distribution in China.

The distribution of IAPs in the New World had increased significantly than the Old World due to influenced by environmental factors (Lonsdale, 1999; Wahyuni et al., 2016). Activities such as trade, transport, landscape fragmentation and modification of atmospheric composition by the humans, facilitate more invasion of IAPs in natural and forest areas (Henderson et al., 2006). These species introduced in other native habitat exhibits more proliferation in invaded areas than the undisturbed region (Hejda et al., 2009). Naturally, these species prefer to invade non forest and riverine forest rather than thick undisturbed forest (Campagnaro et al., 2018). IAPs also facilitate the expansion of other exotic species by altering the species composition, thereby creating a pressure on native vegetation (Medvecka et al., 2018). Studies on the level of invasion in different types of forest habitats were done in broader scale but their outcome is unchanged i.e., more invasion riparian forest due to fragmentation, open canopy and high propagule pressure (Wagner et al., 2017). Although, disturbances were highly connected with invasion but for the riparian forest, artificial disturbances seem to be the main issue in forest habitats (Medvecka et al., 2018). These facilitating results might be due to invasional meltdown, which may have a role in IAPs distribution in forest habitats (Collins et al., 2020). Their impacts resulted in alteration of vegetation comoposition, forest succession, nutrients cycle and reduces in species diversity (Liebhold et al., 2017). With these changes in the recipient environments, these facilitate unfavorable condition for the native vegetation and lead to successful proliferation of the IAPs (Ozolincius et al., 2014). In biological invasion, many researchers specifically work on perspective, distribution pattern and invasion success, though usually unnoticed the expansion of IAPs by habitats (Blackburn et al., 2014; Giorgis et al., 2016). Due to this, study on habitat invasibility had been poorly understood till now (Giorgis et al., 2016).

2.3 Phenophases of Invasive Alien Plant Species

For thousands of years, the subject of phenology had fascinated many environmentalists all around the world (Wolkovich and Cleland, 2011; Piao et al., 2019; Song et al., 2020). With the changing climatic conditions, study of phenology and its shift to changing climate had seen renewed interest among the researchers (Post and Stenseth, 1999; Wolkovich et al., 2014; Fu et al., 2015; Richardson et al., 2018). It is crucial to fully understand the phenological patterns observed in different regions in addition to the underlying explanations behind these patterns for a variety of reasons (Chapman et al., 1999), as the use of phenological data had been widely recognized (Schwartz, 1998; Menzel, 2002). Additionally, these environmental variables, such as precipitation (rainfall) and temperature, affect plant phenology (Sakai et al., 2001; Smith et al., 2012).

The main cause of seasonality changes in plants is precipitation (Korner, 2006; Nord and Lynch, 2009), which can't be unheeded in the phenological studies (Neto et al., 2013). Rainfall promotes a high rate of flowering (Wright and Calderon, 1995), fruiting (Aldrich and Hamrick, 1998) and leaf initiation or flush (Lovejoy et al., 1986) among the plants established at the edge of forest through photosynthetic activities and high insolation (Neto et al., 2013). Nevertheless, temperature also plays an underlying effect on plant phenology through water availability (Shiferaw et al., 2020). According to Rathcke and Lacey (1985) and Ferrara et al. (2017), temperature shows an important role in plant phenology, which helps in regulating the phenological patterns, in which high temperature leads to leaf fall in the edges of forests in comparison to the interior forest (Sizer and Tanner, 1999; Cara, 2006). Moreover, the differences (temperature and precipitation) in the phenology among the plant species led to reduced competition in the plant communities (Shiferaw et al., 2020), growing season length (Cleland et al., 2007) and sustainable food security with stable ecology (Fitchett et al., 2015).

The relation between the phenological patterns of plant species and climatic variables can easily deduce the past or present values of the variables where no data were available (Beaubien and Johnson, 1994), thus the plant species served as the information indicator of local climate data (Diekmann, 1996). Studies on various phenophases like flowering, fruiting, leaf initiation and leaf fall help in deciphering the timing of seasonal duration and phenological behavior of particular species. Following these events during a plant's life cycle such as flowering (Diekmann, 1996; Menzel et al., 2006; Song et al., 2020), fruiting (Heideman, 1989; Chapmann et al., 1999; Wender et al., 2004; Singh and Kushwaha, 2006), leaf initiation and leaf fall (Koptur et al., 1988; Fridley, 2012; Connell and Savage, 2020; Horbach et al., 2023) helps us understand how the species can adapt to its environment and how it works (Tesfaye et al., 2011; Dutta, 2013). In flowering phenology, temperature (Hollister et al., 2005) is facilitated by the association of internal factors (Putterill et al., 2004). Besides temperature, other environmental signals such as, photoperiod (Rathcke and Lacey, 1985), drought (Fox, 1990), day length (Imaizumi and Kay, 2006), soil moisture (Sauer, 1976; Hartmann et al., 2013), rainfall (Rathcke and Lacey, 1985) and snowmelt (Korner, 2003) also regulate flowering in plants (Schnelle, 1955). Sometimes variation in flowering phenology might occur in regional or local variables due to changes in ecological components of an environment (Rafferty et al., 2020). Therefore, flowering phenology plasticity serves as a vital mechanism accountable for the variations in phenological events in response to different environmental factors reacting on the plant (Gugger et al., 2015; Trunschke and Stocklin, 2017).

Similarly, fruiting phenology is limited by abiotic factors, but only a few studies have addressed this relationship (Rathcke and Lacey, 1985), as high temperatures in temperate regions during the summer resulted in significant fruit mortality (Stephenson, 1981) with smaller fruit development (Thompson, 1983). Moreover, in tropics during the dry season, fruit produced were often abnormally small with nonviable seeds (Augsplirger, 1980; Thompson, 1983). Moreover, the environmental factors might have influenced the ripening rates by inducing metabolism in plants (Aalders, 1969). Development timing of fruit is stunted by high temperature (Tukey, 1952), drought or fire-initiated cone dehiscence and relative humidity controls the rate of seed dispersal (Van der pijl, 1972). Fruiting phenology is the least studied in comparison to flowering and leaf phenology because of the difficulties in measuring offspring dispersal success (Rathcke and Lacey, 1985). However, the phenological patterns of flowering and fruiting were found to be interrelated with the changes in annual temperature and rainfall (Smythe, 1970; Foster, 1985; Heideman, 1989). Moreover, there is a correlation between seasonal rainfall and flowering phenology (Opler et al., 1976). Changes occur in environmental variables with the increase of elevation and differences in these variables, i.e., rainfall, temperature and exposure have an impact on the leaf phenology (Koptur et al., 1988; Fogelstrom et al., 2021). In wet forest, leaf phenology might be influenced by rainfall (Augspurger, 1982), as overall synchronization of leafing phenology is less between the species (Milton et al., 1982) though full synchrony within some wet forests were also observed (Koptur et al., 1988). Moreover, high temperature seems to influence the leaf initiation and leaf fall by extending their growing season (Fogelstrom et al., 2021) or by advancing and delaying leafing phenology (Zohner and Renner, 2019; Zani et al., 2020). In addition, the phenological study provides a broad concept to provisional ecological services which is important for conservation actions (Dunham et al., 2018).

In biological invasion, the phenological aspects of invasive alien plant species outcompeting native were not completely understood (Driess, 2016). The invasive alien plant species have created a ruckus all over the world, which continues to exacerbate (Seebens et al., 2017). Many theories suggest that invasive plants have a competitive edge over the native population (Cadotte and Lovett-Doust, 2002), because of functional traits that they possessed (van Kleunen et al., 2010; Mathakutha, 2019). In addition, invasive plants might have some traits that have higher values than coexisting native plants (McAlpine et al., 2008; Dickson et al., 2012). Identifying these specific traits of plants will help in predicting the invasiveness of a species that might become invasive in futures (Sakai et al., 2001; Pysek and Richardson, 2007). One of the important traits in invasive alien plant species is flowering phenology, a key trait for achieving high fecundity (Crawley et al., 1996; Lake and Leishman, 2004; Lloret et al., 2005; Pysek and Richardson, 2007). While the fruiting phenology of invasive species is more favorable in comparison to native at the community level, whether it is overlap or not (Sallabanks, 1993; Gosper, 2004; Godoy et al., 2009; Lediuk et al., 2014). With the increasing climatic condition and disturbances, it seems phenology might have a role in the proliferation of invasive alien plant species (Field et al., 2007). Additionally, invasions were facilitated by the climatic change because of their sensitive phenology (Willis et al., 2010; Wolkovich et al., 2013) and basic niche theory of invasive species (Godoy et al., 2009). Certain fluctuation of resources (Davis et al., 2000), with the opportunity of invasion windows (Drake et al., 2006; Caplat et al., 2010) during the phenological vacant niche allows the invader species more success in rapid colonization in a region through early priority effects and invader plasticity (Wolkovich and Cleland, 2011). The eastern Himalayan region of Indian Himalayan region is one of the fragile ecosystems (Diaz et al., 2003: Kanwal and Lodhi, 2018) in terms of climate change (Cruz et al., 2007; Xu et al., 2009). In Indian Himalayan region, many studies regarding phenology response on climate change have been done during the recent years, especially on trees and shrubs (Shukla and Ramakrishnan, 1982; Ralhan et al., 1985; Rawal et al., 1991; Kikim and Yadava, 2001; Saha and Sundriyal, 2010; Bisht et al., 2014; Negi et al., 2021) following IPCC (2007).

2.4 Ecological Impacts on Soil

Impacts on above-ground terrestrial plant biodiversity and their dynamics by invasive species is globally sound but their effects on the below ground structure and functioning dynamics of soil system often received less attention (Kourtev et al., 2003; Raizada et al., 2008). The success of invasive species in below ground system is related to the interaction between plant and soil biota, understanding this hypothesis can clearly show the effect of IAPS on soil biota (Reinhart and Callaway, 2006). Sometimes, the native species show negative plant- soil feedback in the same soil for consecutive generation while the IAPs response were likely positive plant soil-feedback in the same soil for several years (Vestergard et al., 2015), clearly indicating the soil biota have an important role in biological invasion (Yang et al., 2013). Plant and soil relationship is inter-dependent and changes each other over time, as they were important parts of terrestrial ecosystems (Wang et al., 2007). Therefore, IAPs introduction in native environment led to changes in many ecosystem services provided by plant communities (Castro-Diez et al., 2011). Any changes brought by IAPs in soil lead to sudden change in vegetation composition and their structure and these modification in physio-chemical properties led unbalance in Soil pH, moisture, temperature, organic matter and microbial activities (Ahmad et al., 2019).

IAPs with the capacity to alter the local features of physio-chemical properties of soil through litterfall and root penetration (Jourgholami et al., 2021), have more advanced functional traits such as higher elevated and specified leaf, rapid growth as well as higher use efficiency thn native species (Kumar et al., 2021). Moreover, IAPs effects on soil were often associated with species-specific and site specific (Dassonville et al., 2008), for example *Chromolaena odorata* was able to reduce the heavy metals in polluted soil in Nigeria (Ayesa et al., 2018) and able to improve the soil quality of fallow cropping system of Lamto reserve in Ivory Coast (Tondoh et al., 2013). Similarly, Kone et al. (2012) also observed that *C odorata* in savanna improved the soil biology. Moreover, these species alter the soil in their favor according to its need to growth in tropical ecosystem (Wei et al., 2017).

Morphologically, *C odorata* grows to 2- 3 m in height (Zachariades et al., 2009), but it can reach up to 2-6 m with support from other vegetation (Balangcod and Balangcod, 2021). The leaves were triangularly ovate, brittle stem, fibrous root system could penetrate 0.3 km inside the soil (Holm et al., 1977; Henderson, 2001; Zachariades et al., 2009; Sirinthipaporn and Jiraungkoorskul, 2017), white bluish-lilac flower colors (Moody et al., 1984) and narrow and oblong seeds with white hairs that turn to brown as it becomes dry (Zahara, 2019). The plant thrives in a wide range of well-drained soil (Zachariades et al., 2009; Balangcod and Balangcod, 2021), with low fertility (Mandal and Joshi, 2014a) but does not tolerate shades or semi shades (Binggeli, 1996), waterlogged areas (Crutwell, 1972), frost (Binggeli, 1999) and high year-round rainfall areas (Kriticos et al., 2005). However, once established in any disturb areas such as open woodland and forest margins and gaps (Goodall and Erasmus, 1996; Prasad et al., 1996), it competes aggressively with the native vegetation leading to shading out (Lin et al., 2009). Additionally, it expands vigorously during rainy season; as a result, the vast number of fertile seeds produced leading to successful invasions (Swaine et al., 1997; Honu and Dang, 2000; Prashanthi and Kulkarni, 2005). Although, McFadyen (1988), Binggeli (1999), Kosaka et al. (2010), and CABI (2024) stated that their favorable invasion altitude ranges from 1000-1500 m but distributed up to 2000-3000 m above the sea level (Timbilla, 1998; Gautier, 1992). Moreover, *C. odorata* is unpalatable and noxious to the livestock (Aterrado and Bachiller, 2002), leaving them as healthy invader as native were grazed by the animals.

Eastern Himalaya, an ecological fragile mountain ecosystem with vulnerability to disaster (Diaz et al., 2003) is one of the global biodiversity hotspots and eco regions of the world (Myers et al., 2000; Brooks et al., 2006) as the region is formed due to convergence of the Palearctic and Indo-Malayan realm (Hua, 2012). Moreover, the region is beset with the issues of the plant invasion by *C. odorata* (Rai and Singh, 2021). These species were introduced in India (Calcutta Botanical Garden) as ornamental plant in 1845 (Muniappan and Bamba, 1999) but spread to the north eastern part of India during the first world war (Singh, 2004), later found way to eastern Himalaya (Tripathi et al., 2006). The species had become a dominant and menace species in the lower altitude's region of the eastern Himalaya with its ability to alter the dynamic ecosystem of native population (Lamsal et al., 2018).

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Chapter 3

3.0 Study Area

The present study area lies in the north eastern region of the Eastern Himalayan. The biggest Himalayan state in India, Arunachal Pradesh (AP; 26°28'-29°30'N and 91°30'-97°30'E), is located in this Eastern Himalayan region. It had an area of 83,743 km² and is physically isolated from the rest of the nation by impassable terrain. The international boundaries of Tibet to the north, Bhutan to the west, Myanmar to the east and southern border with Nagaland and Assam encircle this hilly state. The state had a population of about 13.82 lakh, which is roughly 0.11% of India's total population. The population density is exceptionally low at around 17 people per km², in contrast to India's national density of 382 people per km² (Census, 2011). The history of Arunachal Pradesh begins in the British Indian province of Assam, when it was known as the North-East Frontier Tract (IOP, 2019). It was renamed the North East Frontier Agency (NEFA) in 1951 upon post-independence and becomes a union territory in 1972. However, it again changes its name to Arunachal Pradesh, and becomes a complete state in 1987 with Itanagar as its capital (Osik, 1996). Moreover, the state is located in the transition zone between the Himalayan and Indo-Burmese regions which is recognized as the one of the 25 mega biodiversity hotspots of the world (Myers et al., 2000) and also among the globally important eco-region out of 200 (Olson and Dinerstein, 1998). The state is considered as the paradise of the botanists due to huge repository of plant diversity (Kanwal and Lodhi, 2018) and many earlier explorers have left a vivid description as well as fascinating accounts of the region (Furer-Haimendorf, 1962). The recorded forest cover of the state is 66431 km², which constitute 79.32% of geographical area with reduction of 257 km² from the previous 2019 recorded data (FSI, 2021). The state had one-third of the region under the Himalaya and Indo-Myanmar Biodiversity Hotspot, with five globally important eco-regions (Mittermeier et al., 2004). Moreover, biogeographic specific location of these regions contributes exceptionally high species diversity as the eastern Himalaya is the most species rich zone in Himalaya zone (Page et al., 2022).

According to Chowdhury et al. (1996) nearly 50% of Indian flora occurred in this hilly state with an estimation of 7000-8000 flowering species (Datta, 2000). More than 500 orchid's species have been recorded from the state (Chowdhury, 1998). Moreover, 100 species of mammalian and over 600 species of bird species have been recorded from Arunachal Pradesh (Singh, 1994; Datta, 2000). Many ecologists have recorded several new rare species and first report from India (Captain et al., 2019; Wahlsteen and Borah, 2022; Borah et al., 2024) and yield many important discoveries (Kumar and Singh, 1998), yet many regions were still unexplored. Arunachal Pradesh is inhabited by 26 major tribes and 110 sub tribes with each having different cultural and traditional beliefs (Tag et al., 2021). The ethnic tribes largely depend on wetland rice cultivation and shifting or Jhum cultivation followed by hunting, fishing and collection of forest products from the forest which also forms a part of their economy (Nimachow et al., 2008). Some of the major tribes were Aka, Adi, Apatani, Galo, Tai Khamti, Puroik, Mishimi, Monpa, Nyishi, Nocte, Sajolang, Singpho, Tagin, Tangsa and Wangcho (Bushi et al., 2021). These tribes developed a mutual relation with the surrounding forest in order to sustain their livelihood and have a good knowledge about traditional medicines because of their remoteness and majority of rural population (Nimachow et al., 2008; Bushi et al., 2021). Additionally, the plant and its part were used for food supplements, dyeing cloths, veterinary medicines, tribal rituals, handicrafts and local beverages (Murtem, 2000; Sarmah et al., 2000; Tag and Das, 2004).

3.1 Study Site

The intensive study was conducted in the core area of Pakke Tiger Reserve (PTR; 26°55’ to 27°15’N: 92°35’ to 93°10’E; Fig. 3.1) in Pakke Kessang district, Arunachal Pradesh). Pakke Tiger Reserve is located in the western part of sloping and rugged mountains region of Arunachal Pradesh in the Eastern Himalayas (Photo plate 3.1). PTR covered a total geographical area of 861.95 km², which accounts for 9.04% of the total protected areas of this eastern Himalayan state. In addition, PTR covers 45% of the total areas of the newly formed Pakke Kessang district (1932 km²) of the Arunachal Pradesh state. The story of PTR started with the formation of Pakhui Reserve Forest on 1st July 1966, which was later, converted into game reserve in 1977. However, in 2001, the reserve was renamed as Pakhui Wildlife Sanctuary but soon changes its name to Pakke Tiger Reserve on 2002 with the declaration of National Tiger Conservation Authority as 26th Tiger Reserve under Project Tiger. The PTR is completely circled from all sides- to the south and south-east, 349 km² long stretches is bounded by Nameri Tiger Reserve and National Park, Assam. To the east, the Pakke River and Papum Reserve Forest (1064 km²); to the west, the Bhareli or Kameng River, Doimara Reserve Forest (216 km²) and Eaglenest Wildlife Sanctuary; and to the north, by the Bhareli River and Shergaon Forest Division. Moreover, until 1996, selective logging was occurred on commercial scale in the reserve.

The terrain of the PTR and adjoining areas was rugged and undulating hilly mountain. The elevational ranges start from 150 m to 2050 m above the sea level. The reserve is governed by three Range offices namely Seijosa Range office, Tippi Range office and Riloh Range office, with their own area demarcated for jurisdiction. PTR have two major rivers i.e., Pakke and Bhareli or Kameng, which run through the reserve from the east, west and north direction. Moreover, PTR have three important streams inside namely-Khari, Upper Dikorai and Nameri, all running in southerly direction. Most of the higher hills and rugged mountains were in the northern side of the reserve with southward slope toward the Brahmaputra valley. In addition, the central and northern part of the reserve remains inaccessible due to rugged terrain and dense vegetation. In the south eastern side of the reserve, more than 20 villages and small settlements adjacent to the Pakke River with a population of more than 4000 adults' villagers were located. The two main rivers of the reserve acts as obstacle to human disturbances, though villagers venture deep inside the reserve to hunt, fish, and collects forest produces like canes, bamboos, and trees. With the level of penetration, most of the PTR forest had easy access to nearby villagers except for the forest in southern boundary because of its dense vegetation, Pakke River and hilly terrain.

Photo Plate 3.1: Frontal entrance view and undulating landscape of Pakke Tiger Reserve.

3.2 Geology, Rock and Soil

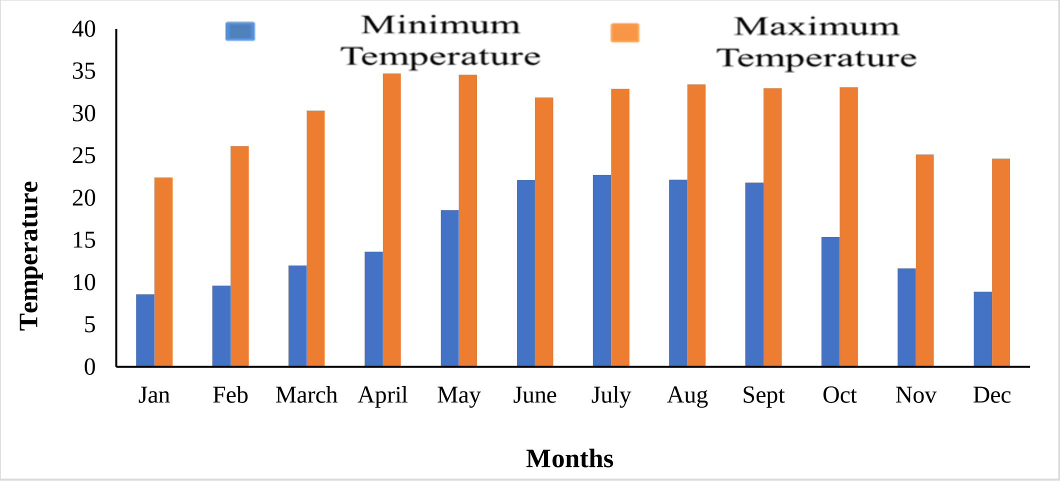
Majority of the soil in the Pakke Tiger Reserve was loam sandy soil with lower water holding capacity. Considering the significant amount of water, the soil receives throughout the year; it was often porous in nature. The hills have somewhat dep, wet, loamy, productive soil with a top layer of humus (Datta, 2001). There were patches of thin soil with pebbles and stones underneath. A layer of sandy loam with different depths and an upper layer of humus covers several small slopes on the sides of foothills and beside rivers and streams (Kumar and Solanki, 2009). The subsoil contains boulders and pebbles. Moreover, there were four types of rocks found in Pakke Tiger Reserve- low grade metamorphic rock, Bicham group of sedimentary rock, Gondwana group of gritty sandstones and carbonaceous shale and sedimentary rocks of tertiary period (Kesari, 2010). The majority of rock discovered in the reserve riverbeds were tertiary sedimentary rocks made up of of conglomerates, pebble and boulder beds, hard stone, and clay and silt-stone.

3.3 Climate and Rainfall

The Pakke Tiger Reserve experiences a tropical and subtropical climate and had cold weather from December to February. Moreover, rainfall occurs throughout the year because of the south-west monsoon (May- September) and the north-east monsoon (December-April; Datta, 2001). Almost 70-80% of rainfalls were received during the period of south west monsoon with thunderstorm often occurring in the months of March-April. However, there was relatively drier period from October to March where many deciduous trees start showing leaf fall (Kumar and Solanki, 2009). The Pakke Tiger Reserve had an average annual rainfall of 2500 mm and the velocity of wind was generally moderate. During the months of May to July, the reserve experience hottest period, with the average maximum temperature crossing above 30°C. In addition, the reserve climate was very warm and humid during the months of June to September. The mean minimum temperature was 15.58±5.37 and the mean maximum temperature was 30.19±4.18 while the average annual relative humidity was 78.48 based on the data from 2019 to 2022, 77.15±8.81 in 2021 (phenology study year). The meteorological data (precipitation, relative humidity, minimum and maximum temperature) from January 2021 to December 2021 were recorded from west bank location (26°56'40.9"N; 92°58'36"E) at an elevation of 566 (Fig. 3.1). These data were obtained from the NASA Earth Science/Applied Science Program-funded POWER Project at the NASA Langley Research Centre (https://power.larc.nasa.gov/).

3.4 Vegetation Types

The vegetation in the entire region was categorized as Assam valley tropical semi evergreen forest (Champion and Seth, 1968). In some areas, evergreen and semi-evergreen vegetation types blend seamlessly into each other. According to Champion and Seth (1968) and FSI



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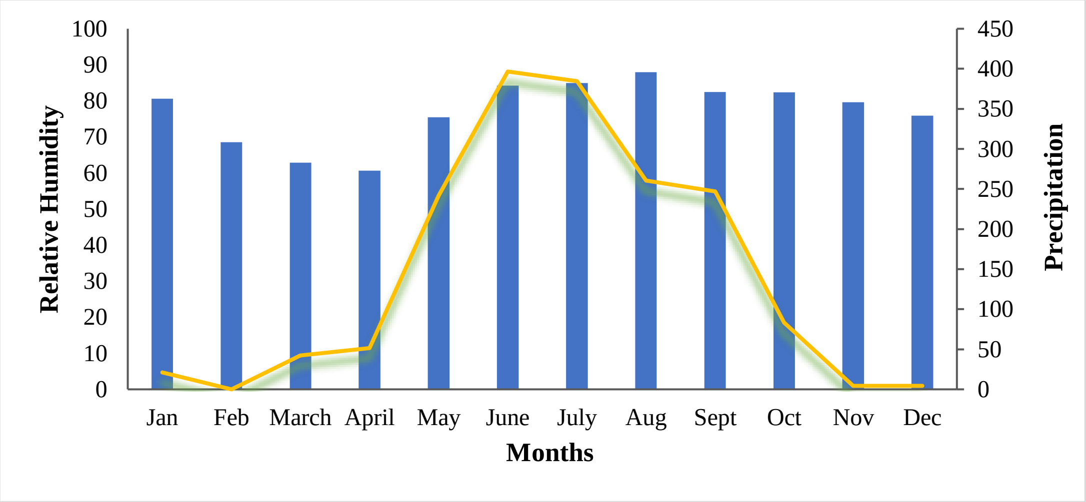

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Figure 3.1: Climatic parameters (A- Mean monthly maximum and minimum temperature and B- monthly precipitation and relative humidity recorded during the phenological observation in Pakke Tiger Reserve.

(2009), the core research area encompasses five main forest types in the Pakke Tiger Reserve (Photo plate 3.2): Assam Alluvial Plain Semi Evergreen Forest (AAPSEF), East Himalayan Mixed Coniferous Forest (EHMCF), East Himalayan Subtropical Wet Hill Forest (EHSTWH), Non-Forest (NF) and Riverine Forest (RF). The forest had dense vegetation and multistoried with epiphytic flora, woody lianas, climbers, shrubs and herbs. The vegetation was very dense with high floral diversity. A total of 234 woody species have been reported from the lower region of the reserve by Datta and Goyal (1997). However, the reserve was expected to harbor 1500 vascular species in which 500 species would be woody species (Datta, 2000). The forest had structural layered system with emergent species emerging out of all the species. The main emergent species in PTR were *Tetrameles nudiflora, Ailanthus integrifolia* and *Liquidambar excelsa* and they were of economic value (Singh, 1991). Hill tops and higher reaches were dominated by *Castanopsis* spp. and *Mesua ferrea* while the stream moist areas have profuse growth of bamboos, cane and palms. Isolated group of *Bombax ceiba, Albizia* spp. and *Duabanga grandiflora* were common near the rivers. The cover area of in each forest types of Pakke Tiger Reserve were calculated with the help of ArcGIS 10.6.1.

3.4.1 Assam Alluvial Plain Semi Evergreen Forest

The Assam alluvial plain semi evergreen forest starts from the riverbank and lowland foothills up to 600 m elevation of the reserve. The forest was represented by tall deciduous and semi evergreen trees with less uniformity in the top layer followed by evergreen trees, shrubs, climbers and lianas. This section of the forest were dominated by *Tetrameles nudiflora, Bombax ceiba, Neolamarckia cadamba, Canarium strictum, Elaeocarpus aristatus, Liquidambar excelsa,* and *Gmelina arborea.* Middle layer of this forest was very dense and with variety of undergrowth including canes and palms. *Livistona jenkinsiana* and some of the cane species occur in this mid-storey. Some of the common tree species- *Ficus* spp.*, Monoon simiarum, Pterospermum acerifolium, Pterygota alata and Cinnamomum glaucescens* grows in association with the shrub's species - *Coffea benghalensis, Holmskioldia sanguinea, Strobilanthes* spp., *Clerodendrum* spp., *Chromolaena odorata* and *Urena lobata.* Moreover, many epiphytic orchid species including *Arundina graminifolia* were recorded from this forest. This forest had the highest presence of invasive alien plant species (IAPs) in the reserve. The total cover area of this forest was 390.69 km² and had a maximum coverage in the reserve.

3.4.2 East Himalayan Mixed Coniferous Forest

The east Himalayan mixed coniferous forest ranges from 400 to 1300 m elevation and had a total area of 364.51 km². It was represented by dense evergreen forest with mixed deciduous trees such as *Magnolia* spp., *Ailanthus integrifolia* and *Liquidambar excelsa* which were the dominant emergent tree species. The middle storey were composed of *Syzygium* spp., *Dysoxylum gotadhora*, *Cinnamomum* spp, *Musa* spp. and *Phoebe lanceolate.* Moreover, there was dense undergrowth of shrubs (*Melastoma malabathricum,* *Clerodendrum colebrookeanum*, *Chromolaena odorata*, *Phlogacanthus spp.* and *Dendrocnide sinuata*) and herbaceous layer (*Ageratum conyzoides*, *Hellenia speciosa*, *Eclipta prostrata, Sonchus oleraceus* and *Acmella paniculata*). In addition, most of the trees were in closed association with epiphytic species.

3.4.3 East Himalayan Subtropical Wet Hill Forest

The east Himalayan subtropical wet hill forest occurs at the altitudes range of 900-2050 m above the sea level, with a total cover of 79.81 km². The hill forest was dense evergreen forest with dominant deciduous trees of good height and density. *Castanopsis* spp., *Magnolia hodgsonii*, *Mesua ferrea* and *Kydia calycina* were some of the dominant tree species in the top layer. Middle storey section of this forest was easily noticeable because varieties of medium sized evergreen trees (*Schima wallichii, Magnolia champaca, Phoebe lanceolate, Morus macroura, Musa* spp.*, Cinnamomum* spp. and *Stereospermum chelonoides*) which were very catchy for the onlookers. In the lower section, shrubby undergrowth (*Clerodendrum* spp., *Impatiens* spp., *Strobilanthes* spp. and *Chromolaena odorata)* and herbaceous layer were always present but grasses was absent in this forest. Several climbers, ferns and orchids were also present in this forest.

3.4.4 Non-Forest

The non-forest in the Pakke tiger reserve does not have forest canopy cover though it spread a total of 17.46 km² of the reserve. Most of the areas were covers with tall grasses (*Cyperus* spp., *Saccharum* spp., *Imperata cylindrica* and *Thysanolaena latifolia*), shrubs (*Chromolaena odorata*, *Crotalaria pallida*, *Citrus medica* and *Urena lobata)* and herbs (*Ageratum conyzoides, Bidens pilosa, Crassocephalum crepidioides, Cuphea carthagenensis* and *Mimosa pudica*) with one or two tree species (*Albizzia* spp. *Dellenia indica, Macaranga* spp. and *Duabanga grandiflora*) on it. These forests were always near the riverine forest and served as the grazing point of elephant, sambar, deer and small mammals. The altitudinal range of this forest was somewhat 150-250 m above the sea level.

3.4.5 Riverine Forest

The riverine forest (RF) was the forest along the main rivers and its streams inside the Pakke Tiger Reserve and it had the least cover area of 9.44 km², out of total area. Species like *Albizzia* spp., *Bombax ceiba, Canarium strictum, Dillenia indica* and *Duabanga*

Photo Plate 3.2: Dominant Forest types of Pakke Tiger Reserve- (A) Assam Alluvial Plain Semi Evergreen Forest, (B) East Himalayan Mixed Coniferous Forest, (C) Non-Forest, (D) Riverine Forest.

grandiflora form the upper layer storey, whereas *Syzygium* spp., *Micromelum* spp. and *Holmskioldia sanguinea* represented the next storey level.Moreover, they were in closed association with *Alpinia* spp., *Hedychium* spp. and *Saccharum* species. The elevational range of riverine forest was 150 m above the sea level.

3.5 Fauna

The Pakke Tiger Reserve was home to different varieties of birds (296), butterflies (284), snakes (46), mammals (60) and many other small species (Datta and Goyal 1997; Datta et al., 1998; Birand and Pawar, 2004; Sondhi and Kunte. 2014; Rambia and Rathore, 2022). The region had great mammal diversity because of its specific location. The PTR was home to six endangered mammal species, as listed by the IUCN Red List (*Axis porcinus, Elephas maximus, Panthera tigris, Prionailurus viverrinus, Cuon alpinus* and *Manis pentadactyla*) and ten vulnerable species (*Bos gaurus, Rusa unicolor, Trachypithecus pileatus, Nycticebus bengalensis, Neofelis nebulosa, Pardofelis marmorata, ursus thibetanus, Arctictis binturong, Aonyx cinerea* and *Lutrogale perspicillata*). Moreover, three large cat species (*Panthera tigris, P. pardus* and *Neofelis nebulosa*) share space with two canids (*Cuon alpinus* and *Canis aureus indicus*) in this reserve. In addition, the reserve give sanctuary to 296 bird species (Datta et al., 1998; Birand and Pawar, 2004), in which six of them were globally threatened according IUCN List (*Leptoptilos javanicus, Carina scutulata, Gyps bengalensis, Haliaeetus leucoryphus, Aceros nipalensis* and *Mulleripicus pulverulentus*). Also, four species of hornbills were found in this reserve, three species (*Buceros bicornis, Aceros undulates, Anthracoceros albirostris*) were easily sighted in lowland and foothills but *Aceros nipalensis* was only sighted in the northern part of reserve which was a highed elevation zone. There is also report of three pheasant (*Gallus gallus, Lophura leucomelanos* and *Polyplectron bicalcaratum*) from the region (Selvan et al., 2013). Furthermore, there had been report of six turtles in which three were endangered/vulnerable species (*Pyxidea mouhotii, Pangshura sylhetensis* and *Nilssonia hurum).* Several studies have also reported significant faunal diversity from PTR (De et al., 2006). There were four primates (Kumar and Solanki, 2008) in PTR - the rhesus macaque (*Macaca mulatta*), Assamese macaque (*M. Assamensis*), the capped langur (*Trachypithecus pileatus*) and the Bengal or northern slow Loris (*Nycticebus bengalensis*). Furthermore, PTR was home to up to sixteen different species of mongooses, weasels, and viverrids.

3.6 Description of the Selected Invasive Alien Plant Species for the Study

A preliminary study was conducted in 36 different locations in the study region covering different forest types (AAPSEF, EHMCF, EHSTWH, NF and RF) along the accessible trails in the forested areas as well as motorable roads. Based on the preliminary study, nine IAPs were selected for the study established on the occurrence of these species and following Botanical Survey of India invasive list (BSI, 2020) in different forest types of Pakke Tiger Reserve. Brief description on each of the selected IAPs were given below (Photo plate 3.3):

3.6.1 *Ageratum conyzoides* L.

Ageratum conyzoides (Asteraceae) is an erect annual herb, endemic to tropical America and expanded globally in tropical and subtropical regions (Wagner et al., 1999; Reddy, 2008). These species were included in the list of top hundred worst invaders (GISD, 2010) and had the capability to colonize any open degraded areas i.e., agriculture fields, forests, wasteland and pasture (Kohli et al., 2006). The word *Ageratum conyzoides* comes from the Greek word "*ageras*" which mean non ageing while the species name derived from the word "*Konyz*", because of its resemblance to *Inula helenium* L. (Kissmann and Groth, 1993). The English name- "Billy goat weed or Goat weed" comes from its unusual scent, which was similar to that of a male goat (Okunade, 2002). *Ageratum conyzoides* can grow in variety of habitats as they were highly adaptable, particularly the disturbed sites (Kohli et al., 2006). Currently, these species were distributed throughout the world (Xaun et al., 2004), reaching up to the elevation of 3000 m above sea level in the eastern Himalaya (Kosaka et al., 2010). In India, prior existence had been reported by Hooker (1882) in "The Flora of British India", but it was believed to be introduced during 1960s as an ornamental plant which later attained the status of invasive in national level (National Focal Point for APFISN, 2005). Moreover, there had been report of *Ageratum conyzoides* becoming menace in different region of India (Kaur et al., 2012), especially north western Himalaya and eastern Himalaya (Kohli et al., 2006; Dogra et al., 2009). Despite the invasive reputation, these species have been widely used in traditional medicines throughout the world (Ming, 1999; Okunade, 2002; Santos et al., 2016; Kaur et al., 2023).


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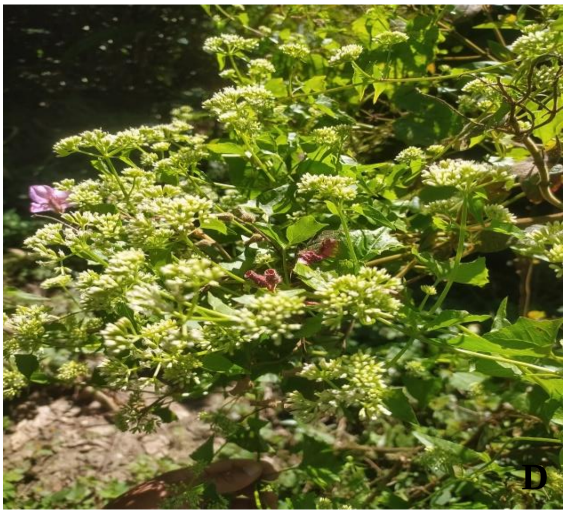
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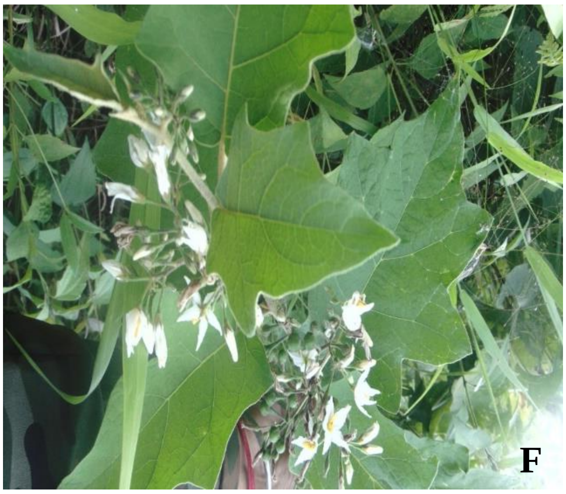
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Photo Plate 3.3: Selected IAPs for the study (A) *Ageratum conyzoides*, (B) *Bidens pilosa*, (C) *Chromolaena odorata* (D) *Mikania micrantha*, (E) *Synedrella nodiflora*, (F) *Solanum torvum*, (G) *Solanum viarum*, (H) *Triumfetta rhomboidea* and (I) *Urena* Lobata

3.6.2 *Bidens pilosa* L.

Bidens pilosa (Asteraceae) is an erect annual herb, native to tropical America which widely distributed throughout the tropical, sub-tropical and warm temperate region of the world (Ballard, 1986; Reddy, 2008; Bartolome et al., 2013). These species can invade a wide range of habitats such as grasslands, forest margins, secondary forests, wastelands, wetlands, roadsides, disturbed land and agricultural fields (GISD, 2024), which led to outcompete the native flora and fauna in the introduced region (Ng et al., 2011; Davidse et al., 2018). *Bidens pilosa* is considered as obnoxious, ruderal invasive weed globally because of explosive reproductive, reducing yield of the next standing crop and capacity to flourish in almost any environment condition (Holm et al., 1977; Young et al., 2010; Silva et al., 2011; Arthur et al., 2012; Kato-Noguchi and Kurniadie, 2024). However, its introduction is unknown in India with probability during 1800s, though Clarke (1876) and Hooker (1882) mentioned their presence in their respective book “Compositae Indicae” and “The Flora of British India” as an ornamental plant. Moreover, this species was listed as highly invasive and environmental nuisance in more than 40 countries of the world (Davidse et al., 2018) and have reached 3000 m altitude in the Indian Himalayan region (Kosaka et al., 2010). Considering this plant’s detrimental effects on the ecosystem, it was frequently utilized in traditional medicines all around the world (Silva et al., 2011; Arthur et al., 2012; Kato-Noguchi and Kurniadie, 2024).

3.6.3 *Chromolaena odorata* L.

Chromolaena odorata is a free-standing invasive shrub species of Asteraceae family, also referred as Siam weed due to its “spreading in from Siam” (Burkill, 1935). This species can invade different types of ecosystems such as tropical rainforests, humid tropical and subtropical forests (McFadyen and Skarratt, 1996; Raimundo et al., 2007) because of wide tolerance of climate (Kriticos et al., 2005). In addition, *C. odorata* had been listed in the 100 worst invasive alien plants and troublesome weed in the world (Lowe et al., 2000; Yu et al., 2014). *C. odorata* introduction to Calcutta Botanic Garden in India as an ornamental plant in 1845 was reported as first entry to Asia, later spread to neighboring countries (Muniappan et al., 2005). *C. odorata* had caused serious problems in the vacant areas where it had invaded such as roadsides, open spaces and forest clearings (Azmi, 2002). The species can grow in all types of soils with low fertility where disturbance had been created (Mandal and Joshi, 2014a), reaching an elevation up to 2000-3000 m above the seas level (Gautier, 1992). However, it had reached up to 1500 m elevation in Indian Himalayan region, which was the most fragile mountain ecosystem in the world (Diaz et al., 2003; Kosaka et al., 2010). Moreover, the eastern Himalaya region is more prone to invasion by *C. odorata* due to increasing anthropogenic disturbances and climate changes (Barik and Adhikari, 2012). Though the species is regarded as the most invasive and harmful species in the world, it had many medicinal importance- wound healing (Sirinthipaporn and Jiraungkoorskul, 2017), antipyretic and analgesic (Owoyele et al., 2008), anti-inflammatory (Suksamrarn et al., 2004), antimicrobial (Stanley et al., 2014) and diuretic (Zahara, 2019).

3.6.4 *Mikania micrantha* Kunth

The climbing perennial weed *Mikania micrantha,* which is endemic to tropical America, belongs to the Asteraceae family is the world most notorious weed (Cronk and Fuller, 1995; Day et al., 2016). *M. Micrantha* was considered as one of the top ten worst weed in the South-east and South Asia (Zhang et al., 2004) and also listed in top 100 worst weed in the world (Lowe et al., 2000). Upon establishment, it eradicates adjacent plant by limiting light under its canopy (Haung et al., 2000), thereby posing a serious threat to the biodiversity (Wang et al., 2004; Zhang et al., 2004). *M. Micrantha* can thrive in any habitats -along the stream, roadsides, forest edges, fence-line, pasture and plantation (Day et al., 2012). Additionally, it can tolerate flooding which may be one of the important factors in their distribution (Yue et al., 2019) and can reach up to 3000m elevation above the sea level (Holm et al., 1977). These species were introduced into India during 1940s because of its vibrant growing nature to use as camouflage for military equipment or as cover plant for tea plantation (Reddy and Raju, 2009; Tripathi et al., 2012). According to Parker (1972), these species was troublesome in the northeast and southwest part of India. *M. Micrantha* can also be utilized in many ways, i.e., as rat poison (Holmes, 1975), as topical ointment (Sankaran, 2015), to treat cut and nausea (Day et al., 2016), as anticancer and anti-inflammatory (Sheam et al., 2020).

3.6.5 *Solanum torvum* Sw.

Solanum torvum is an invasive perennial andromonoecious shrub (Chou et al., 2012), native to West Indies (Welman, 2003), very common in forest clearing, pastures and disturbed areas (Martina et al., 2021). According to Yang et al. (2014), *S. torvum* was considered as invasive plant in more than 32 countries globally because of its weedy and opportunistic characters such as plentiful fruits with large number of seeds (Welman, 2003). These species were adaptive across various soil types from near sea level to an altitude of 2000 m and very drought-resistance (Lim and Lim, 2013). Moreover, *S. torvum* had become naturalized across the humid tropics and subtropics regions of the world (Francis, 2004). It was first introduced in India in 1825 (CABI, 2024) and had become culturally importance due to its multiple uses (Areces-Berazain, 2016). The species can tolerate mild shadow or partial shade throughout the day and grows well in full sunlight, but it cannot thrive under a closed forest canopy (Francis, 2004). However, every portion of the plant was used in traditional medicine to cure various ailments, including anemia, malaria, diabetes, respiratory disorders, skin infections, jaundice, epileptic, hypertension and snakebites (Fui, 1992; Welman, 2003; Schippers, 2004; Kala, 2005). In addition, fruit of these species was used as vegetables in different parts of the world (Lim and Lim, 2013).

3.6.6 *Solanum viarum* Dunal

Solanum viarum, commonly known as tropical soda apple, is a perennial seed propagated weed which belongs to the family of Solanaceae (Randall, 2012). This species is native to tropical America (Reddy, 2008) and later introduced to Africa, India, Nepal, China, West Indies and Mexico (Chandra and Srivastava, 1978; Coile, 1993; Mullahey et al., 1993), where it had invaded both agricultural and natural ecosystem (Mullahey and Akanda, 1996). In addition, *S. viarum* is expected to disseminate to several other subtropical regions around the world (Nee, 1991; Bovini et al., 2014). However, its introduction to India is unknown, though GISIN (2008) and Singh et al. (1998) confirmed it's throughout distribution from the Himalayan foothills in the north to the Nilgiris in the south of India. The species had a reach up to 2000 m elevation in the Himalayan region (Parker, 1992) and anticipated to penetrate the whole eastern Himalayan Forest region of Asia (Qi et al., 2023). *S. viarum* is an aggressive invasive plant which can outcompete native vegetation by crowding or shading them out (Waggy, 2009; Medal et al., 2012), thereby impacting the germination and establishment of native species (Medal et al., 2012). Despite its therapeutic importance in treating cancer, rheumatic, arthritis and Addison's disease (Chandra and Srivastava, 1978; Satyabrata et al., 2000; Nayak and Patil, 2001), the loss of species, harm to human health and livestock and financial losses it generates in many ecosystems ae quite concerning (Singha et al., 2016).

3.6.7 *Synedrella nodiflora* (L.) Gaertn.

Synedrella nodiflora is a dominant invasive weed (Maryanto et al., 2021), native to West Indies (Reddy, 2008; Wagh and Jain, 2018) and a member of Asteraceae family (Gaertner, 1791). The name of this plant is derived from Greek word "*Synedros*"- meaning placing together, i.e., crowdedness of small flower and species name "*nodiflora*" describe the clustering of flower around the nodes (Usharanai and Raju, 2018; Tun, 2020). *S. nodiflora* had been reported as the troublesome weed in seven countries (Holm et al., 1991) and now thoroughly distributed in warmer tropical and subtropical region of 50 Countries including the Australia, China, India, Indonesia, Philippines and the Caribbean (Holm et al., 1997). *S. nodiflora* may benefit from phenotypic plasticity since it might boost adaptive response (Liu et al., 2016) which helps in reaching the elevation of 1900 m above the sea level (Bernal et al., 2015). Moreover, the species invasive traits, i.e., allelopathy activity reduces the growth of surrounding vegetation (Ghayal et al., 2010), thereby promoting invasion (Mooney and Cleland, 2001). Introduction of *S. nodiflora* to India is unknown but it had been reported from several state (Chauhan and Johnson, 2009; Haque et al., 2012; Usharani and Raju, 2018). Regardless of invasive nature, *S. nodiflora* seem to have medicinal values used as vegetable in different part of the world (Burkill, 1985; Rathi and Gopalkrishnan, 2005; Idu and Onyibe, 2007, Adjibode et al., 2015; Tun, 2020).

3.6.8 *Triumfetta rhomboidea* Jacq.

Triumfetta rhomboidea, a facultatively autogamous and perennial shrub (Raju and Rani, 2017), originated from tropical America (Reddy, 2008; Acevedo-Rodríguez and Strong, 2012) belongs to the family of Tiliaceae. *T. rhomboidea* is a competitive and invasive species (Wells et al., 1986), which is widely distributed in the tropical region of Africa, America, Asia and Australia (Bayer and Kubitzky, 2003; Iqbal et al., 2010). The species is listed as one of the major weeds in the Pacific and Asia region (Moody, 1989; Waterhouse, 1997), capable of adapting in variety of soil and environmental conditions (Iqbal et al., 2010). These species were reported to have invaded pastures and forest disturbed areas preventing the establishment of native vegetation (Wells et al., 1986) and develops hazard to human as well as livestock in forested region where these plants were abundant (Motooka, 2003). Due to its fruit having spiny hooked (Raju and Rani, 2017), *T. rhomboidea* can easily transported into new area up to 2700 m above the sea level (Valkenburg and Bunyapraphatsara, 2001; Whitehouse et al., 2001). Moreover, its eradication is very difficult because of its woody bases and robust rot systems (McCormack, 2007). Additionally, *T. rhomboidea* had become a major part of traditional medicine for treatment of various ailments such as tumours, gonorrhea, leprosy, diarrhea and intestinal ulcers (Singh et al., 2020; Kendre and Watke, 2023). Although it is uncertain how *T. rhomboidea* was introduced to India, reports from many states were available (Singh et al., 2020).

3.6.9 *Urena lobata* L.

Urena lobata is an aggressive noxious weed (Austin, 1999; Randall, 2012), originated from tropical Africa (Reddy, 2008; Akomolafe and Rosazlina, 2023) and a member of Malvaceae family. *U. lobata* can grow in variety of soil (Harris, 1981) and had become globally widespread in all tropical and subtropical regions (Ong, 2001; PIER, 2012). These species potentially modify and outcompete the native flora under suitable environmental condition but intolerant of shade (Francis, 2000; Martin and Chanthy, 2009; Florida Exotic Pest Plant Council, 2011). In addition, *U. lobata* seems very versatile which help in formation of monospecific stand (Langeland et al., 2008), due to its spiny hooked fruit that aid in invading to new areas (Austin, 1999; Francis, 2000). These species can be found growing in different types of habitats such as forest, wetland, roadsides, cropland, swampy areas as well as salt marshes (PIER, 2012; Sekar, 2012) and elevated to an altitude of 2000 m (UTPD, 2014). Despite its invasiveness, *U. lobata* is cultivated and traditional used in many tropical countries (Babu et al., 2016; Islam and Uddin, 2017). However, there were reports from many states (Sekar, 2012; Ray et al., 2019), despite the fact that the introduction of *U. lobata* to India is unclear till now.

All these species have been reported from various parts of Arunachal pradesh, but very limited studies were there on their invasion aspect. According to Singh et al., (2021) show unambiguously that IAPs were ascending at greater altitudes than their usual habitat. *Ageratum conyzoides* is one of the most dominant and invasive species under the canopy of *Dendrocalamus hamiltonii* in tropical forests of Arunachal pradesh because of their shade tolerant nature which allows it to become established and maintain dense populations within the understorey (Arunachalam and Arunachalam, 2002; Singh et al., 2002; Kaur et al., 2012). *Bidens pilosa* had been recorded from Arunachal pradesh by few researchers (and reported that this species may be spreading in the himalayan region (Kosaka et al., 2010; Singh et al., 2021). *Chromolaena odorata* is a dominant weed at lower altitudes and invaded the degraded forest and abandoned jhum fallows in the north eastern region (Rao, 1977). *Mikania micrantha* invades the new areas, suppressing the growth of the associated plant species and forming a dense cover over the host plant reducing their ability to receive light at lower altitudes in north eastern India (Tripathi et al., 2012). *Solanum torvum* is widely distributed in north eastern India (Deka et al., 2012) and is known to be an ethno-medicinal plant species in the eastern Himalayan region (Bushi et al., 2021). *Solanum viarum* often occupies the space of neighboring plants affecting the growth of native plants in the forested areas (Welman, 2003) and reported to have ethno-medicinal values (Bushi et al., 2021). *Synedrella nodiflora* is a noxious weed and generally found along roadsides, in open or partial shady and moist places in association with *Ageratum conyzoides, Bidens pilosa, Cynodon dactylon* and other herbaceous plants (Lal et al., 2009). *Triumfetta rhomboidea* was generally found in open places, along the roadsides and forest margin in the north eastern region and traditionally used to cure gonorrhea by tribal communities (Monlai, 2013; Balkrishna et al., 2021). Similarly, *Urena lobata* had also been reported from Arunachal pradesh and commonly found along the roadside and forested areas (Monlai, 2013). Moreover, this species is widely used for treating malaria and pneumonia by the tribal communities of this region (Wangpan et al., 2019).

3.7 Nativity

In order to ascertain the nativity of the species, the *Index Kewensis Plantarum Phanerogamarum* and its supplements (Anonymous, 1883–1885; 1886–1970) was utilized. This method recognizes the species only originated from the Himalayan region and Indian Orientals were considered as native; all other species were regarded as non-native or alien to the area. The World flora online (WFO) plant list (http://www.worldfloraonline.org/ and http://powo.science.kew.org/) and articles by Reddy (2008), Chevalier (1947), and Kuete (2017) were used to identify the nativity of the species in cases where the data was unavailable in *Index Kewensis Plantarum Phanerogamarum* and its supplements (Anonymous 1883–1885; 1886–1970).

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Chapter 4

Distribution and Invasion Patterns of Selected Invasive Alien Plant Species

4.0 Introduction

Forests occupy around 30 to 40% of the territory covered by vegetation on Earth (Waring and Schlesinger, 1985). Forests are predicted to inhabit most landscapes where annual precipitation exceeds around 25 cm, with the exception of polar areas where energy restrictions limit vegetation development (Neilson, 1995). Forests play a significant role in biospheric processes, but 60,000 km² of primary forest are lost or altered annually. Despite this, primary forests still account for more than one-third of all forests (Vitousek, 1994; FAO, 2006; Schmitt et al., 2009). Forest ecosystems supply a wide range of goods and services, including fuelwood, food, timber, and non-timber forest products (NTFP). The Millennium Ecological Assessment (MEA, 2005) categorizes these ecological functions as delivering, regulating, sustaining, and cultural. Forests may support high levels of biodiversity, with tropical forests standing out for their concentration of endemic species and species richness (Mittermeier et al., 1998; Brooks et al., 2006). Forest and tree cover make up nearly one-fourth of India's geographical area, with natural forest accounting for approximately 64% (FSI, 2021).

Mountain ecosystems account for about 25% of terrestrial biological diversity and cover 27% of the planet's physical area (Körner, 2000). However, mountain ecosystems are susceptible to the rapid pace of global development (Körner and Spehn, 2019) and must be conserved before annihilation. Because vegetation in mountain ecosystems belongs to the first trophic level, it must be well recorded and measured (Khan et al., 2012). When analyzing vegetation, records that include a broad variety of features of plant diversity are particularly useful (Kier et al., 2005). The value of recording plant variety in mountain ecosystems cannot be overemphasized, especially given recent changes in land use, infrastructure development, unsustainable tourism, habitat fragmentation, and climate change (EEA, 2002). However, several early explorers and scholars examined the forest vegetation of the eastern Himalayan area in depth (Hajra et al., 1996, 2008, 2009; Roy and Behera, 2005; Deb et al., 2008; Dutta et al., 2013; Saikia et al., 2017). Furthermore, the eastern Himalayan area is particularly rich in endemic and non-endemic species that must be protected from ever-increasing human activity (Saikia et al., 2017).

IAPs presented a special threat in this sensitive environment by altering the forest's species composition, resulting in a variety of negative changes (Fei et al., 2014). Policymakers and land managers have expressed worry about IAPs' mechanisms for invading their introduced areas or regions (Pathak et al., 2019). Control over them is critical, given their potential to inflict financial and environmental damage (Rai and Singh, 2020). The first and most important stage in developing a management strategy for an alien plant is determining its distribution within its newly extended range (Auld and Johnson, 2014). Studying diverse forest types might substantially improve our knowledge of IAP invasion patterns and management tactics (Pathak et al., 2019; Dhakal et al., 2024). Studies across various forest types may help us understand how plant communities operate and develop in response to environmental changes and establishment limitations (Verheyen et al., 2003; Verstraeten et al., 2013). Furthermore, Lazzaro et al. (2020) demonstrated that IAP invasiveness influences forest patterns and community organization. As a result, it is critical to assess the invasiveness of alien plant species in different forest types, as well as their distribution patterns.

There is a dearth of research on plant invasion patterns in various forest types from the IHR (Kosaka et al., 2010; Mandal and Joshi, 2014a; Pathak et al., 2019; Mehra et al., 2023), despite extensive research on the subject worldwide (Ohsawa, 1986; Pysek et al., 2002; Vilà et al., 2007; Fei et al., 2008; Hernández et al., 2014; Dyderski and Jagodzinski, 2021). Even though the east Himalayan region had seen relatively little invasion research, only Kosaka et al. (2010) have studied the distribution along elevation gradients. As a result, there is no empirical dataset or proof of any dominant IAPs study, and the eastern Himalayan area is totally devoid of biological invasion research. Notwithstanding notable progress in the cataloguing of alien plants, much remains unclear regarding the degree of invasion in various habitats and the habitat preferences of specific alien species in specific regions (Chytrý et al., 2005). Because of this, it is necessary to do a thorough study on biological intrusions and assess their impact on the local ecology. Considering the aforementioned, the current study would be a major attempt to assess the invasiveness and distribution patterns of IAPs in different kinds of forests in the Himalayan region. Therefore, the results of the research should further our understanding of the impacts of plant invasion and provide recommendations for possible management approaches for both the IHR as a whole and the region in particular.

4.1 Methodology

The data collected from the various PTR forest types forms the basis of the methodology, which is explained below:

4.1.1 Distribution and Invasion Assessment

The study, which aimed to conduct an extensive investigation, was carried out in five distinct forest types: the Assam Alluvial Plain Semi Evergreen Forest, the East Himalayan Mixed Coniferous Forest, the East Himalayan Subtropical Wet Hill Forest, the Non-Forest, and the Riverine Forest of PTR (Champion and Seth, 1968; FSI, 2009). According to Bapu and Nimasow (2017), the range's forest cover had seen notable changes, which led to the selection of different kinds of forests. Using ArcGIS 10.6.1 in the research region, a 5 × 5 km spatial grid was created in the GIS domain, to which all sample data were spatially associated (Fig. 4.1). Mungi et al. (2021) have determined that the 25 km^2 spatial scale is suitable for comprehending the evaluation of IAPs within the research region. In each grid, a number of plots ranging from three to fifteen were spread out to check for invasion and distribution. The five forest types were identified using the GIS domain. In order to document the shifting vegetation, the plots were arranged at random within the vicinity of the accessible and covered IAPs, maintaining a minimum distance of 400 m. Two hundred and eight sampling plots of 20m x 20 m were placed up for data collection in these five forest types; the samples included 65 in AAPSEF, 84 in EHMCF, 27 in EHSTWH, 16 in NF, and 16 in RF, respectively (Table 4.1).

Individual-based rarefaction and extrapolation curves were used as a unified method to assess the species richness for each kind of forest. In order to ensure sampling completeness, the nonparametric estimator values from chao estimation (Chao et al., 2020), individual rarefaction values from PAST Software version 5.1 and observed values from the sample plots were compared with the maximum richness in each forest type. The richness calculated from the sampled based rarefaction curves was used to evaluate the richness of various forest types with distinct effort. The species accumulation curve is presented to measure the sampling efficacy for the full research region. The diversity of IAPs and their relative contribution to the overall species richness, or the percentage of the entire community cover that they comprise, were measured to determine the extent of invasion (Hierro et al., 2005). In order to understand the level of invasiveness in each forest types, the cover percentage of each chosen species in the sample plots was measured to gauge the invasiveness of various forest types in order to track the invasion of the chosen IAPs across them. The mean cover had been used to extrapolate to the hectare level, indicating the invasiveness of certain IAPs in various forest types. Moreover, the study more focused on the relative densities and relative abundance, instead of assuming they are same, following Curtis and McIntosh (1950). As relative density gives the proportional of individuals of a particular species relative to the total number of the individuals of all species communities whereas relative abundance provides a sense of evenness of the community and often uses the number of individuals.

4.1.2 Vegetation Sampling

The present study was conducted in five different forest types namely, Assam Alluvial Plain Semi Evergreen Forest, East Himalayan Mixed Coniferous Forest, East Himalayan Subtropical Wet Hill Forest, Non-Forest, and Riverine Forest of Pakke Tiger Reserve (Champion and Seth, 1968; FSI, 2009) To determine the richness and distribution of Invasive Alien Plant species, a survey of the flora inside the core region is conducted using the following techniques:

a) *Reconnaissance survey*: Prior to the actual vegetation sample, a reconnaissance survey was carried out at the intended research site to see if the chosen IAPs were present in the area. To get a better understanding of the core region, a total of 36 distinct locations around the study site and within the PTR's five forest types were studied. The survey locations were located in steep terrain, lower plains, riverine habitats, meadows, and accessible by trails and motorized highways. The survey was done from 7 November, 2019 to 25 December, 2019.

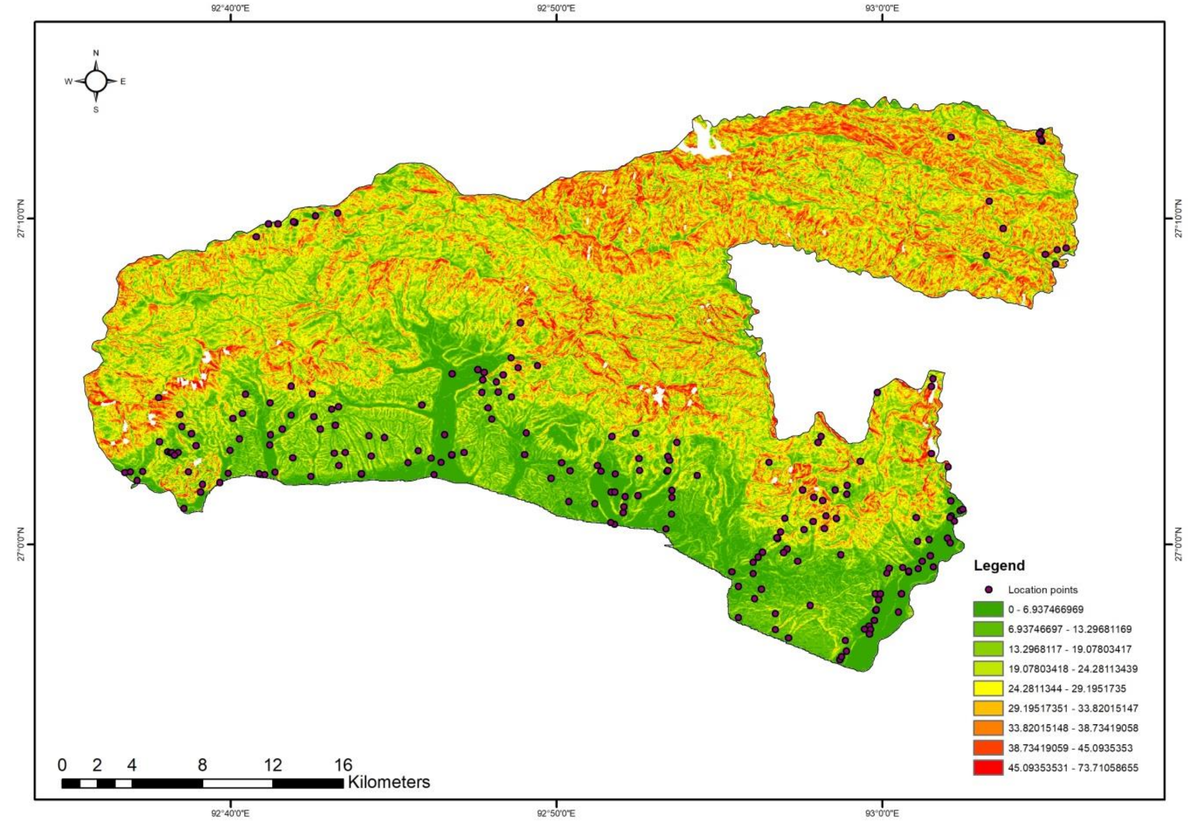


Figure 4.1: Map of the study area with survey location points across different forest types.

b) *Vegetation sampling design*: Between September 2021 and February 2023, sampling was carried out in all five forest types within the study area in order to determine the overall diversity spectrum and to ensure that all plant diversity. For the phytosociological survey in the research region, 3–15 plots with a 500 m gap between each plot were put up at each sample location (grid), contingent on space availability and accessibility. For the purpose of gathering data, a random sampling procedure was carried out at each of the five distinct forest types' research plots. The nested quadrat design was used to quantify the flora (Misra, 1968; Peet et al., 1998; Rai, 2006). Traditional sampling plots were 20×20m for the trees, and two 5×5m plots for the shrubs and saplings were placed at the diagonal corners. To measure the greatest involvement of many species in the survey's interpretative quality, five 1×1m plots for herbs and seedlings were placed at each of the four corners and one in the middle of the 20×20m plot.

Table 4.1: Details of sampling efforts across five forest types of the Pakke Tiger Reserve.

Forest types	**Area (km²)**	**Plots (400m²)**	**Total individuals**	**No. of trees**	**No. of shrubs**	**No. of herbs**
AAPSEF	390.69	65	137	59	23	55
EHMCF	364.51	84	133	67	21	45
EHSTWH	79.81	27	81	36	15	30
NF	17.49	16	77	33	16	28
RF	9.44	16	80	37	15	28

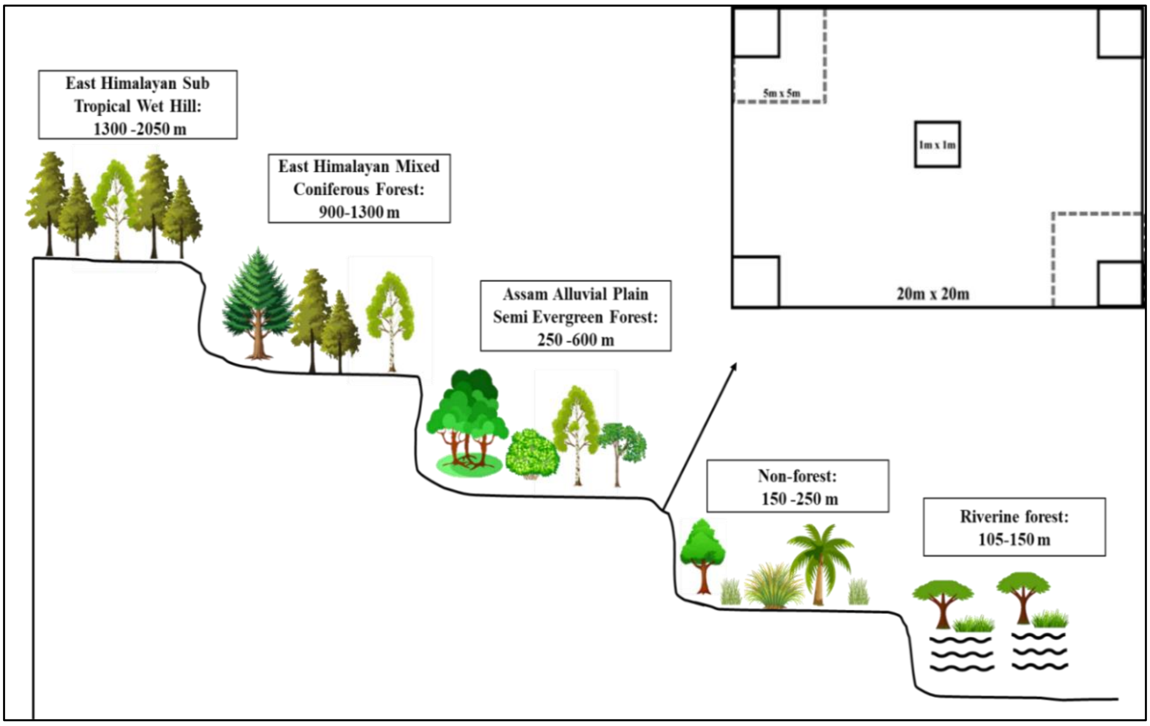


Figure 4.2: Diagrammatic representation of the research design for evaluating the vegetation in the studied area.

According to Dash et al. (2021), circumference was used to distinguish between three tree life stages: mature trees (C > 31.5 cm at 1.37 m above ground level), saplings (C = 10.5–31.4 cm), and seedlings (C < 10.4 cm). The total cover of the species within each category was computed for each vegetation plot, and the number of species within each of these categories was recorded. Moreover, the levels of invasion in all IAPs in each forest types are expressed as the proportion of IAPs to total number of species per site (Proportion of

IAPs =Number of IAPs/Total number of species x 100). Figure 4.2 offers the schematic sample plan for the phytosociological investigation. For each forest type, a separate record of the floral composition of the specific sample site is kept for the epiphytes, lianas, and climbers along their host.

c) *Sampling strategy*: One of the most important steps in conducting ecological research is choosing an appropriate sampling method that will ensure the collection of trustworthy data for statistical analysis and ecological inference (Swacha et al., 2017). The field data for the current investigations were gathered using the stratified random sampling (SRS) approach (Hansen et al., 1953). According to Roleček et al. (2007), the SRS helps acquire variable data sets with appropriate representations of plots created in unusual vegetation or forest types while avoiding the source of bias. Furthermore, Goedickemeier and Kienast (1997) found that it accurately depicts the vegetation pattern with modest sampling effort.

Given that the existing quadrats usually have a square frame, there is a strong possibility that they will sample a species inside them. Comparable samples from areas with consistent size and shape may be gathered by using a quadrat (Peet et al., 1998; Sharma, 2005). Sampling was conducted in the PTR's hilly and mountainous terrain across a variety of forest types at varying elevations and slopes. The kind of survey determines the quadrat size, which is often established experimentally in the field (Misra, 1968). A standard Eastern Himalayan quadrate size was used for this research investigation (Das and Lahiri, 1997; Rai, 2006). The tools and equipment utilized in the field research include a GPS (Garmin eTrex 10), measuring tape (100 m), tailor's tape (1.5 m), field notebook, Ziplock polythene bags, a nylon rope, a white coloured flag, colour tags, pens, pencils, sharpeners, erasers and data sheets.

d) *Specimen identification*: A thorough floristic study was carried out on both motorized roads and the accessible paths in the forested geographic areas. Every item was captured on camera, and detailed descriptions along with its local names, GPS coordinates, uses, habitat, date, and survey day were entered into the field notebook. While many of the species were recognized in the field, those that could not be identified were collected and examined at the herbariums of the Botanical Survey of India (Itanagar-Regional Centre) and the Wildlife Institute of India (Dehradun). Furthermore, the plant specimens were identified following the references of Bor (1940), Hajra et al. (1996, 2008, 2009), Reddy (2008), and Kanjilal (1934, 1936, 1938, 1939, 1940).

e) For the nomenclature and the family delimitation for each species <https://wfoplantlist.org/, https://www.ipni.org/and https://powo.science.kew.org/> were consulted. The collected specimens and photographs were deposited in the herbarium of Wildlife Institute of India (Dehradun).

4.1.3 Data Analysis

Various species diversity indices were computed using Paleontological Statistics (PAST) software, version 5.1 (Hammer et al., 2001) and R software, version 4.3.3. The density data were reported on a per hectare basis after being averaged across the plots. The total number of species for each kind of living form in the sampled region was used to calculate species richness. The created datasets were examined to determine the extent of an alien species each of the five distinct forest types contributed. The methods for determining each species' important value index, basal area, density, abundance and frequency in the community were analyzed by following Curtis (1959), Misra (1968), Mueller-Dombois and Ellenberg (1974), Sharma (2005); Basistha et. al. (2010). The species diversity in a given community was determined using a variety of diversity indices, including those developed by Shannon and Wiener (1963). Whitford (1949) proposed the abundance to frequency (A/F) ratio as a measure of contagiousness, and it was used to compute the distribution pattern of plant species. According to Curtis and Cotton (1956), the ratio indicates regular (0.025), random (0.025-0.05), and infectious (>0.05) dispersion. For the analysis of invasiveness in sampling plots in each five forest types were done by following Chytrý et al. (2005), Adhikari and Babu (2008), Medvecka et al. (2018) and Chaudhary et al. (2019). Jaccard similarity coefficient was employed to study the association of invasive plant species between forest types following Mao et al. (2023). The level of invasion in each different forest types are analyzed by following Dar et al. (2018). Moreover, sampling completeness was obtained using the iNEXT package in Software R 4.3.3 by following Gotelli and Colwell (2011) and Chao et al. (2020). The kernel density estimation method of species abundance was used to simulate the distribution pattern of the selected species in the study area following Chengzhen et al. (2000). All the related formulas used in the analysis were given in Box 1.

$$\text{Density} = \frac{\text{Numbers of individuals of a species in all quadrats}}{\text{Number of quadrats studied}}$$

$$\text{Relative density (RD)} = \frac{\text{Density of a species} \ \text{X} \ 100}{\text{Density of all species}}$$

$$\text{Frequency} = \frac{\text{Number of quadrate of occurrence of a species} \ \text{X} \ 100}{\text{Number of quadrats studied}}$$

$$\text{Relative frequency (RF)} = \frac{\text{Frequency of a species} \ \text{X} \ 100}{\text{Frequency of all species}}$$

$$\text{Abundance} = \frac{\text{Number of individuals of a species}}{\text{Number of quadrat of occurrence of a species}}$$

$$\text{Relative abundance} = \frac{\text{Abundance of a species} \ \text{X} \ 100}{\text{Abundance of all species}}$$

$$\text{Abundance/ frequency (A/F) ratio} = \frac{\text{Abundance of a species}}{\text{Frequency of a species}}$$

$$\text{Mean Basal area (MBA)} = \frac{C^2}{(4 \ \text{X} \ 3.14)}$$

$$\text{Total basal area (TBA)} = \text{MBA of a species X Density of the species}$$

$$\text{Relative dominance (RDo)} = \frac{\text{TBA of a species} \ \text{X} \ 100}{\text{TBA of all species}}$$

$$\text{Important value index (IVI)} = \text{RF} + \text{RD} + \text{RDo}$$

$$\text{Shannon Diversity index (H)} = -\sum[(pi) \ \text{X} \ 1n \ (pi)]$$

*Where pi= ni/N (individuals of a given species, N= total number of individuals in a community), 1n= natural logarithm, C= mean circumference and $\sum$ = summation

Box 1. Formulae's for calculating different vegetation parameters

4.2. Results

The findings are broken down into three sections: (1) Species richness and sampling accuracy of Pakke Tiger Reserve; (2) the species diversity of IAPs in Pakke Tiger Reserve; and (3) Distribution and invasion patterns of selected IAPs in Pakke Tiger Reserve.

4.2.1 Species Richness and Sampling Accuracy of Pakke Tiger Reserve

The study identified a total of 151 species, consisting of 70 tree species belonging to 35 families with 54 genera, 24 shrub species under 20 genera belonging to 13 families, 48 herb species belongs to 46 genera and 21 families, 4 climber species climbers under 3 genera and 4 families, 4 grass species belong to one family under 3 genera and one sedge species under one genus with one family. The present study report 109 perennial species and 48 annual species belonging to 57 different families and 124 different genera. Moreover, Pakke Tiger Reserve is represented by 51 different geographical locations around the world. This information may be seen in Table 4.2. The family Asteraceae had the highest number of species (18), making it the most dominant family. It was followed by Malvaceae (11), Fabaceae (10), Acanthaceae (8), Lamiaceae and Meliaceae (7 each), Solanaceae (5) and Poaceae (5; Fig. 4.4a). Among the total number of species, 63 species were native to the Himalayan area and Indian Oriental (Indo-Himalaya) regions. Out of these, there were 46 tree species, 9 shrub species, 4 herb species, 3 climber species and 1 grass species. On the other hand, there were 88 non-native species in these regions. Among the non-natives, there were 24 tree species, 15 shrub species, 44 herb species, 3 grass species, 1 climber species and 1 sedge species. The dominant species among the native plants were Acanthaceae and Malvaceae, each with five species. The two families with four species each, Lamiaceae and Meliaceae, came next. Elaeocarpaceae, Euphorbiaceae, Fagaceae, Myristicaceae, Mrytaceae, Rubiaceae, Arecaceae, Burseraceae, Combretaceae, Elaeocarpaceae and Melastomataceae each contained two species. Only one species each was present in the remaining families (Fig. 4.4b). In the case of non-native species, the Asteraceae family had the highest number of species (18), followed by Fabaceae (7), Malvaceae and Solanaceae (6 each), Poaceae (4), Acanthaceae, Lamiaceae and Meliaceae (3 each), Elaeocarpaceae, Moraceae, Myrtaceae and Rubiaceae (2 each) and the remaining families with only one species each (Fig. 4.4c).

Total species richness across the different forest types ranged between 76 - 137 and was highest (137) belonging to 54 families and 112 genera in AAPSEF and lowest (77), under 67 genera and 40 families in NF, respectively. Tree species richness ranged from 32 to 67 and was recorded highest (67) in EHMCF followed by AAPSEF (59; Table 4.13). With tree species belonging to the Lauraceae and Fagaceae families, specifically *Castanopsis tribuloides, Castanopsis indica* and *Mesua ferrea*, dominated the upper region, while *Pterospermum acerifolium, Sterculia villosa, Monoon simiarum, Duabanga grandiflora* and *Stereospermum chelonoides* dominate the lower plain and foothills. The bulk of the trees in NF, which had the lowest recorded values (32) were *Vitex glabrata, Dillenia indica* and *Duabanga grandiflora*. The range of shrub species richness was 15 to 23, with AAPSEF having the greatest score (23) and EHSTWH having the lowest (15). The largest herb species richness (28–58) was identified in AAPSEF, whereas the lowest (28 each) was observed in NF and RF.

AAPSEF had the maximum tree density (600 ind/ha), whereas NF had the lowest (300 ind/ha). The results showed that the sapling densities were highest in EHMCF (1100 ind/ha) and lowest in RF (575 ind/ha). EHMCF reported the highest seedling density (8800 ind/ha), whereas AAPSEF recorded the lowest seedling density (3625 ind/ha). In terms of shrub density, AAPSEF had the highest (2600 ind/ha), while EHMCF had the lowest (900 ind/ha). It was discovered that the herb density was lowest in EHMCF (16000 ind/ha) and greatest in RF (60000 ind/ha).

Pterospermum acerifolium, Sterculia villosa, Monoon simiarum, Duabanga grandiflora and *Stereospermum chelonoides* predominate in the lower plain and foothills, while Lauraceae and Fagaceae tree species *Castanopsis tribuloides, Castanopsis indica* and *Mesua ferrea* - dominate in the upper region. Additionally, the main emergent species in this distinctive layered forest structure of PTR are *Tetrameles nudiflora* and *Ailanthus integrifolia*, both of which are indigenous to the Himalayan region. Native plant and shrub species that cover the ground layers of the examination area include *Clerodendrum colebrookeanum, Melastoma malabathricum, Phlogacanthus curviflorus, Strobilanthes cusia, Osbeckia nutans, Hellenia speciosa, Lepidagathis incurva, Phanera vahlii,* and *Poikilospermum suaveolens*. These varieties contribute to the region's rich biodiversity.

Table 4.2. List of the species recorded from the study area.

Species	**Family**	**Habit**	**Nativity***
Climber			
Convolvulus arvensis L.	Convolvulaceae	Perennial	Geront. temp
Piper griffithii C.DC.	Piperaceae	Perennial	Reg. Himal
Thunbergia coccinea Wall. ex D.Don	Acanthaceae	Perennial	Reg. Himal.; Burma
Thunbergia grandiflora Roxb.	Acanthaceae	Perennial	Ind. or.; Burma
Grass			
Eleusine indica (L.) Gaertn.	Poaceae	Annual	Cosmop. trop. et subtrop
Saccharum officinarum L.	Poaceae	Perennial	N Guin
Saccharum spontaneum L.	Poaceae	Perennial	West As. trop
Thysanolaena latifolia (Roxb. ex Hornem.) Honda	Poaceae	Perennial	Ind.or
Herb			
Adenostemma lavenia (L.) Kuntze	Asteraceae	Perennial	As. trop et subtrop
Mitracarpus hirtus (L.) DC.	Rubiaceae	Annual	Am. trop; Mexic
Achyranthes aspera L.	Amaranthaceae	Annual	Geront. trop
Acmella paniculata (Wall. ex DC.) R.K.Jansen	Asteraceae	Annual	Java
Acmella radicans (Jacq.) R.K.Jansen	Asteraceae	Annual	S Am. trop

Ageratina adenophora (Spreng.) R.M.King & H.Rob.	Asteraceae	Perennial	Mexic
Ageratum conyzoides L.	Asteraceae	Annual	Am. trop
Alternanthera sessilis (L.) DC.	Amaranthaceae	Perennial	Reg. trop
Bidens pilosa L.	Asteraceae	Annual	Am. trop
Blumea lacera (Burm.f.) DC.	Asteraceae	Annual	Am. trop
Cardamine hirsuta L.	Brassicaceae	Annual	Am. trop
Hellenia speciosa (J.Koenig) S.R.Dutta	Costaceae	Perennial	Ind. or.; Malaya
Cleome rutidosperma DC.	Cleomaceae	Annual	Am. trop
Corchorus aestuans L.	Malvaceae	Annual	Am. trop
Crassocephalum crepidioides (Benth.) S.Moore	Asteraceae	Annual	Am. trop
Crotalaria pallida Aiton	Fabaceae	Annual	Am. trop
Cuphea carthagenensis (Jacq.) J.F.Macbr.	Lythraceae	Annual	Braz
Eclipta prostrata (L.) L.	Asteraceae	Annual	Am. trop
Floscopa scandens Lour.	Commelinaceae	Annual	As. et Austral. trop
Galinsoga parviflora Cav.	Asteraceae	Annual	Am. trop
Impatiens balsamina L.	Balsaminaceae	Annual	Am. trop
Imperata cylindrica (L.) Raeusch.	Poaceae	Perennial	Am. trop
Lepidagathis incurva Buch.-Ham. ex D.Don	Acanthaceae	Perennial	Ind. or
Ludwigia octovalvis (Jacq.) P.H.Raven	Onagraceae	Annual	Afr. trop

Mikania micrantha Kunth	Asteraceae	Perennial	Am. trop
Malvastrum coromandelianum (L.) Garcke	Malvaceae	Annual	Am. trop
Mesosphaerum suaveolens (L.) Kuntze	Lamiaceae	Annual	Am. trop
Mimosa pudica L.	Fabaceae	Annual	Braz
Nicotiana plumbaginifloia Viv.	Solanaceae	Annual	Am. trop
Oxalis corniculata L.	Oxalidaceae	Annual	Europ
Parthenium hysterophorus L.	Asteraceae	Annual	N Am. trop
Phanera vahlii (Wight & Arn.) Benth.	Fabaceae	Perennial	Ind. or
Phaulopsis dorsiflora (Retz.) Santapau	Acanthaceae	Perennial	Madag
Physalis angulata L.	Solanaceae	Annual	Am. trop
Poikilospermum suaveolens (Blume) Merr.	Urticaceae	Perennial	Ind. or.; Malaya
Scoparia dulcis L.	Plantaginaceae	Annual	Am. trop
Senna tora (L.) Roxb.	Fabaceae	Annual	S Am. trop
Sida acuta Burm.f.	Malvaceae	Annual	Am. trop
Solanum nigrum L.	Solanaceae	Annual	Eurasia
Solanum viarum Dunal	Solanaceae	Perennial	Am. trop
Sonchus oleraceus L.	Asteraceae	Annual	Meditter
Spermacoce ocymoides Burm.f.	Rubiaceae	Annual	Cosmop. trop
Stachytarpheta indica (L.) Vahl	Verbenanceae	Annual	Am. trop

Stephania japonica (Thunb.) Miers	Menispermaceae	Perennial	Japan.
Synedrella nodiflora (L.) Gaertn.	Asteraceae	Annual	West indies
Tridax procumbens L.	Asteraceae	Perennial	Centr Am. trop
Xanthium strumarium L.	Asteraceae	Annual	Am. trop
Youngia japonica (L.) DC.	Asteraceae	Annual	S Am. trop
Sedge			
Cyperus iria L.	Cyperaceae	Annual	Am. trop
Shrub			
Alpinia nigra (Gaertn.) Burtt	Zingiberaceae	Perennial	Indian Subcontinent to China
Buddleja asiatica Lour.	Scrophulariaceae	Perennial	As. trop.; Malaya
Chromolaena odorata (L.) R.M.King & H.Rob.	Asteraceae	Perennial	Am. trop
Citrus medica L.	Rutaceae	Perennial	Asia, Mediterr, Assyria, Persia
Clerodendrum colebrookeanum Walp.	Lamiaceae	Perennial	Ind. or.; Malaya
Clerodendrum infortunatum L.	Lamiaceae	Perennial	Ind. or.; Arch. Mal
Coffea benghalensis B.Heyne ex Roth	Rubiaceae	Perennial	Ind. or. (Bengal)
Datura metel L.	Solanaceae	Annual	Am. trop
Dendrocnide sinuata (Blume) Chew	Urticaceae	Perennial	China
Holmskioldia sanguinea Retz.	Lamiaceae	Perennial	Ind. or
Lantana camara L.	Verbenaceae	Perennial	Am. trop

Melastoma malabathricum L.	Melastomataceae		Ind. or.; Malaya
Ocimum tenuiflorum L.	Lamiaceae	Perennial	Geront. trop
Osbeckia nutans Wall. ex C.B.Clarke	Melastomataceae	Perennial	Reg. Himal
Oxyspora paniculata (D.Don) DC.	Melastomataceae	Perennial	Reg. Himal
Phlogacanthus curviflorus (Nees) Nees	Acanthaceae	Perennial	Reg. Himal
Phlogacanthus thyrsiformis (Roxb. ex Hardw.) Mabb.	Acanthaceae	Perennial	India.
Senna alata (L.) Roxb.	Fabaceae	Annual	West indies
Senna occidentalis (L.) Link	Fabaceae	Annual	S Am. trop
Solanum torvum Sw.	Solanaceae	Perennial	West indies
Strobilanthes cusia (Nees) Kuntze	Acanthaceae	Annual	Ind. or.; Burma; China
Strobilanthes paniculata (Nees) Miq.	Acanthaceae	Perennial	Indo-China (Laos)
Triumfetta rhomboidea Jacq.	Malvaceae	Perennial	Am. trop
Urena lobata L.	Malvaceae	Annual	Afr. trop
Tree			
Aesculus assamica Griff.	Sapindaceae	Perennial	Reg. Himal
Aglaia lawii (Wight) C.J.Saldanha	Meliaceae	Perennial	Ind. or
Aglaia spectabilis (Miq.) S.S.Jain & S.Bennet	Meliaceae	Perennial	Burma
Ailanthus integrifolia Lam.	Simaroubaceae	Perennial	Ind. or
Albizia lebbeck (L.) Benth.	Fabaceae	Perennial	Geront. trop

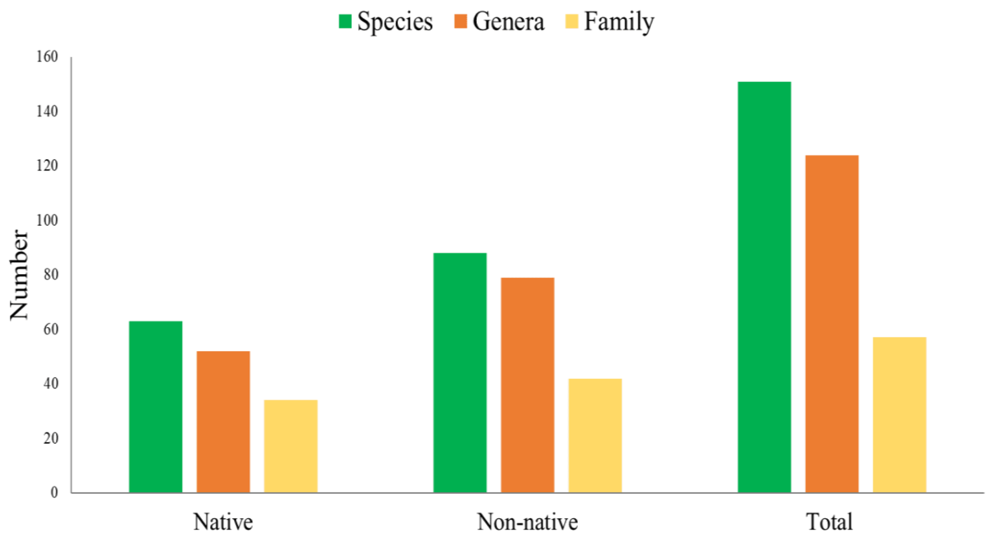
Albizia procera (Roxb.) Benth.	Fabaceae	Perennial	As. trop.; Austral
Liquidambar excelsa (Noronha) Oken	Altingiaceae	Perennial	Malaya
Aphanamixis polystachya (Wall.) R.Parker	Meliaceae	Perennial	Ind. or. (Assam)
Aquilaria malaccensis Lam.	Thymelaeaceae	Perennial	Malaya
Artocarpus chama Buch.-Ham.	Moraceae	Perennial	India.
Azadirachta indica A.Juss.	Meliaceae	Perennial	Java
Baccaurea ramiflora Lour.	Phyllanthaceae	Perennial	Cochinch
Bauhinia purpurea L.	Fabaceae	Perennial	Ind. or.; Burma; China
Bauhinia variegata L.	Fabaceae	Perennial	Ind. or.; Burma; China
Bombax ceiba L.	Malvaceae	Perennial	Am. austr
Canarium resiniferum Bruce ex King	Burseraceae	Perennial	Reg. Himal
Canarium strictum Roxb.	Burseraceae	Perennial	Ind. or
Castanopsis tribuloides (Sm.) A.DC.	Fagaceae	Perennial	Reg. Himal.; Burma
Castanopsis indica (Roxb. ex Lindl.) A.DC.	Fagaceae	Perennial	Reg. Himal
Chisocheton cumingianus (C.DC.) Harms	Meliaceae	Perennial	Ins. Philipp
Choerospondias axillaris (Roxb.) B.L.Burtt & A.W.Hill	Anacardiaceae	Perennial	China
Chukrasia tabularis A.Juss.	Meliaceae	Perennial	Ind. or
Cinnamomum bejolghota (Buch.-Ham.) Sweet	Lauraceae	Perennial	Reg. Himal
Cinnamomum glaucescens (Nees) Hand.-Mazz.	Lauraceae	Perennial	Reg. Himal

Dillenia indica L.	Dilleniaceae	Perennial	As. trop
Duabanga grandiflora (Roxb. ex DC.) Walp.	Lythraceae	Perennial	Ind. or
Dysoxylum gotadhora (Buch.-Ham.) Mabb.	Meliaceae	Perennial	Ind. or
Elaeocarpus aristatus Roxb.	Elaeocarpaceae	Perennial	Ind. or
Elaeocarpus floribundus Blume	Elaeocarpaceae	Perennial	As. trop
Elaeocarpus sphaericus (Gaertn.) Heer	Elaeocarpaceae	Perennial	Ins. Philipp.
Elaeocarpus varunua Buch.-Ham. ex Mast.	Elaeocarpaceae	Perennial	Ind. or.; Malaya
Endospermum chinense Benth.	Euphorbiaceae	Perennial	China; Malaya
Ficus elastica Roxb. ex Hornem.	Moraceae	Perennial	N. Guin
Gmelina arborea Roxb. ex Sm.	Lamiaceae	Perennial	Ind. or.; Malaya
Gynocardia odorata R.Br.	Achariaceae	Perennial	Ind. or
Horsfieldia kingii (Hook.f.) Warb.	Myristicaceae	Perennial	Reg. Himal
Knema angustifolia (Roxb.) Warb.	Myristicaceae	Perennial	Reg. Himal.; Malaya
Kydia calycina Roxb.	Malvaceae	Perennial	Ind. or.; Burma
Livistona jenkinsiana Griff.	Arecaceae	Perennial	Reg. Himal
Macaranga denticulata (Blume) Müll.Arg.	Euphorbiaceae	Perennial	Reg. Himal.; Malaya
Macaranga peltata (Roxb.) Müll.Arg.	Euphorbiaceae	Perennial	Ind. or
Magnolia champaca (L.) Baill. ex Pierre	Magnoliaceae	Perennial	Malaya
Magnolia hodgsonii (Hook.f. & Thomson) H.Keng	Magnoliaceae	Perennial	Reg. Himal

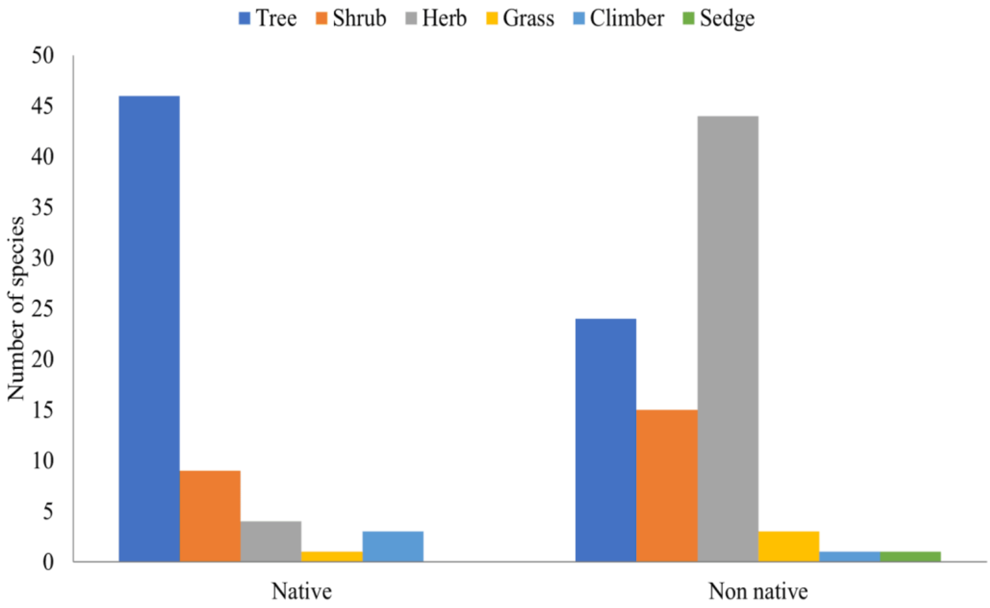
Mesua ferrea L.	Calophyllaceae	Perennial	Ind. or
Moringa oleifera Lam.	Moringaceae	Perennial	Ind. bor. occ.
Morus macroura Miq.	Moraceae	Perennial	Ind. or
Musa cheesmanii N.W.Simmonds	Musaceae	Perennial	Ind. or. (Assam)
Neolamarckia cadamba (Roxb.) Bosser	Rubiaceae	Perennial	Ind. or. (Bengal)
Oroxylum indicum (L.) Kurz	Bignoniaceae	Perennial	As. trop
Phoebe lanceolata (Nees) Nees	Lauraceae	Perennial	Ind. or.; Burma
Phoenix rupicola T.Anderson	Arecaceae	Perennial	Reg. Himal
Monoon simiarum (Buch.-Ham. ex Hook.f. & Thomson) B.Xue & R.M.K.Saunders	Annonaceae	Perennial	Reg. Himal.; Burma
Pterospermum acerifolium (L.) Willd.	Malvaceae	Perennial	Ind. or.; Java
Pterospermum lanceifolium Roxb. ex DC.	Malvaceae	Perennial	Ind. or
Pterygota alata (Roxb.) R.Br.	Malvaceae	Perennial	Ind. or
Schima wallichii (DC.) Korth.	Theaceae	Perennial	Reg. Himal.; Malaya
Sterculia villosa Roxb. ex Sm.	Malvaceae	Perennial	Ind. or
Stereospermum chelonoides (L.f.) DC.	Bignoniaceae	Perennial	Ind. or.; Burma
Syzygium cumini (L.) Skeels	Myrtaceae	Perennial	Ind. or
Syzygium jambos (L.) Alston	Myrtaceae	Perennial	As. trop
Syzygium zeylanicum (L.) DC.	Myrtaceae	Perennial	Ind. or.; Malaya
Syzygium syzygioides (Miq.) Merr. & L.M.Perry	Myrtaceae	Perennial	Java

Terminalia arjuna (Roxb. ex DC.) Wight & Arn.	Combretaceae	Perennial	Ind. or
Terminalia chebula Retz.	Combretaceae	Perennial	As. trop
Terminalia myriocarpa Van Heurck & Müll.Arg.	Combretaceae	Perennial	Reg. Himal.; Burma
Tetrameles nudiflora R.Br.	Tetramelaceae	Perennial	Ind. or.; Malaya
Vatica lanceifolia (Roxb.) Blume	Dipterocarpaceae	Perennial	Reg. Himal.; Burma
Vitex glabrata R.Br.	Lamiaceae	Perennial	Ins. Philipp.
Zanthoxylum rhetsa (Roxb.) DC.	Rutaceae	Perennial	Ind. or
Ziziphus mauritiana Lam.	Rhamnaceae	Perennial	Afr. trop et India

Note*: Africa (Afr.), America (Am.), Asia (As), Australia (Austr), Brazil (Braz), Boreal (Bor), Central (Centr), Cosmopolitan (Cosmop), Europe (Europ), Gerontia (Geront), Himalayan Region (Reg. Himal), Indian Oriental (Ind Or), Insular (Ins), Indonesia (Java), Malay Archipelago (Arch. Mal), Madagascar (Madag), Mediterranean (Mediterr), Mexico (Mexic), New guinea (N Guin), Occidentalis (Occ), Philippines (Phillipp), Vietnam (Cochinch), Subtropical (Subtrop), North (N), South (S), Temperate (Temp), Tropical (Trop); et =and.

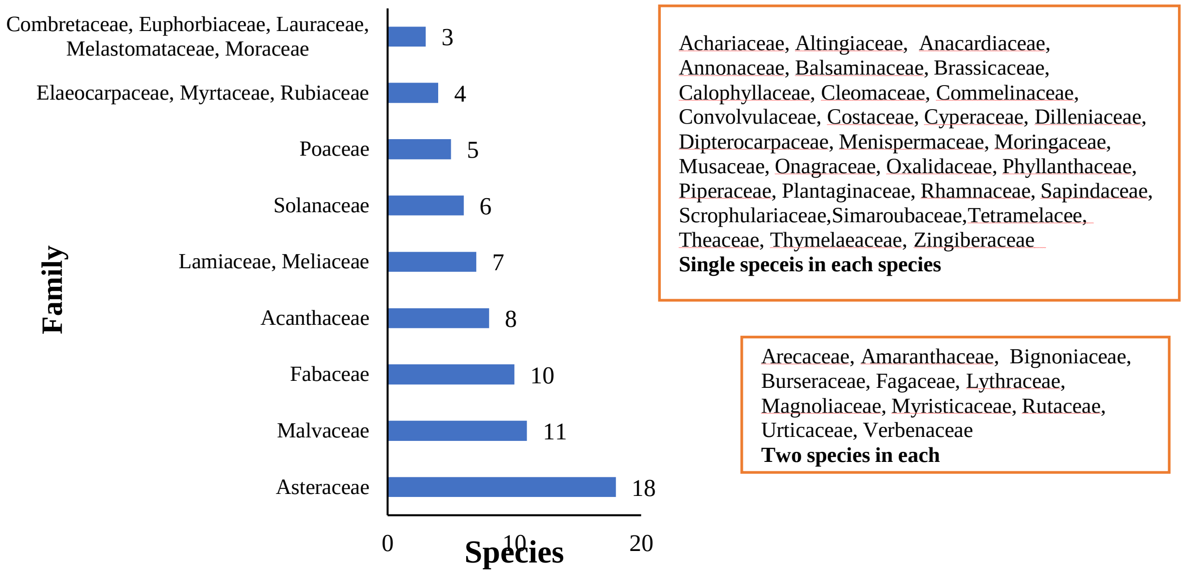


(a)

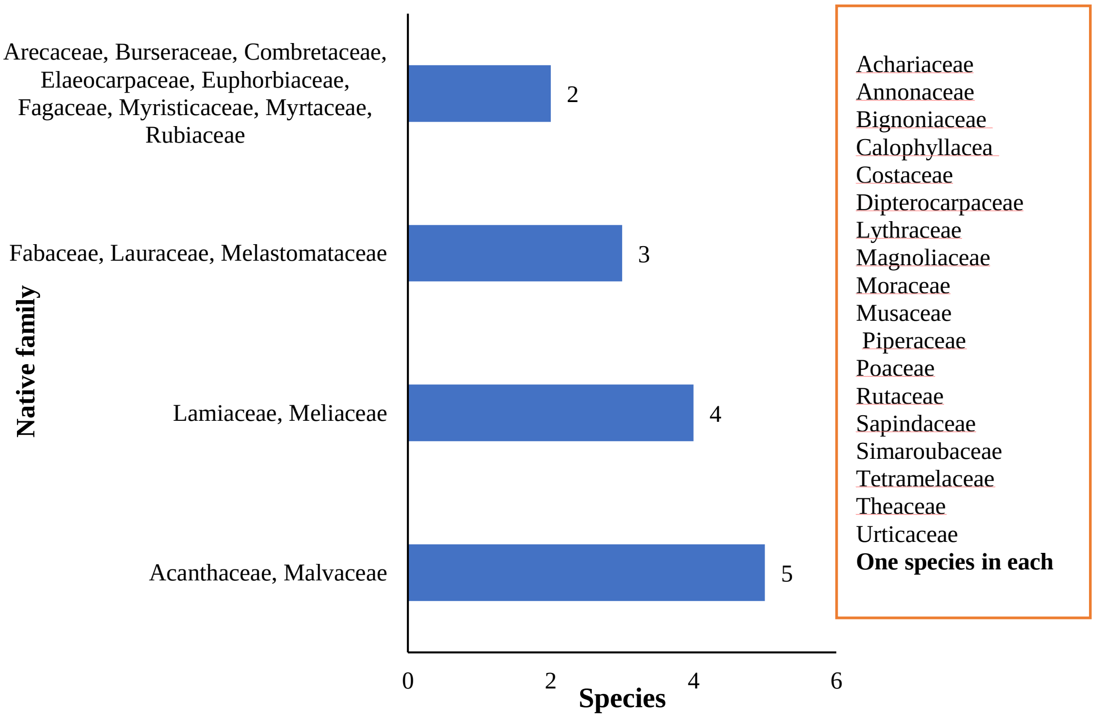


(b)

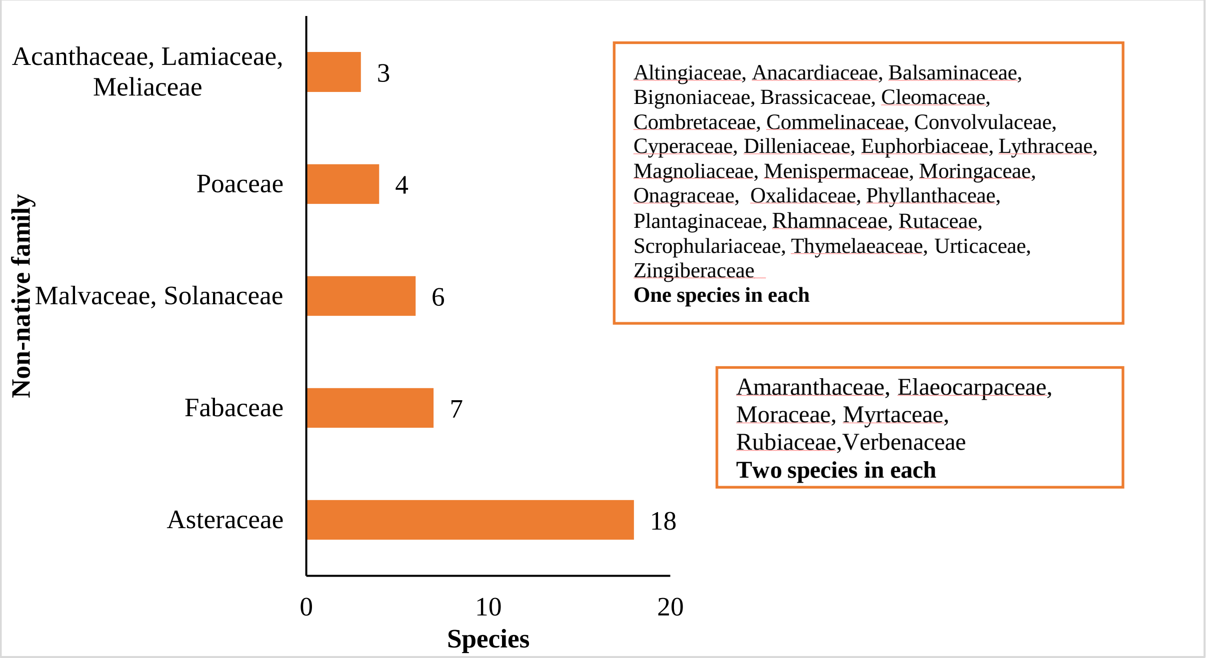
Figure 4.3: Floristic diversity of the study area of (a) Species richness (b) with native and non-native classification


Combretaceae, Euphorbiaceae, Lauraceae, Melastomataceae, Moraceae
3
Elaeocarpaceae, Myrtaceae, Rubiaceae
4
Poaceae
5
Solanaceae
6
Lamiaceae, Meliaceae
7
Acanthaceae
8
Fabaceae
10
Malvaceae
11
Asteraceae
18
Family
0
10
20
Species
Achariaceae, Altingiaceae, Anacardiaceae, Annonaceae, Balsaminaceae, Brassicaceae, Calophyllaceae, Cleomaceae, Commelinaceae, Convolvulaceae, Costaceae, Cyperaceae, Dilleniaceae, Dipterocarpaceae, Menispermaceae, Moringaceae, Musaceae, Onagraceae, Oxalidaceae, Phyllanthaceae, Piperaceae, Plantaginaceae, Rhamnaceae, Sapindaceae, Scrophulariaceae,Simaroubaceae,Tetramelacee, Theaceae, Thymelaeaceae, Zingiberaceae
Single speceis in each species
Arecaceae, Amaranthaceae, Bignoniaceae, Burseraceae, Fagaceae, Lythraceae, Magnoliaceae, Myristicaceae, Rutaceae, Urticaceae, Verbenaceae
Two species in each

(a)


Arecaceae, Burseraceae, Combretaceae, Elaeocarpaceae, Euphorbiaceae, Fagaceae, Myristicaceae, Myrtaceae, Rubiaceae
2
Fabaceae, Lauraceae, Melastomataceae
3
Lamiaceae, Meliaceae
4
Acanthaceae, Malvaceae
5
Native family
0
2
4
6
Species
Achariaceae
Annonaceae
Bignoniaceae
Calophyllacea
Costaceae
Dipterocarpaceae
Lythraceae
Magnoliaceae
Moraceae
Musaceae
Piperaceae
Poaceae
Rutaceae
Sapindaceae
Simaroubaceae
Tetramelaceae
Theaceae
Urticaceae
One species in each

(b)



(c)

Figure 4.4: Family wise distribution of the (a) whole study and according to (b) native and (c) non-native

4.2.2 Sampling Completeness

The observed results (Fig. 4.5) were compared with the Chao1 estimate values and rarefaction accumulation curve values from PAST in order to verify the sampling completeness of the vegetation composition in the various forest types of PTR. The data clearly indicates that the sample completeness in each forest type was achieved by the observed values, which agreed with the values of the Chao1 estimate.

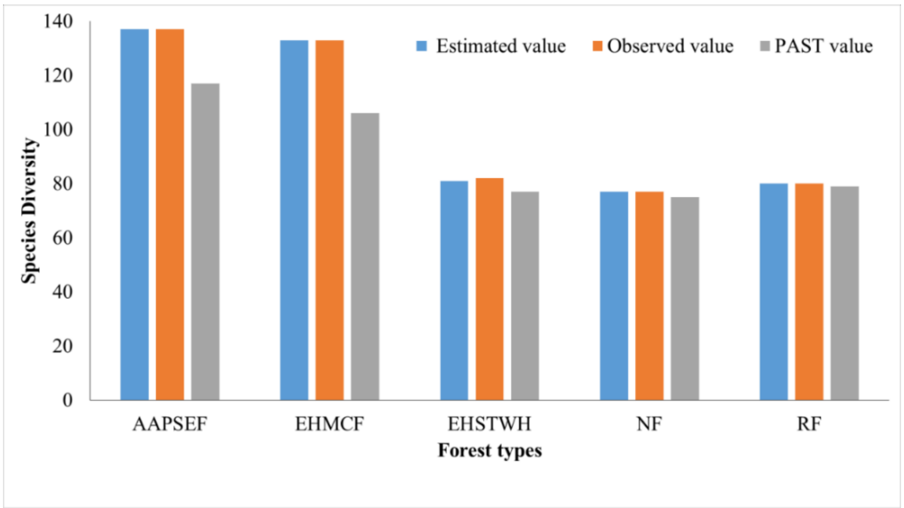


Figure 4.5: Sampling completeness through species richness estimation techniques for the five forest types in the study area.

4.2.3 Diversity Estimation

The study observed that the rarefied accumulation curve of the whole community displayed asymptote stability in all forest types (Fig. 4.6). This clearly revealed that the species richness in each forest type is acceptable for the research. Similarly, the species accumulation curve exhibited asymptote in tree diversity, shrub diversity and herb diversity in each analyzed forest type. In the 208 sampled plots, 151 species were recorded but the richness estimators suggested the asymptotic local richness in each of the forest types to be 139 species in Assam Alluvial Plain Semi Evergreen Forest (137 observed), 133 species in East Himalayan Mixed Coniferous Forest (133 observed), 82 species in East Himalayan Subtropical Wet Hill (81 Observed), 82 species in non-Forest (76 Observed) And 81 species in Riverine Forest (80 Observed) by Chao1 estimator. Similarly, rarefaction was done to evaluate the species accumulation curves of the trees in all forest kinds and demonstrate that the species accumulation curve of trees in all forest types displays the identical stability. The extrapolation curves also demonstrate that they recorded the same number of the species in the phytosociological survey in the floristic survey. From the extrapolation curves, the number of recorded species in Assam Alluvial Plain Semi Evergreen Forest is 59 species (lower and upper 95% confidence interval are 59 and 73), 67 species in East Himalayan Mixed Coniferous Forest (lower and upper 95% confidence interval are 67 and 70), 37 species in East Himalayan Subtropical Wet Hill (lower and upper 95% confidence interval are 36 and 44), 32 species in Non-forest (lower and upper 95% confidence interval are 33 and 52) and 37 species in Riverine forest (lower and upper 95% confidence interval are 37 and 47). In shrubs, the species accumulation curve suggests that local richness of the recorded species displays asymptotic stability in all studied plots in all forest types. In shrub, the estimated and observed values are identical with each other in all forest types. Moreover, in herbs also the same resemblance exists in observed and calculated value (Fig 4.6).

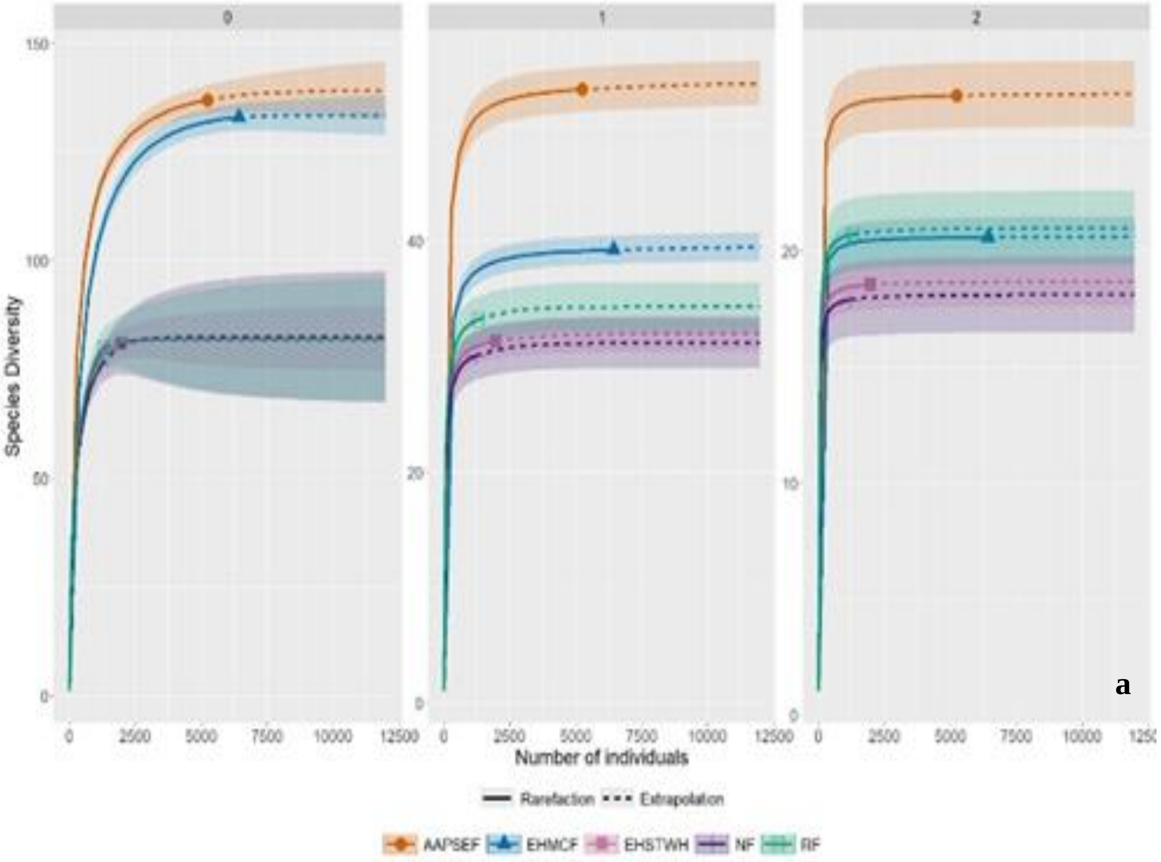
4.3 Species Diversity of Invasive Alien Plant Species in Pakke Tiger Reserve

From the study's geographic area, 43 IAPs have been identified (Table 4.3): 33 herbs, 6 shrubs, 2 grasses and 1 each of sedge and climber. Eleven distinct geographical areas are represented by these species. Tropical America is the preferred habitat for the greatest number of species (27) and is followed by Tropical South America, Brazil, Central America, Europe, the Mediterranean, Mexico, North America and Tropical West Asia (Fig.

4.7). Asteraceae was the dominant families, followed by Fabaceae and Solanaceae with five each, Malvaceae with three, Poaceae, Tiliaceae and Verbenaceae with two each and the remaining families, which included one each of Balsaminaceae, Brassicaeceae, Cleomaceae, Cyperaceae, Lamiaceae, Oxalidaceae and Plantaginaceae (Fig. 4.8). There are no trees among the IAPs recorded from the study area; instead, there are herb species such as *Ageratina adenophora, Ageratum conyzoides, Bidens pilosa, Blumea lacera, Chromolaena odorata, Cardamine hirsuta, Cleome rutidosperma*, shrub species like *Datura metel, Senna alata, Senna occidentalis, Senna tora, Solanum torvum*, and *Urena lobata*, climber species like *Mikania micrantha* and grass species like *Imperata cylindrica and Saccharum spontaneum*; and sedge species like *Cyperus Iria*.

4.3.1 Species Richness of Invasive Alien Plant Species in Pakke Tiger Reserve

With 41, AAPSEF's species richness stands out as most invaded and hosts the most IAP species (Fig. 4.9). Given only 13 IAP species, this sharp difference to the EHSTWH points to a possible susceptibility of the AAPSEF ecosystem to biological incursions. By contrast, with 26 and 21 species respectively, the EHMCF and NF had a somewhat high IAP species richness. At 20 IAP species, the RF had a somewhat lower count, suggesting maybe a greater degree of resistance or resilience in this environment. *Ageratum conyzoides, Bidens pilosa, Chromolaena odorata, Crassocephalum crepidioides, Solanum torvum, Solanum viarum, Synedrella nodiflora, Triumfetta rhomboidea* and *Urena lobata* are IAPs that are common in all five forest types and indicate their wide distribution and ability to thrive across the varied forest environments. However, some IAPs are specifically uniquely found in forest types i.e., in AAPSEF (*Acmella radicans, Blumea lacera, Cardamine hirsuta, Cleome rutidosperma, Corchorus aestuans, Datura metel, Nicotiana plumbaginifolia*, Senna *alata, Senna occidentalis, Senna tora, Sida acuta, Sonchus oleraceus, Tridax procumbens, Xanthium strumarium* and *Youngia japonica*. In EHSTWH, only one was found (*Ageratina adenophora*) and 12 in RF i.e., *Crotalaria pallida, Cyperus iria, Imperata cylindrica, Ludwigia octovalvis, Malvastrum coromandelianum, Mikania micrantha, Mimosa pudica, Oxalis corniculata, Parthenium hysterophorus, Physalis angulata, Saccharum spontaneum* and *Stachytarpheta jamaicensis* were found to be adapted to the unique environmental conditions and ecological niches within those particular ecosystems.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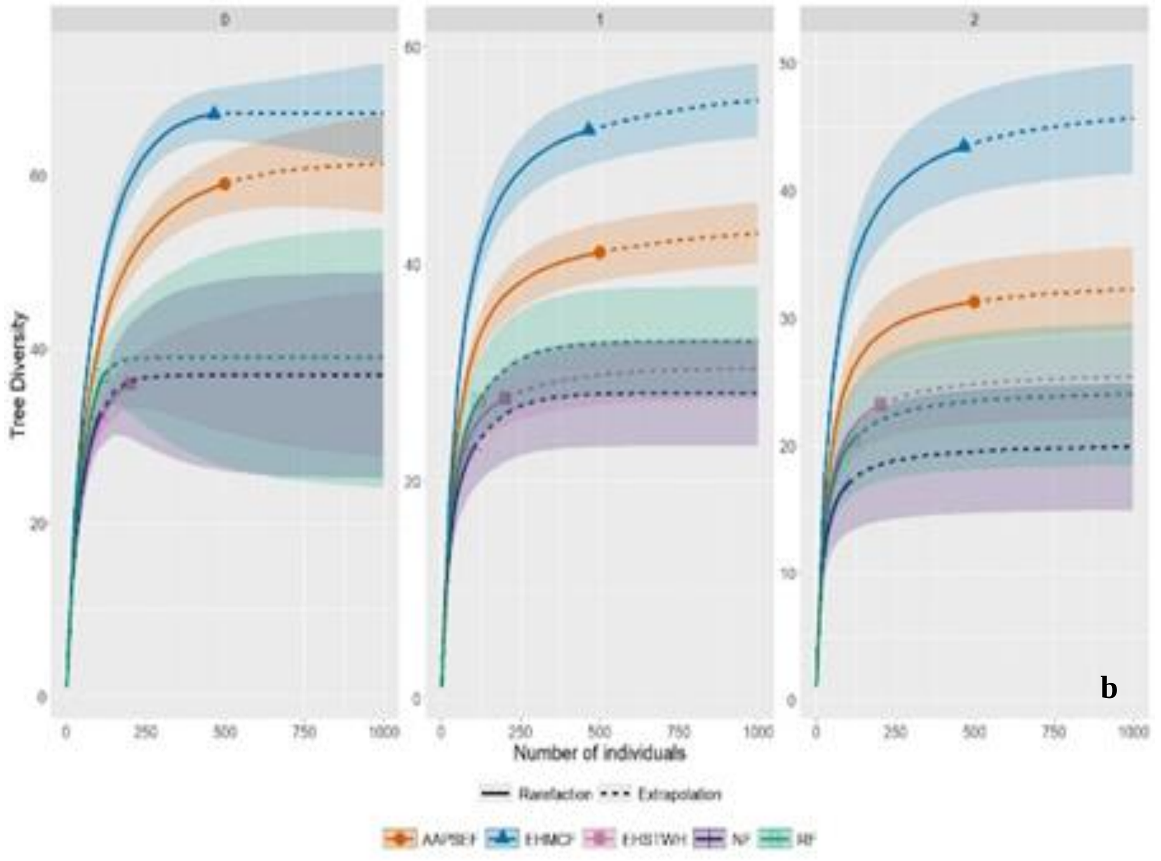
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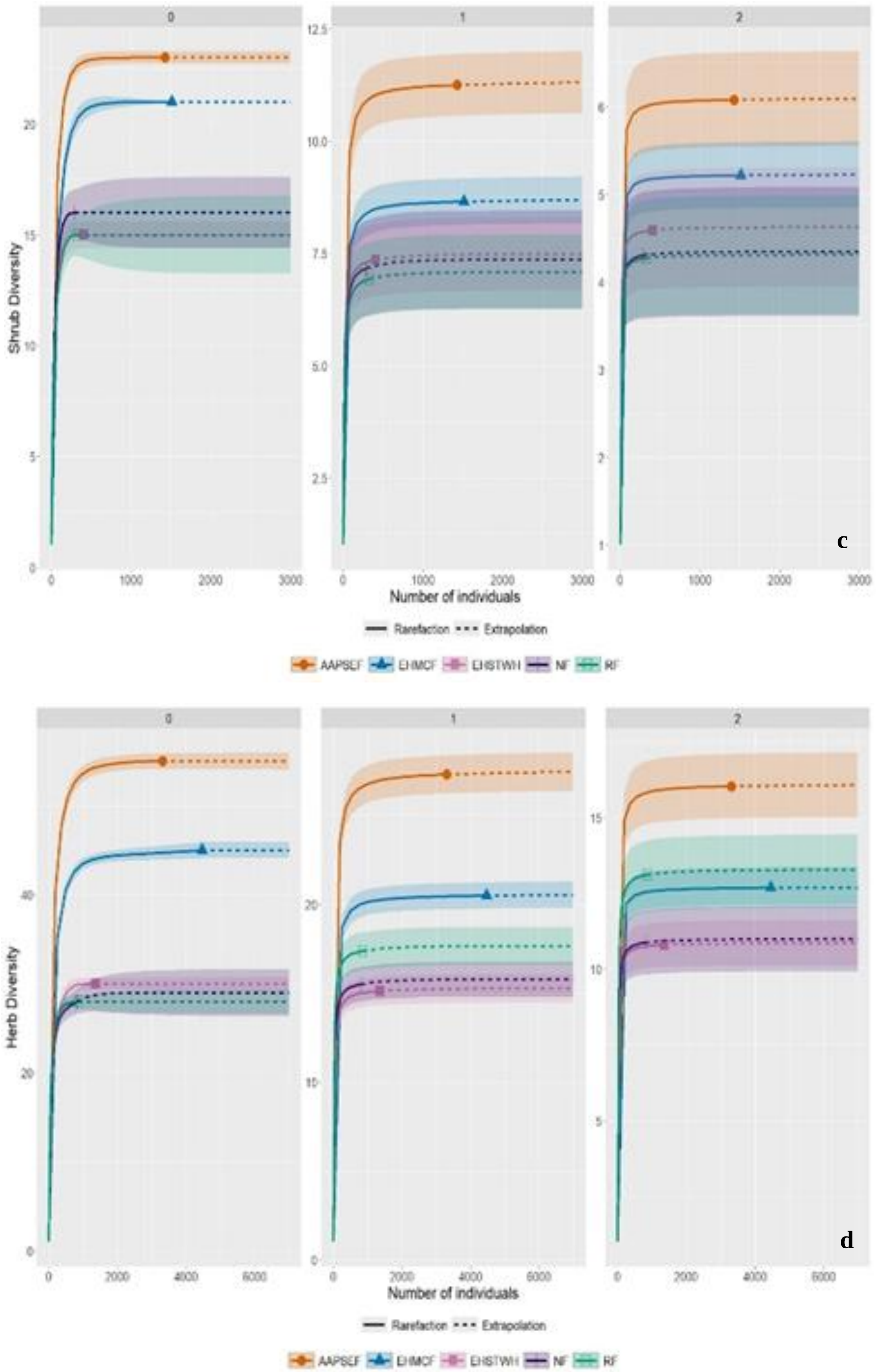


Figure 4.6: Individual- based rarefaction (solid line) and extrapolation curves (dashed lines) for the (a) whole diversity, (b) Trees diversity (c) Shrub diversity and (d) Herb diversity in different forest types in study area. *(0- Chao richness, 1- Chao Simpson and 2- Chao Shannon)

Table 4.3. List of invasive alien plant species recorded in the study area

Species	**Family**	**Habit**	**Nativity**
Saccharum spontaneum	Poaceae	Grass	West As. trop
Acmella radicans	Asteraceae	Herb	S Am. trop
Ageratina adenophora	Asteraceae	Herb	Mexic
Ageratum conyzoides	Asteraceae	Herb	Am. trop
Bidens pilosa	Asteraceae	Herb	Am. trop
Blumea lacera	Asteraceae	Herb	Am. trop
Cardamine hirsuta	Brassicaceae	Herb	Am. trop
Cleome rutidosperma	Cleomaceae	Herb	Am. trop
Corchorus aestuans	Malvaceae	Herb	Am. trop
Crassocephalum crepidioides	Asteraceae	Herb	Am. trop
Crotalaria pallida	Fabaceae	Herb	Am. trop
Eclipta prostrata	Asteraceae	Herb	Am. trop
Galinsoga parviflora	Asteraceae	Herb	Am. trop
Impatiens balsamina	Balsaminaceae	Herb	Am. trop
Imperata cylindrica	Poaceae	Herb	Am. trop
Ludwigia octovalvis	Onagraceae	Herb	Afr. trop
Mikania micrantha	Asteraceae	Herb	Am. trop
Malvastrum coromandelianum	Malvaceae	Herb	Am. trop
Mesosphaerum suaveolens	Lamiaceae	Herb	Am. trop
Mimosa pudica	Fabaceae	Herb	Braz
Nicotiana plumbaginifloia	Solanaceae	Herb	Am. trop
Oxalis corniculata	Oxalidaceae	Herb	Europ
Parthenium hysterophorus	Asteraceae	Herb	N Am. trop
Physalis angulata	Solanaceae	Herb	Am. trop
Scoparia dulcis	Plantaginaceae	Herb	Am. trop
Senna tora	Fabaceae	Herb	S Am. trop
Sida acuta	Malvaceae	Herb	Am. trop
Solanum viarum	Solanaceae	Herb	Am. trop
Sonchus oleraceus	Asteraceae	Herb	Mediterr
Stachytarpheta indica	Verbenaceae	Herb	Am. trop
Synedrella nodiflora	Asteraceae	Herb	West indies
Tridax procumbens	Asteraceae	Herb	Centr Am. trop
Xanthium strumarium	Asteraceae	Herb	Am. trop
Youngia japonica	Asteraceae	Herb	S Am. trop
Cyperus iria	Cyperaceae	Sedge	Am. trop
Chromolaena odorata	Asteraceae	Shrub	Am. trop
Datura metel	Solanaceae	Shrub	Am. trop
Lantana camara	Verbenaceae	Shrub	Am. trop
Senna alata	Fabaceae	Shrub	West indies
Senna occidentalis	Fabaceae	Shrub	S Am. trop
Solanum torvum	Solanaceae	Shrub	West indies
Triumfetta rhomboidea	Malvaceae	Shrub	Am. trop
Urena lobata	Malvaceae	Shrub	Afr. trop

Note*: Africa (Afr.), America (Am.), Asia (As), Brazil (Braz), Central (Centr), Europe (Europ), Mediterranean (Mediterr), Mexico (Mexic), North (N), South (S), Tropical (Trop)

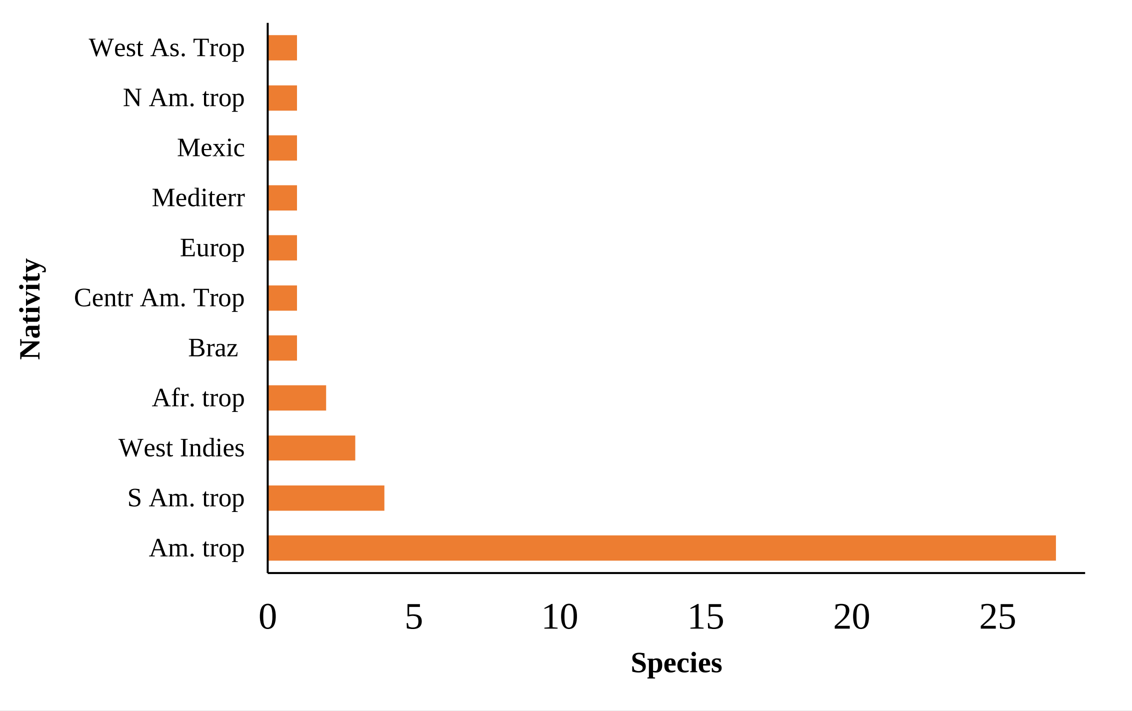


Figure 4.7: Geographical regions of IAPs present in the study area

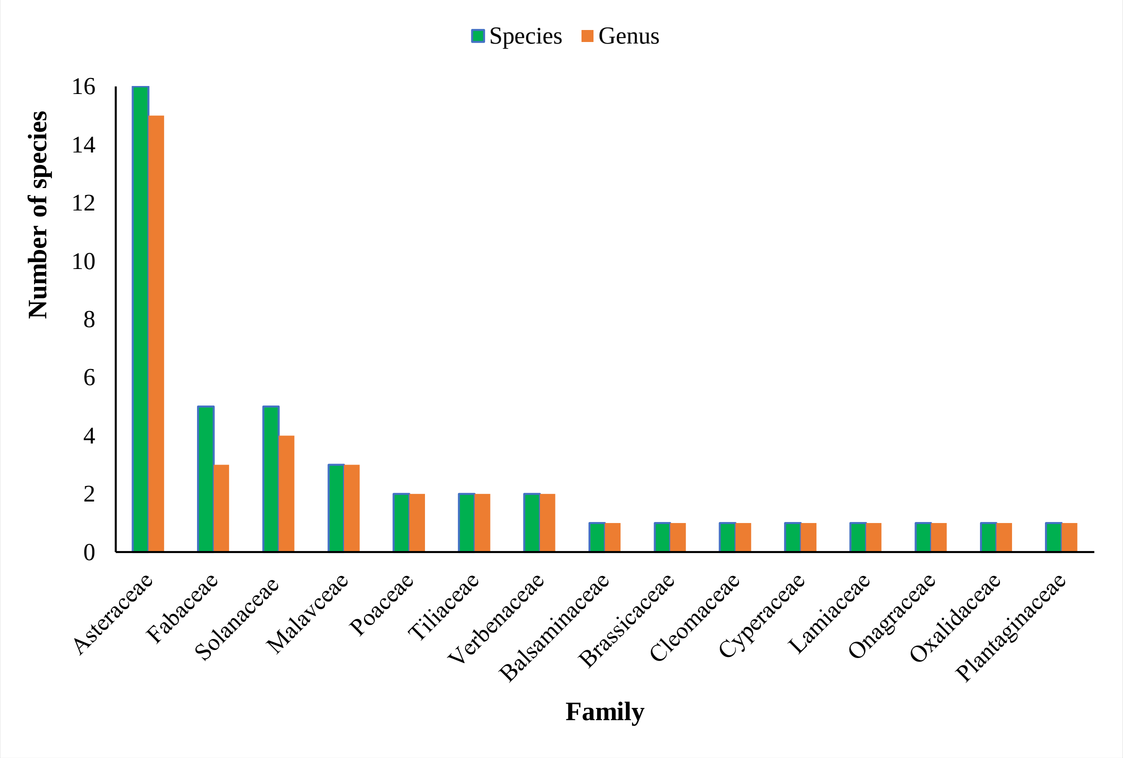


Figure 4.8: Family wise distribution of IAPs with number of genera in the study area.

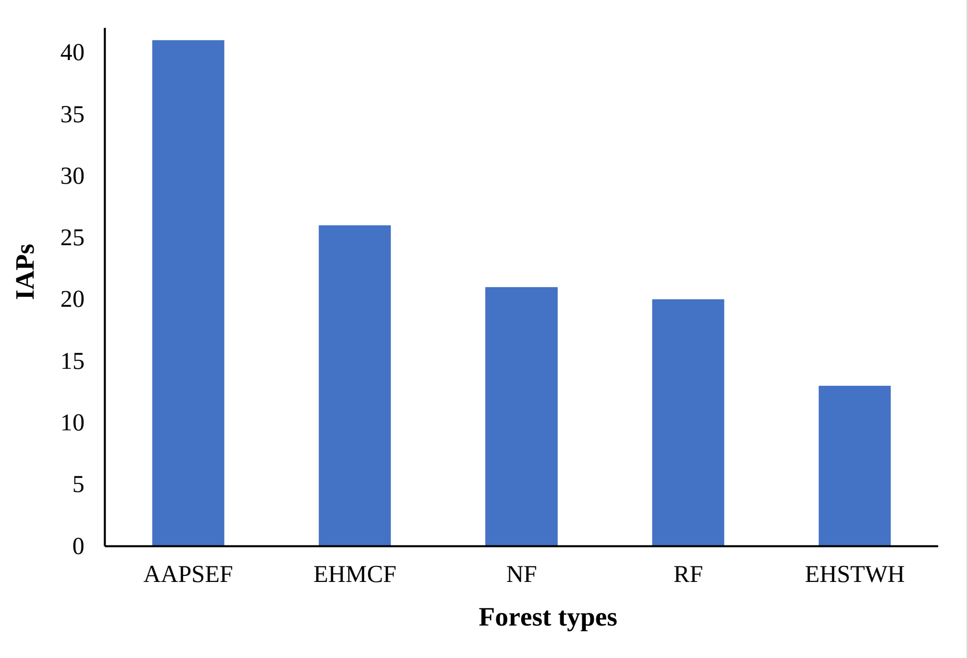


Figure 4.9: Species richness of IAPs in the study area.

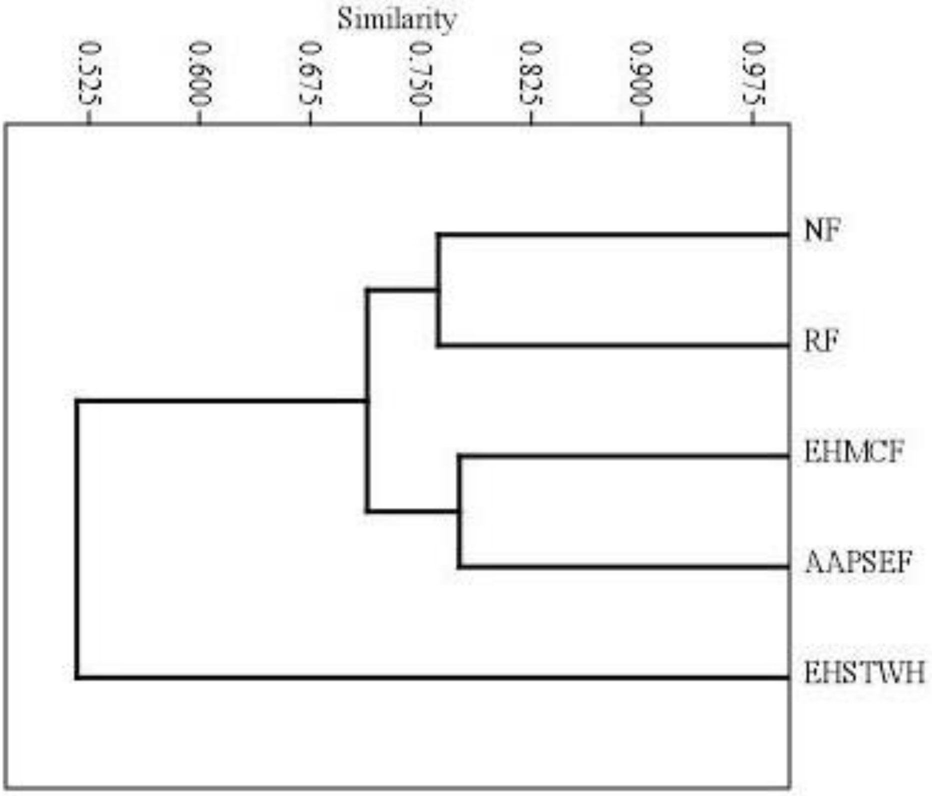


Figure 4.10: Hierarchical clustering of IAPs in different forest types according to Bray-Curtis similarity index

4.3.2 Jaccard Similarity Index of Invasive Alien Plant Species Between Forest Types

According to Jaccard Similarity Index with Cophenetic Correlation value of 0.9841, the composition of IAPs across five distinct forest types shows that they cluster into two primary groups (Fig. 4.10). This index measures the similarity association of plant between forest types and uses Paired Group (UPGMA- Unweighted Pair Group Method with Arithmetic Mean) for evaluation. First, there is a strong resemblance between the plant species assemblages of NF and RF as they constitute a single, closely related cluster. Second, the second cluster is made up of EHMCF and AAPSEF, indicating that their plant communities are more different from those of the NF-RF group. In contrast to the other forest types, EHSTWH exhibits the most different behavior, branching out first from the bigger cluster and suggesting the presence of a distinct plant community composition. Additionally, EHMCF and AAPSEF exhibit comparatively high similarity inside the second cluster, suggesting that they have a greater number of plant species in common than their relationship with EHSTWH.

4.3.3 Level of Invasion

Based on all forest types, the current investigation found that AAPSEF had the greatest amount of invasion while EHSTWH had the lowest level (Fig. 4.11). In the current research region, the overall degree of invasion reduced from 30% to 16% as measured by the percentage of IAPs to all plant species. Additionally, the areas with the greatest amount of IAP invasion were AAPSEF, NF, RF, EHMCF, and EHSTWH, in that order. These findings unequivocally demonstrate that higher levels of disturbance in forest types - such as those near roads, populated areas, livestock, and human interference - are associated with an increased incidence of invasions. On the other hand, compared to other forest types, the EHSTWH at higher elevations have less incursions, suggesting that disturbance in this form of forest is minimal.

4.4 Distribution and Invasion Pattern of Selected Invasive Alien Plant Species in Pakke Tiger Reserve

4.4.1 *Ageratum conyzoides*

Ageratum conyzoides was found to be prevalent in all forest's types of the study area. In the EHSTWH forest type (Table 4.4), it had the greatest relative abundance (3.98%), followed by NF (3.72%) while the least was observed in AAPSEF (2.63%). However, NF forest type had the highest relative density (23.09%), indicating some degree of dominance and minimum in RF (18.50%).

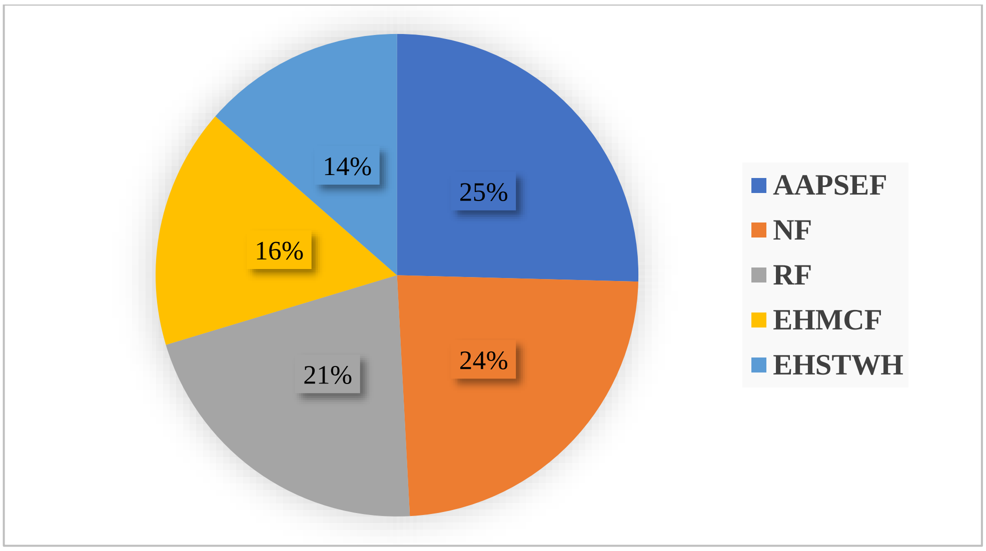


Figure 4.11: The overall level of invasion (%) in each forest types of study area.

Table 4.4: *Ageratum conyzoides* distribution across the different forest types in the study area.

Forest Types	Relative Density (%)	Relative Abundance (%)	Abundance /Frequency Ratio	Cover Area (Km²)	Invasion (%)
AAPSEF	19.07	2.63	0.06	77.33	19.79
EHMCF	20.82	2.67	0.07	74.92	20.55
EHSTWHF	21.72	3.98	0.09	14.79	18.53
NF	23.10	3.73	0.05	1.75	10.00
RF	18.51	2.94	0.05	1.10	11.67

A. conyzoides was found to have a maximum density of 37,931.03 individuals per hectare in the EHSTWH forest and minimum in 30,888.88 individuals per hectare in RF. The species' overall abundance trended similarly, reaching its high in EHSTWH and declining in RF. All forest types had a contagious, or clustered, distribution pattern for *A. conyzoides*, suggesting the existence of unique patches or aggregations in each type of forest, as the abundance to frequency (A/F) ratios was observed to be more than 0.05. Moreover, the study showed that these species had invaded 19% of the regions of each forest type, including AAPSEF (20%), EHMCF (20%), EHSTWH (18%), NF (10%), and RF (11%). For each of the study area's forest types, the *Ageratum conyzoides* density was computed. NF had the lowest invasion density at 1,000±133.66 m² followed by AAPSEF

(1,978.28±139.22 m²) and EHMCF (2,055.31±136.33 m²). These species' total cover areas were 77.325 km² in AAPSEF, 74.919 km² in EHMCF, 14.79 km², 1.749 km², and 1.01 km².

4.4.2 *Bidens pilosa*

Bidens pilosa was also found to be present in all of the study area's forests. It showed the highest relative abundance (3.49%) and relative density (10.19%) in the NF forest type, where it was the main herb layer (Table 4.5). Conversely, the species exhibited the lowest relative density (7.07%) and relative abundance (2.18%) under the AAPSEF forest type. The species was determined to have the lowest relative density (4.24%) in the EHSTWH forest type and the maximum relative density (5.24%) in the NF forest type.

Table 4.5: Distribution of *Bidens Pilosa* across the different forest types in the study area.

Forest Types	Relative Density (%)	Relative Abundance (%)	Abundance /Frequency Ratio	Cover Area (Km²)	Invasion (%)
AAPSEF	6.22	2.34	0.15	40.39	10.34
EHMCF	7.08	2.19	0.13	44.91	12.32
EHSTWHF	4.24	2.82	0.23	9.48	11.88
NF	10.19	3.49	0.10	1.46	8.33
RF	5.46	2.78	0.17	0.64	6.79

In the EHSTWH, there were 26,875 *Bidens pilosa* individuals per hectare, whereas in the NF forest, there were 31,250 individuals per hectare. The species' abundance varied depending on the kind of forest, from 2.687 to 3. 125. *Bidens pilosa* was observed to have a contagious distribution pattern over all forest types, much like *A. conyzoides*. Furthermore, these species have taken over 10% (40.39 km²) of AAPSEF, 12% (44.90 km²) of EHMCF, 9% (9.47 km²) of EHSTWH, 8% (1.45 km²) of NF, and 6% (1.10 km²) of RF of each forest type. *Bidens pilosa* exhibits the highest invasion in EHMCF (1231.959±124.76 m²) followed by 1187.5±144.76 m² and the lowest invasion in RF (678.571±126.88 m²).

4.4.3 *Chromolaena odorata*

The species had established as a dominating shrub species layer along the roadside in all study locations' forest types. This species had the greatest relative abundance (10%) and relative density (46%) of all invasive species in NF, whereas it had the lowest relative abundance (6.5%) and relative density (39%) in EHMCF. On the other hand, EHSTWH was discovered to have the greatest relative density (51%) and almost the same relative abundance as NF and RF (Table 4.6). The species' density varied from 1815.38 to 2533.33 individuals per hectare, with RF having the greatest density and EHSTWH having the lowest.

Table 4.6: Distribution of *Chromolaena odorata* across the different forest types in the study area.

Forest Types	Relative Density (%)	Relative Abundance (%)	Abundance /Frequency Ratio	Cover Area (Km²)	Invasion (%)
AAPSEF	43.05	6.92	0.06	72.10	18.45
EHMCF	39.30	6.60	0.07	72.90	20.00
EHSTWHF	51.40	10.14	0.08	10.85	13.59
NF	46.83	10.99	0.06	2.76	15.77
RF	47.30	10.59	0.07	1.89	20.00

The abundance was 4.53 to 5.38, with RF having the greatest abundance and NF having the lowest. Based on the value derived from the abundance (A) and frequency (F) ratio, the species displayed contagious distribution across all forest types, indicating that the species is prevalent throughout the research region regardless of elevation. Moreover, it was shown that among all forest types, EHMCF (72.90 km²) and RF (1.89 km²) had the largest invaded total cover, while EHSTWH (10.84 km²) had the lowest. The density per plot in the invasion region ranged from 1359±262.31 m² to 2000±144.44 m², with RF having the greatest density and EHSTWH having the lowest.

4.4.4 *Mikania micrantha*

Mikania Micrantha in the study region was distributed in all forest types with *Chromolaena odorata* in close proximity forming a bonding pair. With regard to relative density and abundance, the species had the greatest values in EHSTWH (Table 4.7). Its relative abundance with NF and RF is almost same, but its relative density differs. This species is the least common in AAPSEF (13.60%), with EHMCF (13.84%) having the highest relative density. These species varied in abundance, with RF having the greatest and NF having the lowest, ranging from 3.70 to 4.47. There are 370873.33 individuals per hectare in NF and 44761.90 individuals per hectare in EHMCF, respectively, depending on the species. Similar to EHSTWH and RF, the species developed a dense population in the EHMCF. Based on frequency and abundance ratios, the distribution pattern of these species indicates that they are shared by all kinds of forests meaning contagiousness. The percentage cover of invasion by these species was greatest in EHMCF (68.51 km²; 18%), EHSTWH (12.76 km²; 16%) and AAPSEF (58.78 km²; 15%) in their respective total forest cover. The invasion by density per plot in each forest type was 1504±152.95 m² in AAPSEF, 1879.699±134.38 m² in EHMCF, 1600±148.06 m² in EHSTWH, 979.166±142.89 m² in NF and 1309.523 ±143.59 m² in RF respectively.

Table 4.7: Distribution of *Mikania micrantha* across the different forest in the study area.

Forest Types	Relative Density (%)	Relative Abundance (%)	Abundance /Frequency Ratio	Cover Area (Km²)	Invasion (%)
AAPSEF	13.60	2.99	0.12	58.79	15.05
EHMCF	13.84	3.15	0.14	68.52	18.80
EHSTWHF	16.78	4.46	0.14	12.77	16.00
NF	12.09	4.15	0.12	1.71	9.79
RF	12.52	4.26	0.17	1.24	13.10

Table 4.8: Distribution of *Solanum torvum* across the different forest types in the study area.

Forest Types	Relative Density (%)	Relative Abundance (%)	Abundance /Frequency Ratio	Cover Area (Km²)	Invasion (%)
AAPSEF	4.07	2.43	0.25	49.72	12.73
EHMCF	5.19	2.66	0.26	35.49	9.74
EHSTWHF	3.85	3.72	0.44	6.65	8.33
NF	5.98	3.78	0.21	1.75	10.00
RF	4.53	3.23	0.27	0.47	5.00

4.4.5 *Solanum torvum*

The species had the greatest relative richness and density in EHMCF in the research region (Table 4.8). These species are the most abundant in NF, with EHSTWH coming in second. In RF, these species have the lowest relative abundance and density. The distribution regions of these species had similar abundances in AAPSEF, EHMCF, and EHSTWH compared to NF and RF. The species' density ranged from 1000 individuals per hectare to 1263.15 in EHMCF, while its abundance varied from 2.5 in RF to 3.15 in EHMCF. The species is contagious in all forest types, which include 49.72 km² in AAPSEF, 35.49 km² in EHMCF, 6.65 km² in EHSTWH, 1.74 km² in NF and 0.42 km² in RF, according to distribution patterns. The overall cover percentage of invasion varies by forest type: 12.72% in AAPSEF, 9.73% in EHMCF, 8.33% in EHSTWH, 10% in NF and 5% in RF. The invasion by density per plot is 1272±130.65 m², with each forest type following in increasing sequence (973±106.79 m², 833.33±100 m², 1000±83.66 m² and 500±70.71 m²).

4.4.6 *Solanum viarum*

The species is found in NF with the greatest relative abundance and density, and AAPSEF with the lowest relative abundance and density. Comparably, it was found that NF had the greatest relative density and abundance whereas EHSTWH had the lowest (Table 4.9). These species vary in abundance from 3.05 in AAPSEF to 3.64 in EHMCF, while their density varied from 30512.82 ind/ha in AAPSEF to 36440.67 ind/ha in EHMCF. Furthermore, these species are just as contagious across forest types as *Solanum torvum*, according to the abundance and frequency ratio. In each of their respective forest cover areas, these species cover 40.60 km² (10.39%) in AAPSEF, 54.03 km² (14.82%) in EHMCF, 7.61 km² (9.54%) in EHSTWH, 1.61 km² (9.23%) in NF and 0.66 km² (7%) in RF. (Fig. 4.23d). In case of invasion, the density per plot for each forest type was as follows: 1039.47±166.94 m² in AAPSEF, 1482.45±167.91 m² in EHMCF, 954.54±103.57 m² in EHSTWH, 923.07±170.97 m² in NF and 700±126.49 m² in RF.

Table 4.9: Distribution of *Solanum viarum* across the different forest types in the study area.

Forest Types	Relative Density (%)	Relative Abundance (%)	Abundance /Frequency Ratio	Cover Area (Km²)	Invasion (%)
AAPSEF	4.89	2.23	0.18	40.61	10.39
EHMCF	4.98	2.55	0.25	54.04	14.82
EHSTWHF	2.17	2.57	0.37	7.62	9.55
NF	9.24	4.00	0.15	1.61	9.23
RF	6.52	3.58	0.23	0.66	7.00

4.4.7 *Synedrella nodiflora*

The species exist across various forest types in the study site; however, they are concentrated in EHMCF, NF, and RF. The species had the lowest relative abundance and relative density in AAPSEF and the greatest relative abundance and density in NF. In contrast, EHSTWH had the lowest relative density and abundance, whereas NF had the highest. Between 2.44 and 3.76, the abundance was highest in RF and lowest in EHSTWH (Table 4.10). The density ranged from 24444.44 to 37692.30 persons per hectare, with RF having the highest density and EHSTWH having the lowest. Additionally, the distribution patterns demonstrate contagiousness in all kinds of forests, where it is concentrated everywhere. Furthermore, the highest levels of invasion across all forest types were seen in AAPSEF at 38.64 km² (9.90%), EHMCF at 41.08 km² (11.27%), EHSTWH at 4.87 km² (6.11%), NF at 1.88 km² (10.78%) and RF at 0.94 km² (10%). According to the forest types, the invasion area in terms of density per plot were, in ascending order, 990.19±144.25 m², 1127.11±134.39 m², 611.11±158.99 m², 1078.94±134.64 m² and 1000±116.58 m².

Table 4.10: Distribution of *Synedrella nodiflora* across the different forest types in the study area.

Forest Types	Relative Density (%)	Relative Abundance (%)	Abundance /Frequency Ratio	Cover Area (Km²)	Invasion (%)
AAPSEF	2.85	4.58	0.37	38.68	9.90
EHMCF	4.40	4.39	0.28	41.09	11.27
EHSTWHF	2.80	6.08	0.54	4.88	6.11
NF	5.56	6.78	0.18	1.89	10.79
RF	1.69	4.92	0.40	0.94	10.00

4.4.8 *Triumfetta rhomboidea*

The species is more common in AAPSEF, EHMCF, and RF than in other forest types. The species had the maximum relative abundance (8.47%) and relative density (5.55%) in NF, and the lowest relative abundance (5.06%) and relative density (7.47%) in EHMCF. However, the maximum relative density (11.14%) and relative abundance (8.11%) were found in RF, whereas EHSTWH had the lowest relative density (3.73%) and relative abundance (5.19%). The abundance of this species ranges from 3 to 4.12, with the greatest in RF and the lowest in EHSTWH (Table 4.11). The density of these species varied from 1200 individuals per hectare in EHSTWH to 1650 in RF. The total cover area invaded by this species in each forest type was 63.69 km² (16.30%) in AAPSEF, followed by 42.30 km² (11.60%) in EHMCF, 6.98 km² (8.75%) in EHSTWH, 1.53 km² (8.75%) in NF and 0.88 km² (9.375%) in RF. The invasion area by density per plot in each forest type was 1630.434±126.99 m², 1160.714±161.51 m², 875±200 m², 875±100 m² and 937.5±64.08 m² in increasing order. The abundance by frequency ratio indicates a contagious nature across all forest types in the research region.

Table 4.11: Distribution of *Triumfetta rhomboidea* across the different forest types in the study area.

Forest Types	Relative Density (%)	Relative Abundance (%)	Abundance /Frequency Ratio	Cover Area (Km²)	Invasion (%)
AAPSEF	6.95	5.35	0.20	63.70	16.30
EHMCF	7.48	5.07	0.22	42.31	11.61
EHSTWHF	3.74	6.08	0.41	6.98	8.75
NF	5.56	8.47	0.28	1.53	8.75
RF	11.15	8.11	0.17	0.89	9.38

4.4.9 *Urena lobata*

Urena lobata is widely distributed across the study area, including AAPSEF, EHMCF, and RF. It exhibited the greatest relative abundance and density in NF but the lowest relative abundance and density in EHMCF (Table 4.12). The species had the highest relative density and abundance in NF and the lowest relative density and abundance in AAPSEF. The abundance varied from 2.91 to 3.76, with the lowest in EHSTWH and the greatest in APPSEF. The species density varied between 1165.21 individuals per hectare in EHSTWH to 1505.88 individuals per hectare in AAPSEF. The total area invaded by these species throughout the forest types were 67.02 km² (17.15%) in AAPSEF, 59.73 km² (16.38%) in EHMCF, 7.80 km² (9.78%) in EHSTWH, 1.40 km² (8.40%) in NF and 1.006 km² (10.66%) in RF. Furthermore, the density per plot by invasion area in each forest type was 1715.686±147.77 m² in AAPSEF, 1638.88±142.87 m² in EHMCF, 978.260±108.35 m² in EHSTWH, 840.90±99.021 m² in NF and 1066.66±105.56 m² in RF. Based on the abundance and frequency ratios, the distribution patterns of AAPSEF, EHMCF, EHSTWH and RF values were larger than 0.05, demonstrating their contagiousness in four forest types, but NF values fell between 0.025 and 0.05, showing random distribution in these forest regions.

Table 4.12: Distribution of *Urena lobata* across the different forest types in the study area.

Forest Types	Relative Density (%)	Relative Abundance (%)	Abundance /Frequency Ratio	Cover Area (Km²)	Invasion (%)
AAPSEF	15.75	5.57	0.10	67.03	17.16
EHMCF	24.27	5.11	0.07	59.74	16.39
EHSTWHF	20.87	5.91	0.07	7.81	9.78
NF	27.38	7.59	0.05	1.47	8.41
RF	18.24	7.08	0.08	1.01	10.67

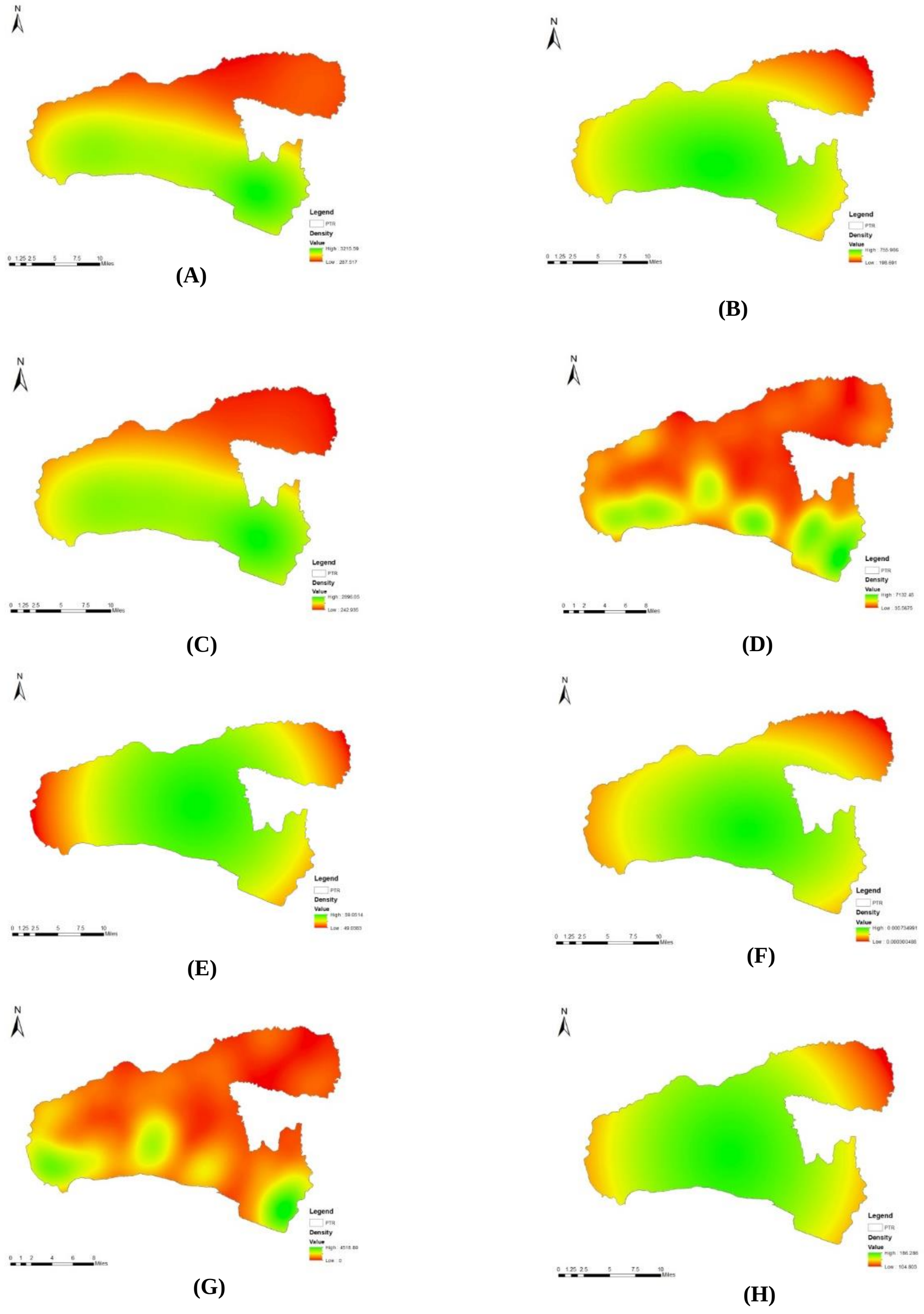
4.5 Mapping of Invasive Alien Plant Species by the Kernel Density Estimation Method

The selected species distribution pattern in the study region was simulated by using the kernel density estimation method through species abundance **(**Chengzhen et al., 2000). Mapping of each species were formulated through relative abundance in ArcGIS to provide their relative distribution with distinct colors differentiation indicating their presence and absence in the study area (Fig. 4.12).

4.6 Discussion

4.6.1 Vegetation Composition

Due to the high biological value of this habitat and the rapidity of forest loss, the CBD had asked that Parties "establish biologically and geographically representative networks of Protected Areas" and evaluate the representativeness of these networks with respect to different types of forests (CBD, 2009). Increased biological invasion, infrastructure development, urbanisation, and other factors are putting more stress on forests. On the other hand, the hazard of declining environmental services to society is made worse by climate change (Ashutosh and Roy, 2021). Consequently, it is essential to regularly monitor and evaluate forest resources in order to identify the causes of deterioration early on and implement the best planning, policy, and management strategies for protecting and replenishing the resource. However, throughout time, a comprehensive examination of the forest vegetation in the state had been carried out (Singh et al., 2002; Behera and Kushwaha, 2007; Rana and Gairola, 2009; Saikia et al., 2017; Dash et al., 2021). However, the study region, which is mostly associated with the hornbill population, had not received much scholarly attention (Datta, 1998; Velho and Krishnadas, 2011; Dasgupta et al., 2022).



(A)

(B)

(C)

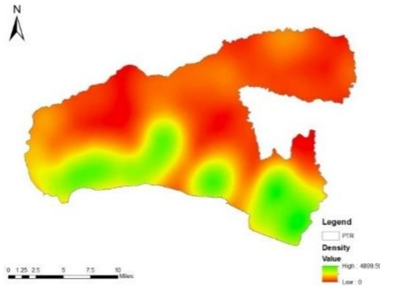
(D)

(E)

(F)

(G)

(H)



(I)

Figure 4.12: Distribution maps of selected IAPs -A- *A. conyzoides*, B- *B. pilosa*, C- *C. odorata*, D- *M. micrantha*, E- *S. torvum*, F- *S. viarum*, G- *S. nodiflora*, H- *T. rhomboidea* and I- *U. lobata* (*orange colour- lower presence and green colour- abundantly distributed).

Based on logging history, the study area's tree density is similar to Datta (1998) stated range of 257 ind/ha to 686 ind/ha. The tree density found by logging and unlogging forests during a 16-year recovery period is comparable to the values in the range of 114 ind/ha to 601 ind/ha reported by Dasgupta et al. (2022) and Velho and Krishnadas (2011). The study's tree density, which ranges from 300 to 600 ind/ha, includes all kinds of forests. This range is consistent with earlier observations made by researchers from the same study site. In addition, the results showed that the Shannon Wiener index ranged from 3.30 to 4, which is comparable to the range of 3.1 to 3.5 noted in earlier research (Velho and Krishnadas, 2011).

Table 4.13: Details of the study sites' vegetation

		Forest types				
Vegetation parameters	**Habit**	**AAPSEF**	**EHMCF**	**EHSTWHF**	**NF**	**RF**
Species richness	Tree	59	67	36	33	37
	Shrub	23	21	15	16	15
	Herb	55	45	31	28	28
Total		137	133	82	77	80
Species diversity (H')	Tree	3.176	3.957	3.319	3.13	3.312
	Shrub	2.42	2.158	1.995	1.97	1.935
	Herb	3.307	3.02	2.716	2.74	2.855

4.6.2 Distribution of Invasive Alien Plant Species in the study area

Mountains are key features of the Earth’s surface and host a substantial proportion of the world’s species (Antonelli et al., 2018). Mountains are diversity cradles because their diversifications in morphological, physiological, or behavioral traits enable species to better use mountainous niches (Hughes and Atchison, 2015). Due to this a great variety of flora and fauna find home in mountains (Körner, 2004). Mountain communities and ecosystems are experiencing environmental degradation, social and economic decline, and marginalization due to the unprecedented pressures of globalization, urbanization, and mass tourism (Jansky et al., 2002). Furthermore, changes in the mountain environment are also facilitated by climatic conditions (Rangwala and Miller, 2012). Climate change had also been associated with the proliferation of IAPs in natural forests, including *Ageratina adenophora*, *Ageratum conyzoides, Bidens pilosa, Chromolasena odorata, Lantana camara, Mesosphaerum suaveolens, Mimosa pudica, Mikania micrantha, Parthenium hysterophorus, Senna tora, Senna occidentalis* and *Xanthium strumarium*, (Shrestha and

Shrestha, 2019). As the mentioned IAPs are present in the current study area, this is anticipated to have an effect on native species' ability to compete.

In general, mountain ecosystems have evaded invasion due to their harsh climate and low human population densities (Kueffer et al., 2013). Nevertheless, in recent years, there had been an increase in the number and variety of IAPs in mountain ranges (Johnston and Pickering, 2001; Pickering et al., 2008), suggesting that the probability of invasion impacts increasing in the future (Williamson and Fitter, 1996). Furthermore, the danger posed by IAPs to high biodiversity will probably rise as a result of globalization and climate change even though they are mostly found in lowland environments (Pauchard et al., 2009).

However, it is anticipated that shifting climate conditions would accelerate the pace of IAPs growth and migration across elevation ranges (Hulme 2017; Dainese et al. 2017). Numerous IAPs species in hilly regions like the Himalaya have already started to takeover higher mountain elevations (Pauchard et al. 2009). Therefore, in order to monitor these species and develop management strategies to control their invasion, it is necessary to understand their patterns of distribution and level of invasion in different forest habitats or types.

In the present study, the species richness of the IAPs across the different forest types indicate a decrease in the number proportion with the increased of the elevation from 13 in EHSTWH to 41 in AAPSEF. Taking into account the elevational gradient as shown by earlier studies, these are typical findings for mountainous regions (Alexander et al., 2009). However, the numbers of the IAPs rises in AAPSEF followed by EHMCF and NF as opposed to RF and EHSTWH, which can be owing to the fact that the forest types is more disturbed as it is comparatively close to the road or villages as there is frequent activity of human interferences. The decrease in IAPs species richness with rising elevation is explained by Pauchard et al. (2009), who state that propagules from human sources distribute unidirectional at low height before moving to higher elevation. A common occurrence in mountain ecosystems is the pattern of species turnover with height, which is referred to as the "elevation gradient”.

On the other hand, a unimodal distribution of elevation indicates that the IAPs species richness in the Himalayan region had a high proportion of IAPs up to 2000 m asl, after which it reduces (Khuroo et al., 2011). Though, it the elevation is restricted to 2050m, the unimodal distribution is discovered since the present research exclusively examines forest types. The research does, however, conform to the overall pattern of declining IAPs abundance with elevation, as reported by Bhattarai et al. (2014) throughout the Himalayan area. According to Averett et al. (2016), the distribution of IAPs in mountainous forest types seems to be most strongly associated with elevation. This study demonstrates that the overall species richness proportion of IAPs is negatively correlated with elevation. The results are in accordance with many global studies (McDougall et al., 2005; Arévalo et al., 2005; Kalwij et al., 2008) and some studies from IHR (Kosaka et al., 2010; Mustaqeem et al., 2018).

The level of invasion in each habitat or forest types is one of the crucial points which need to be understand for effective management and planning of IAPs control (Myśliwy et al., 2014). In the present study, the results shows that the AAPSEF, NF and RF which are near to roads, villages and human interferences are more invaded as compared to other forest types was observed, which are in accordance with the global studies (Vilà et al., 2007; Medvecká et al., 2018; Potter et al., 2023).

4.6.3 Distribution of Selected Invasive Alien Plant Species

IAPs have posed a major risk to human health (Jones, 2019), ecosystem services, biodiversity, and environmental quality in the area (Pejchar and Mooney, 2009; Bartz and Kowarik, 2019). IAPs have put native biodiversity at considerable risk all across the globe, whether they are found in aquatic, terrestrial, or island habitats (Enserink, 1999). The Himalayan area, which is regarded as one of the world's biodiversity hotspots, is more vulnerable to IAP invasion (Khuroo et al., 2021). Due to the region's fast economic expansion, which accounts for the majority of IHR, IHR is under unprecedented anthropogenic pressure. This will eventually result in the introduction and proliferation of IAPs (Pathak et al., 2019). Furthermore, IHR's ecologically sensitive alpine regions are continually grappling with the severity of climate change and IAP invasion, which is becoming worse on all fronts (Mack et al., 2000; Tripathi et al., 2012). It is recognised that certain IAPs that pose a major threat to local residents and forest dynamics are rapidly expanding along the IHR.

A number of them, including *Ageratum conyzoides, Mikania micrantha, Chromolaena odorata, Lantana camara, Ageratina adenophora* and *Parthenium hysterophorus*, have attracted the attention of scientific organisations and are now the focus of extensive study due to their successful invasion rate. All the same, most IAPs stay away from forested, steep locations. But according to reports, *Ageratum conyzoides* and *Lantana camara* are encroaching on the forest floor vegetation, endangering its variety and richness (Kohli et al., 2006). Comparably, the lower and middle Himalayan regions are seeing a fast encroachment of *Parthenium hysterophorus* (Kohli et al., 2004).

One of the problematic weeds that is quickly encroaching on IAPs across a variety of habitats in tropical and subtropical countries is *Ageratum conyzoides*, sometimes referred to as billy goat weed (Batish et al., 2009). Although the species is found around the world, it originated in tropical America (Reddy, 2008). According to Waterhouse (1993), it is ranked as the 15th most problematic weed in Southeast Asia and the Pacific Ocean. According to reports (Kosaka et al., 2010), the species is one of the invasive species in the IHR that may develop up to an elevation of 3000m. Additionally, the species - which is a prominent weed - can readily shift into the eastern Himalayan region's tropical forest (Singh et al., 2002; Arunachalam and Arunachalam, 2002). The species was detected in all of the forest types studied in this research, with the greatest RD and highest RA in the NF and EHSTWH, respectively. According to research by Arunachalam and Arunachalam (2002), these species predominate in the tropical forests of Arunachal Pradesh under the canopy of *Dendrocalamus hamiltonii*. The species exhibits an increasing tendency of RD and RA with the rising elevation when the range as a whole is analysed. Beyond the research region, the species may, however, exhibit declining patterns as elevation rises. Dogra et al. (2009) also documented the decline in native species richness that occurred in the IHR's wooded regions when *Ageratum conyzoides* invaded those areas. It is anticipated that this species will spread over the eastern and western Himalayan regions (Lamsal et al., 2018). Additionally, it was observed by Ray et al. (2019) that the species had a greater capacity to aggressively colonise the lower regions of Arunachal Pradesh. As a result, the present research offers an evaluation of the species' range and the biological danger it poses to the forest's understory vegetation and native flora.

Bidens pilosa, a tropical American native and one of the world's most poisonous invasive plants, had spread to temperate, tropical, and subtropical areas with success (Ballard, 1986; Kosaka et al, 2010; Bartolome et al., 2013). The species can more easily invade new environments due to its increased phenotypic plasticity and adaptability (Malik et al., 2016). Additionally, the distribution of species in the western and eastern Himalayan parts of IHR is gradually increasing as elevation rises from subtropical to subalpine regions (Kosaka et al., 2010; Rana et al., 2019; Khatri et al., 2022; Verma et al., 2023). Its current elevational ranges in the western Himalaya are predicted to increase its distribution northward by 682 m asl; yet, reports place its presence at 3000 m in the eastern Himalayan area (Kosaka et al., 2010; Thapa et al., 2018). Though it had been seen in both undisturbed woods and grasslands, *Bidens pilosa* seems to favour disturbed environments (Khatri et al., 2022; Osaki et al., 2022). In the present research, these species were also found in the core, undisturbed forests of the EHSTWH and EHMCF. According to the findings, NF, AAPSEF, and EHMCF had higher levels of species richness. For this species, open forests with little canopy cover above had the highest RD and RA values. The species in question prefers open spaces to grow, and the research area's NF offers the right conditions for the species with higher canopy gap, which may account for the high values of the aforementioned characteristics in this forest. The invasion dynamics of *Bidens pilosa* in Himalayan ecosystems have not received much attention, despite the species' broad occurrence in the IHR.

Chromolaena odorata, often referred to as Siam weed, is one of the scrambling weeds (Lioger, 1997) that are said to be the most pervasive in South East Asia, the Pacific Islands, India, Australia, and central and western Africa (McFadyen, 2003). According to Lowe et al. (2000), this species is among the 100 worst invasive alien plants in the world because of its special traits, which include a high reproductive capacity, a dispersion mechanism, and the ability to compete in an environment that it had invaded successfully (Joshi, 2001). Roadsides, open spaces, and forest clearings are frequent places for this species to invade (Azmi, 2002), dry deciduous forests and inner shrub jungles and degraded forests in north-eastern India (Rao, 1977; Prashanthi and Kulkarni, 2005). The species reported in this research is widespread in all kinds of forests, with EHSTWH having the greatest RD and RA. In the present research, it is discovered that the species is among the most prevalent invaders and that it is urgent to address. Further evidence that some disturbance is necessary for these species' invasion comes from the observation of these species establishing on both motorable roads and accessible trails within the study region. Rao (1977) also showed that AAPSEF, NF, and RF had comparable species richness abundances and distributions in the eastern Himalayan region's lower belt. The species' current acceptable habitats are below 1500 metres; however, it is anticipated that its range will expand to include the eastern Himalayan area of the IHR (Lamsal et al., 2018). Furthermore, under future climate circumstances, Barik and Adhikari (2012) also emphasized that the eastern Himalayan area is more vulnerable to invasion by *Chromolaena odorata*, with a minor extension towards the north. Thus, the present research offers an evaluation of the distribution pattern and invasiveness of this species in various forest types.

According to Cronk and Fuller (1995), *Mikania micrantha* is considered the most infamous intruder. It is considered to be among the top 100 worst invasive alien species in the world (Lowe et al., 2000) and a major danger to biodiversity (Wang et al., 2004). Born in tropical America, the species had expanded to tropical forest zones in north-eastern regions (tropical rain forests, moist tropical or subtropical regions) and had become a threat to the eastern Himalayan Forest in recent years (Tripathi et al., 2012; Rameshprabu and Swamy, 2015). According to Li et al. (2013), the species' distribution is dependent upon the disturbances. The species, which climbs both small and large trees and totally covers their canopy, was seen more often in EHSTWH, RF, and NF. At lower attitude, Tripathi et al. (2012) saw this same pattern in northeastern India. The present research also found that there were significant RD and RA in EHSTWH and RF. Furthermore, *Mikania micrantha* had a high possibility of invading the eastern Himalayan region, according to Rameshprabu and Swamy (2015). According to present investigation, the extreme invasiveness of the species was observed in both wet forest regions and riverine habitats.

Native to the West Indies, *Solanum torvum*, sometimes referred to as turkey berry, had an invasive quality that had made it a successful colonizer (Welman, 2003). The plant is recognised to be an invasive weed of wooded regions, often along river banks, forest clearings, and other disturbed habitats (Welman, 2003; Francis, 2004). It had naturalized across the tropics and subtropics of the globe, reaching elevations of up to 2000 metres (CABI, 2024). Due to its many advantages, this species had grown to be valued in Indian and Chinese culture (Jain et al., 1986; An-ming and D'Arcy, 1986; Roddick, 1991). Comparing the NF to other forest types, the data demonstrate that the RD and RA are substantially more abundant. Even Nevertheless, as the research area's forest kinds go higher, the abundance falls. Furthermore, given these characteristics cannot exist under a closed forest canopy, the high levels of the parameters indicated in these forest types may be the result of open regions with substantial disturbance from human activity (Francis, 2004). According to earlier research, the NF area's high disturbance levels and proximity to rivers and roads provide ideal conditions for species to grow (Lim and Lim, 2013; Martina et al., 2021).

Solanum viarum is a fast-growing and aggressive invasive plant that is projected to spread to other tropical and subtropical regions throughout the globe due to its invasive trait (Nee, 1991). The species is often connected with human disturbances (Medal et al., 2012) and grows in the understory of rainforests, grasslands, and shrubby thicket dry forests (Nee, 1991). This plant is recognised as an important weed for a variety of traditional and modern purposes (CABI, 2024) and is found across India (GISIN, 2008). However, its invasion patterns within India have not been well studied. The study discovered that NF had the greatest RD and RA, followed by RF, AAPSEF, EHMCF, and RF. Recent research indicates that this species, which is mostly found in the eastern Himalayas at elevations below 2000 metres (Parker, 1992), encroaches on natural habitats such riverine regions and the borders of forests. Furthermore, it is anticipated to proliferate as an invasive species across the grasslands and forest regions of the whole eastern Himalayan region of Asia (Qi et al., 2023). The species' higher RD and RA values in NF also indicate its predilection for disturbed environments associated with human activity, since the forest types are quite close to the road. Furthermore, the present research found a similar pattern of invasion in lower foothill forest types in the Himalayan area to that described by Singh et al. (1998).

Synedrella nodilfora is an invasive species that originated in the West Indies but had since spread around the globe, particularly in tropical and subtropical regions (CABI, 2024). Holm et al. (1991) identified this species as a dangerous and aggressive pest in seven countries, and it is a frequent plant in gardens, disturbed areas, and wastelands. These species are widespread across disturbed tropical and subtropical regions and they exhibit resilience to a wide variety of environmental stressors (CABI, 2024). The species' invasiveness had not received much attention, but research had shown that its seeds can contaminate legume cover crops (Tasrif et al., 1991), serve as a host for insects, pathogens, and root knot nematodes (Kemp, 1998), and affect the local native population. In NF and RF, the species was common due to the high degree of disturbance in the study location. Nonetheless, many IHR studies have shown that they are present in the eastern and western Himalayan lower foothills (Shankar et al., 2012; Nayak et al., 2024). The species was common in NF and RF due to the high degree of disturbance in the study area. The greater RD and RA values of the species at NF and RF might be attributed to the continual interference of human activities and cattle, which provides free space for them to develop and multiply. Furthermore, many forest types are located near roadways and have seen significant disturbance.

As a competitive invasive shrub, *Triumfetta rhomboidea* is found throughout the tropics and subtropics of the world (Iqbal et al., 2010). It grows in a variety of disturbed habitats, including wastelands, pastures, open forest hillsides, forest clearings, woodland margins, and cultivated areas (CABI, 2024). The plant is endemic to tropical America; its competitive invasion of pastures and damaged wooded areas hinders the establishment of native species (Lay, 1950; Reddy, 2008). Consequently, it is classified as invasive in twelve countries, including India (CABI, 2024), where there is abundant (Motooka, 2003). In the present research, the species was more common in RF than it was in NF or other forest types. Both kinds of forests might have been greatly disturbed by human activity as well as by cattle. According to Verma and Joshi (2021), the western Himalayan area showed similar distribution trends. Furthermore, the species uses the disturbances (CABI, 2024) and goes to a new habitat (Wells et al., 1986; Valkenburg and Bunyapraphatsara, 2001; Raju and Rani, 2017), therefore invading a fresh territory. Since the forest type is more disturbed and closer to the road, the species seems to favour disturbance, as shown by the highest values of RD and RA for this species in RF. The present research also found that these species have begun to develop thick stands in various forest types, which is restricting the expansion of native populations as described by Motooko (2003). The findings of the present research support a different interpretation of McCormack's (2007) finding that these species have little effect in intact ecosystems.

Invaded most of the tropical and subtropical parts of the globe, *urena lobata* had been classed as an aggressive noxious weed (Randall, 2012) and fast-growing plant with capabilities of establishing dense patches and monospecific stand (Langeland et al., 2008). Furthermore, the plant may develop in damaged environments and replace the natural flora by means of thick stands (Francis, 2004). But little is known about the ecology of these species, which makes management of their spread difficult (Austin, 1999; Awan et al., 2014). In the present work, the species was present conspicuously in NF and EHMCF and in invasiveness; percentage values for AAPSEF, EHMCF, and RF were correspondingly high. With a presence of 0.5 to 0.75% of presence, these species are also projected to grow in the eastern Himalayan lower foothills (Ray et al., 2019). The greater value of RD and RA of the species in NF and EHMCF might be related to great disturbance in these forest types which have enabled to created dense population and creating a haven to proliferate more in the region. Francis (2004) also reported the species invasiveness in some habitats including disturbed forest, shrubland, riparian areas, forest margins, roadsides; similarly, the species invasion had been observed in the riverine areas as well as the temporary forest of the study area, where there is accessible paths or roadside.

4.7 Conclusion

The present study concludes that the five forest types of Pakke Tiger Reserve are invaded by invasive alien plant species and their proliferation are expanding rapidly. The patterns of IAPs distribution do not exhibit any trends but vary at every forest type as the kinds of the study region start from riverine area up to the summit of the mountain in the core forest (105-2040 m asl). These IAPs preferred the disturbed forest sites of AAPSEF, NF and RF due to increased anthropogenic activities. The sites are heavily invaded by *A. conyzoides, C. odorata* and *M. micrantha* in combines, along the motorable as well as trekking roads while *T. rhomboidea* and *U. lobata* are more near the riverine and non-forest areas. Although the higher elevation sites of EHMCF and EHSTWHF have fewer invasions in comparison to AAPSEF, NF and RF. this indicates that in higher altitude, IAPs seems to have less impacted due to non-disturbances of human and thick forest canopy cover as they are more in open areas. With these species, some native species are also growing near the invaded areas, this native species should be considered for replacing them after removal of IAPs. These native species should be taken into consideration while preparing management strategies as this could be the solution for controlling the invaded forested region of the study area. The findings also reveal the same tendencies as those of the worldwide research on the mountainous environment, namely declining of the IAPs along the higher elevation in forest types. The declining IAPs percentage in line with decreasing disturbance also implies that the major cause of IAPs' spread in the studied region is disturbance. Developing proper management strategies is the only option in the eastern Himalayan region to counter these alarming dangers.

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Chapter 5

Phenology of Selected Invasive Alien Plant species

5.0 Introduction

Phenology is the scientific study of the chronological occurrence of life cycle events associated with the seasons (Rathcke and Lacey, 1985; Morisette et al., 2009). The term phenology is originated from the Greek word “phaino”, which means to show or to appear and integrated predictor of species procedures to environmental events (IPCC, 2014). The pioneer of phenology as a discipline was Carl Linnaeus, who described the aims and approaches of phenological investigation in his book “*Philosophia Botanica*” in 1751. Plant phenology is the study of seasonal changes of life events such as flowering, fruiting, leaf initiation and leaf falls (Barve et al., 2009; de Beurs and Henebry, 2008), which contributes significantly to terrestrial ecosystems in production systems and acts as a primary climate change predictor (Ollerton and Lack, 1992; Penuelas and Filella, 2001; Fitter and Fitter, 2002; Cleland et al., 2007; Chuine, 2010; Cara Donna et al., 2014; Thackeray et al., 2016; Zettlemoyer et al., 2019). Phenological patterns have a major role in determining the seasonal movement of materials and energy between the atmosphere and the ground surface (Chuine et al., 2000). In addition, the patterns are considered accurate and sensitive markers of climate changes (Beaubien and Freeland, 2000; Sparks et al., 2009).

According to Cornelius et al. (2013) and Bucher and Romermann, (2020), phenological patterns show a substantial influence on the climatic conditions such as temperature, precipitation (rainfall), soil and moisture especially in the mountainous regions (Machado et al., 1997; Marques et al., 2004; Korner, 2007; Kopp et al., 2020). Due to complication of mountain topography, it further led to higher rate of beta diversity, species richness and high degree of turnover with altitudes (Elsen et al., 2018; Ahmad et al., 2020). These climatic variables are considered the most significant antecedent parameters of the environment that regulate plant phenology (Reeves and Coupland, 2000; Zhao et al., 2012). However, these climatic conditions do not directly induce or synchronize phenological phenomena (Hamann, 2004) but indirectly trigger phenological patterns in the higher latitude’s ecosystem (Souza and Funch, 2017). Therefore, understanding phenological patterns aids knowledge of the biological characteristics and regulatory aspects of several phenophases such as leaf fall, leaf initiation, flowering and fruiting (Kikim and Yadava, 2001). However, there is a lack of studies investigating phenological pattern or phenophases of invasive species in relation to climatic variables. Therefore, this strongly meaningful research was carried out to study phenological research in the region. The present study focused on the invasive alien plant species phenological patterns or phenophases in relation to climatic variables viz., temperature, precipitation and humidity in five different forest types of Pakke Tiger Reserve, Arunachal Pradesh. The research examined the whole annual cycle of the selected nine invasive alien plant species in response to climatic variables. The eastern Himalayan region of Pakke Tiger Reserve is relatively unexplored from the research perspective in terms of phenological events. Therefore, this region holds within immense riches which are yet to be completely uncovered. The present study investigated the effect of temperature, precipitation and humidity on the phenophases of nine invasive alien plant species along the 105 to 2050 m elevation in the eastern Himalayan region of Arunachal Pradesh.

5.1 Methodology

5.1.1 Study Area

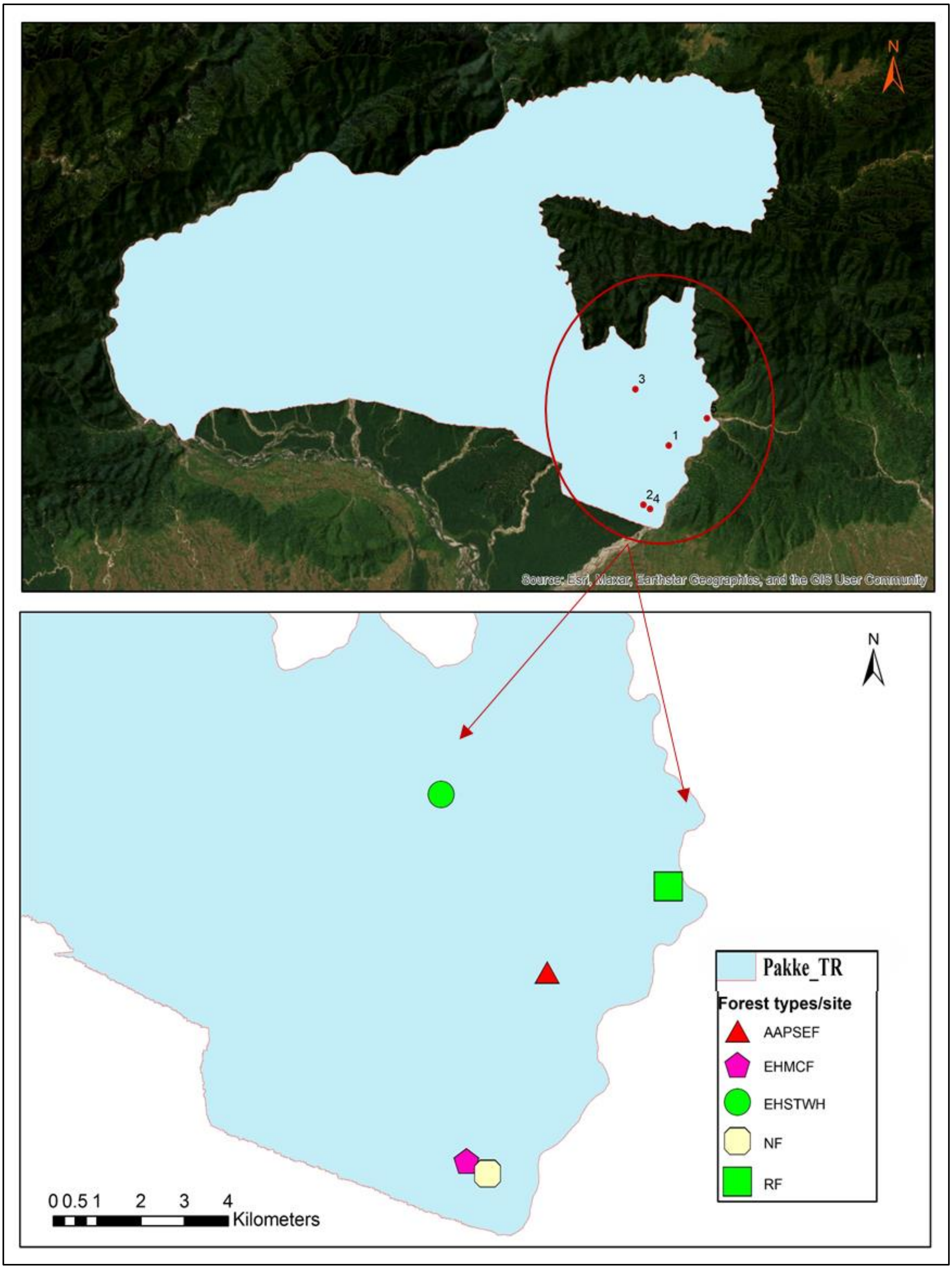
The study area, Pakke Tiger Reserve (PTR; $26^{o}55'$ to $27^{o}15'$E: $92^{o}35'$ to $93^{o}10'$N; Figure 5.1) lies in the western part of Arunachal Pradesh in Indian Himalayan region. The reserve covers an area of 861.95 km^2, with elevation ranging from 150 to 2050m above the sea level. Details descriptions of the study area, climatic conditions and selected species are given in chapter 3. The phenological observation was carried out for a period of one year during January 2021 to December 2021. The location of west bank is selected for the study because all forest sites i.e., Assam Alluvial Plain Semi Evergreen Forest (AAPSEF), East Himalayan Mixed Coniferous Forest (EHMCF), East Himalayan Subtropical Wet Hill Forest (EHSTWH), Non-Forest (NF) and Riverine Forest (RF; Champion and Seth, 1968; FSI, 2009) are located within a range of 50 km, with same or less macro-climatic condition for phenological observation (Table 5.1). In each of these five forest types, 400 m^2 (20X20m) plot was demarcated to study and monitored the major phenophases of the selected invasive species. All the selected invasive species were considered for the study because of their abundance and invasiveness after a month-long reconnaissance survey in 36 different locations across different forest types of Pakke Tiger Reserve. Monthly variation in precipitation, relative humidity and temperature (minimum and maximum) during the study period is given in fig. 5.2.

5.2 Selected Species for the Study

Nine invasive alien plant species namely, *Ageratum conyzoides* L., *Bidens pilosa* L., *Chromolaena odorata* (L.) R.M. King & H. Rob., *Mikania micrantha* Kunth and *Synedrella nodiflora* (L.) Gaertn. of Asteracceae family, *Solanum torvum* Sw. and *Solanum viarum* Dunal of Solanaceae family, *Triumfetta rhomboidea* Jacq. and *Urena lobata* L. of Malvaceae family were selected for the phenophases study. These are the most dominant invasive species common to five forest types i.e., Assam Alluvial Plain Semi Evergreen Forest (AAPSEF), East Himalayan Mixed Coniferous Forest (EHMCF), East Himalayan Subtropical Wet Hill Forest (EHSTWHF), Non-Forest (NF) and Riverine Forest (RF) of Pakke Tiger Reserve. Among these selected species five of them are herbaceous (*Ageratum conyzoides, Bidens pilosa, Mikania micrantha, Solanum viarum* and *Synedrella nodiflora*) while rest are shrubs (*Chromolaena odorata, Solanum torvum, Triumfetta rhomboidea* and *Urena lobata*).

5.3 Phenological Monitoring

To record the major phenophases (viz., flowering, fruiting, leaf initiation and fall) of the selected invasive species, four 5 x 5m plot for shrub (each for four shrub species) and five 1 x 1m plot for herbaceous (each for herb species) were laid in each forest types of Pakke Tiger Reserve. Thus, in total, twenty plots of shrubs and twenty-five plots of herbs across the five different forest types in the study transect. For shrub and herb species, phenological records were taken on the whole population of each species individually (Negi et al., 2022). All of the individuals growing inside the plot were identified as a ‘population’ for each species (Plos et al., 2024). The plots were laid in the area where maximum invasiveness was observed. The phenological patterns viz., flowering, fruiting, leaf initiation and fall were monitored visually following USA-NPN protocols (Denny et al., 2014).



Source: Esri, Maxar, Earthstar Geographics, and the GIS User Community

Figure 5.1: Map of the study area in Pakke Tiger Reserve showing phenological observation sites in five forest types. AAPSEF-Assam Alluvial Plain Semi Evergreen Forest, EHMCF -East Himalayan Mixed Coniferous Forest, EHSTWH-East Himalayan Subtropical Wet Hill Forest, NF-Non-Forest and RF-Riverine Forest.

The USA-NPN protocol, also known as status-based monitoring, is used in the present study because the selected invasive species are herbaceous and shrubs, not trees and observation are based on population. From these protocols, two phenophases from vegetative phenophases, one from reproductive phenophases and one phase from fruit or seed phenophases were considered for observation. A weekly population-level phenophases monitoring was conducted in accordance with the guidelines provided by Cornelius et al. (2011) and Panchen et al. (2014). The onset of a phenophases was defined as occurring in around 10% of the individual under observation and its peak was reached when it transpired in over 80% of the individuals (Ralhan, 1985; Negi, 1989; Thapa and Baburam, 2023). The specific phenophases was deemed to have culminated when the remaining 10% of individuals displayed it (Negi et al., 2022). These protocols are designed to quantify the onset, duration and intensity of plants phenophases (Denny et al., 2014) as well as status monitoring of events such as presence or absence of leaves, flower, fruit and initiation of plants during a repeated series of observation (Morellatoet al., 2010; Crimmins et al., 2013).

Table 5.1: Study site selected for phenophases observation in different forest types of Pakke Tiger Reserve.

Forest types/site	Latitude	Longitude	Elevation (m asl)	Distance to village (km)
AAPSEF (Mobuso)	26.987778	93.003444	243	1.8
EHMCF (West Bank)	26.950778	92.981222	234	1.6
EHSTWH (Laling)	27.025722	92.981778	857	6
NF (West Bank)	26.947812	92.98592	159	0.5
RF (Langka)	27.003278	93.033333	177	0.9

*****AAPSEF- Assam Alluvial Plain Semi Evergreen Forest, EHMCF- East Himalayan Mixed Coniferous Forest, EHSTWH- East Himalayan Subtropical Wet Hill Forest, NF- Non-Forest & RF- Riverine Forest.

5.4 Calculation and Statistical Analysis

For the calculation of phenophases, average of climatic parameters i.e., precipitation, relative humidity, minimum and maximum temperature (mean value of month) for the whole studied period was taken for convenience of data analyses. These averages exhibit a standard model of species development in the observation year (Orlandi et al., 2007), giving a clear-cut data for the analyses. Moreover, duration of each phenophases was counted in number of days and duration of months in the studied period for easy calculation. The variability of the phenological data was analysed by examining each phenophases of the invasive species over the course of a year in the study area. In order to determine the difference in each forest types, mean of the each phenophases with standard deviation (SD) in a month, ranges describing the duration of each phenophases in a month, duration giving the total number of days of each phenophases in each forest sites and coefficient of variation (CV) evaluating the degree of homogeneity of each phenophases in each species across the forest types of Pakke Tiger Reserve.

In addition, the mean value of the coefficient of variation across the different forest types explains the inter-annual variation of phenophases in the whole study area. Moreover, coefficient of variation was calculated following the standard formula (standard deviation/mean) and tabulated based on the mean values of each species phenophases, given in percentage. The CV defines the homogeneity if less or equal to 30% while more than 30% indicate heterogeneity in the phenophases. The phenological data of phenophases are recorded in binary data matrix (1= phenophases presence, 0= phenophases absent) against the months of a year i.e., January to December (Khan et al., 2018). The phenological stage of the species was calculated by adding the total number of phenophases stage of months in a year (Khan et al., 2015). Furthermore, variation of phenophases between the forest types was analysed by a repeated-measure analysis of variance (ANOVA) followed by Post-hoc Tukey pairwise test (Ronnenberg et al., 2011), if found significant differences.

Linear regression, correlation and canonical correspondence analysis (CCA) was employed to determine the association between climatic parameters and phenophases in the habitats affected by the invasive alien plant species. The phenophases of the selected invasive species were found similar across the different forest types and combine into one data set for all analyses following Dech and Nosko (2004). Moreover, canonical correspondence analysis was carried out to find the influence of environmental variables by comparing phenophases data with the meteorological matrix of precipitation, relative humidity, minimum and maximum temperature data (Hegazy et al., 2012). These analyses were based on the annual mean values discovered for each species. The relationship between phenophases and climatic parameters was analysed using Pearson correlation (Orlandi et al., 2007). Also, linear regression analysis was used to determine the relationship between the climatic variables and phenophases in which climatic variables were used as independent variables while phenophases as dependent variables

(Lou et al., 2007: Ullah et al., 2022). The Pearson correlation coefficient (R^2) was determined for linear regression for the present study. All the descriptive statistics, correlation, regression, canonical correspondence analysis (CCA) and analysis of variance (ANOVA) were performed using Microsoft excel and PAST software version, 5.0.2 (Hammer et al., 2001).

5.5 Results

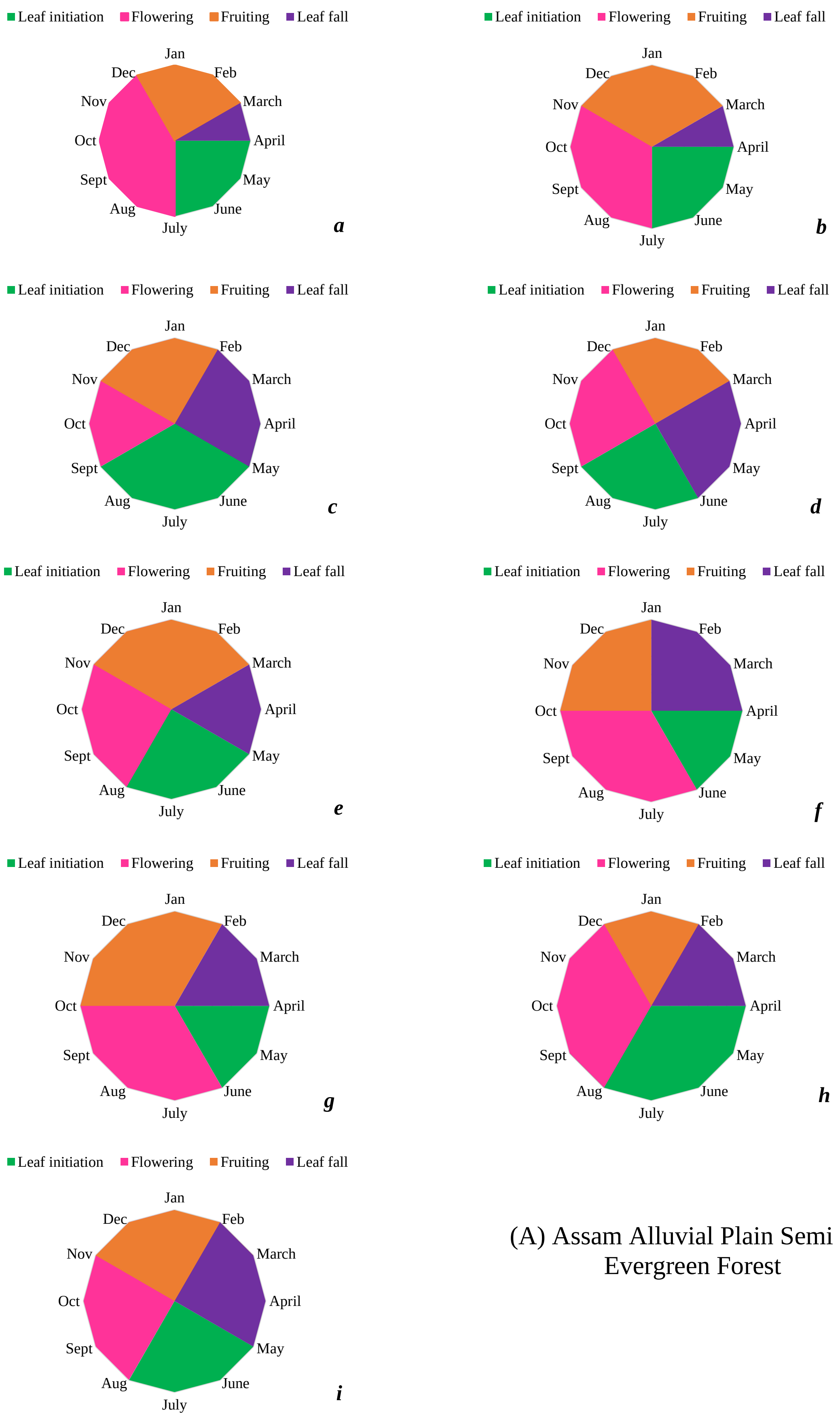
The location of the phenological data carried out in five different forest types i.e., Assam Alluvial Plain Semi Evergreen Forest, East Himalayan Mixed Coniferous Forest, East Himalayan Sub-Tropical Wet Hill, Non-Forest and Riverine Forest of Pakke Tiger Reserve during the phenophases observation are summarized in the Table 5.1. The phenograms of nine selected species during the study period in each forest types are shown in Fig. 5.2. The phenograms clearly display the different phenophases in each month of the year and difference in phenophases in different forest types of Pakke Tiger Reserve.

5.5.1 Phenophases

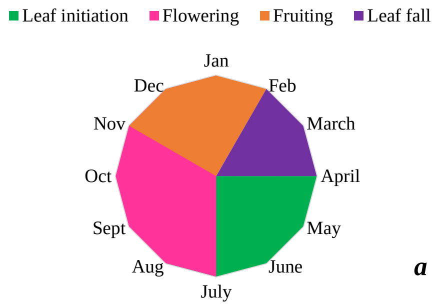
In the study area, phenophases (Flowering, Fruiting, Leaf fall and Leaf initiation) were observed in order to understand the effect of climatic parameters (Precipitation, Relative Humidity, Minimum and Maximum Temperature) during the one-year study period. The study revealed that the meteorological variables have certain influences on the phenophases of these invasive alien species in the study region. Moreover, some of the invasive species exhibited early phenophases in comparison among the different forest types in study area (Fig. 5.2).

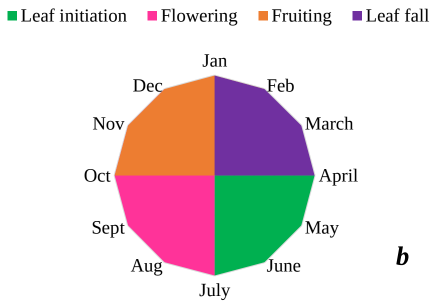
5.5.1.1 Leaf Initiation

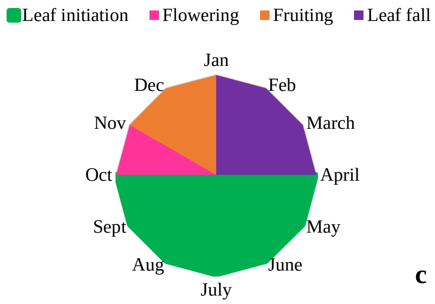
The present study shows that *Ageratum conyzoides* had an average coefficient of variation value of 61.4% indicating their asynchronous leaf initiation across the study area (Table 5.2). The highest coefficient of variation was observed in EHSTWH (67.7%) and lowest in NF (35.9%). The initiation period was longest in NF (12-31 days) and shortest in both EHMCF and RF (03-31 days) respectively (Table 5.2). Moreover, their duration was highest in EHMCF (108±12.9 days) and lowest in EHSTWH (59±13.3 days) with a mean average of 85±12.5 days in a year. Leaf initiation continued for a long period starting from late April to early August of the study period.

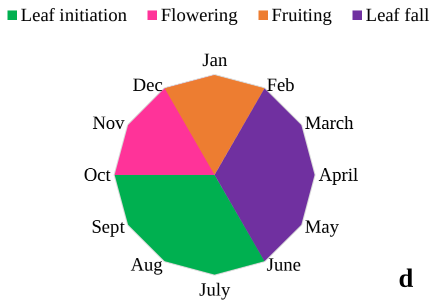


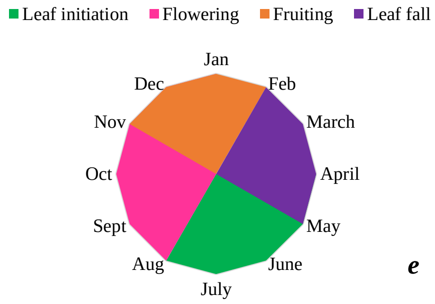
(A) Assam Alluvial Plain Semi Evergreen Forest

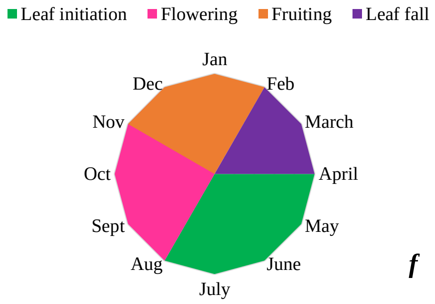


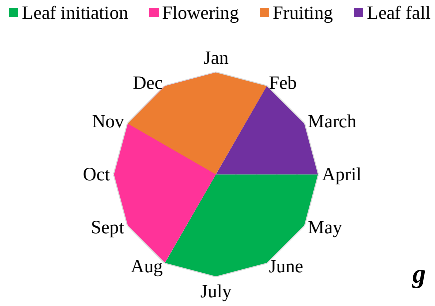


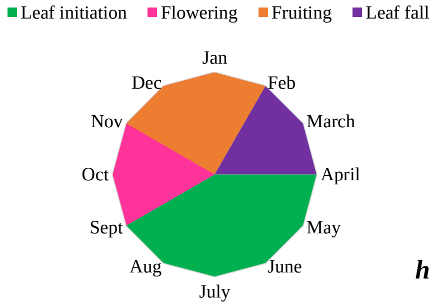


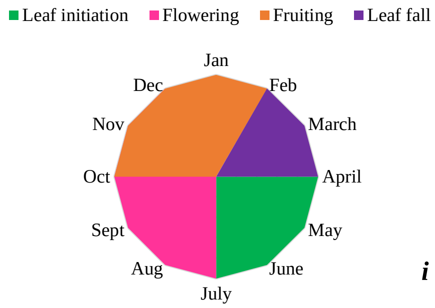




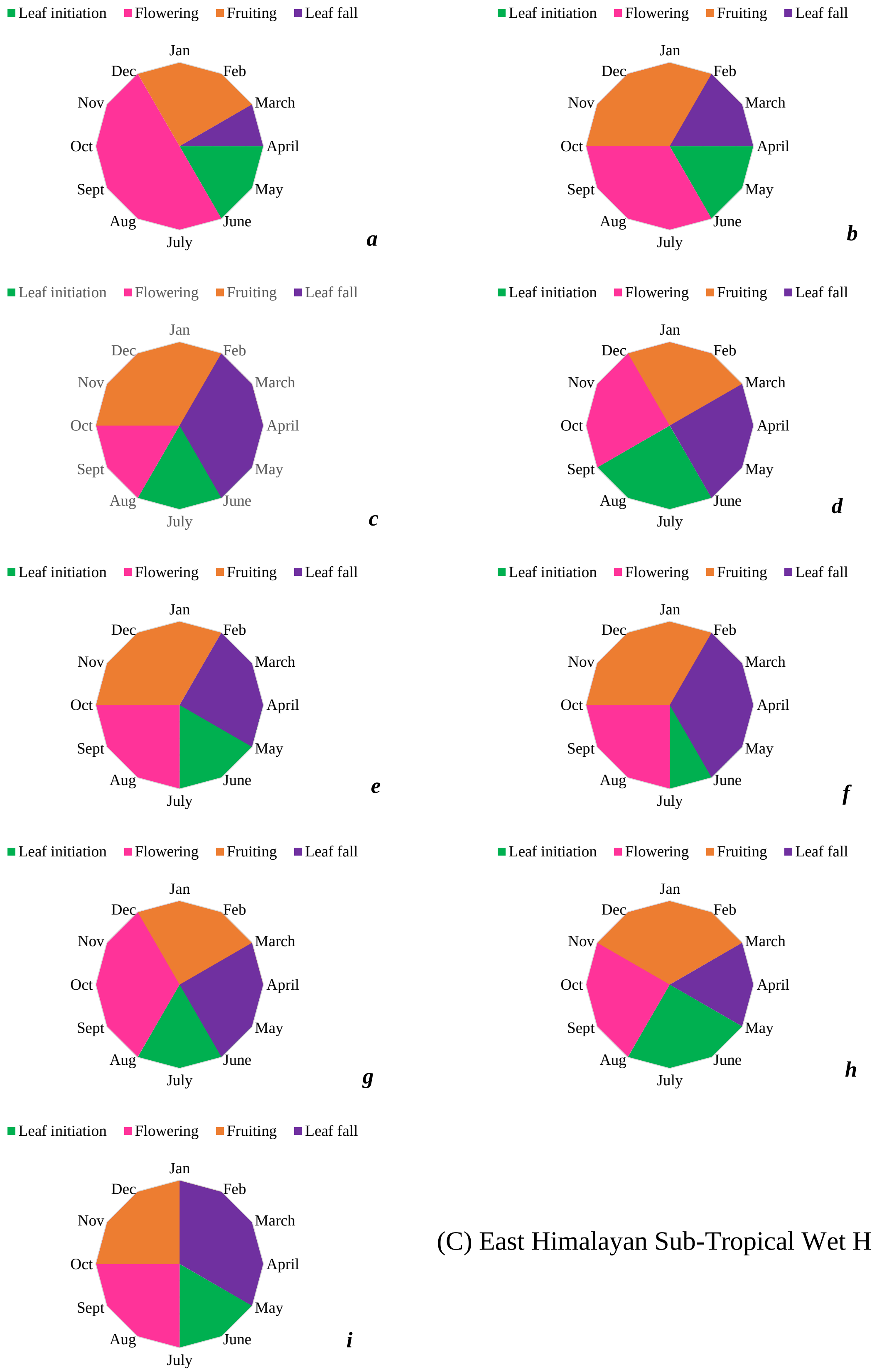




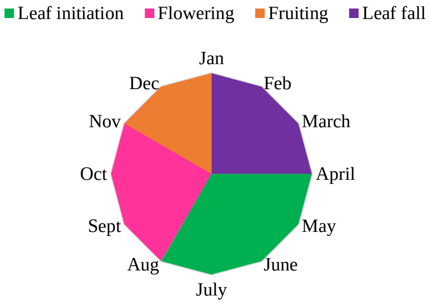




(B) East Himalayan Mixed Coniferous Forest



(C) East Himalayan Sub-Tropical Wet Hill



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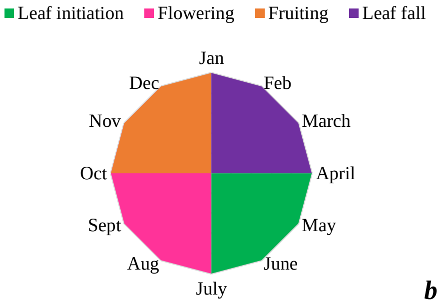

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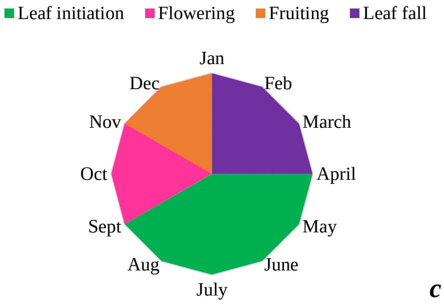

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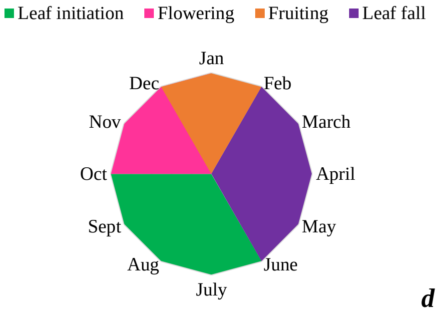

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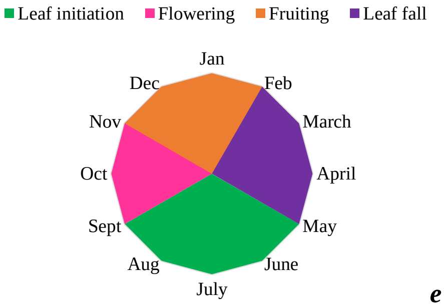

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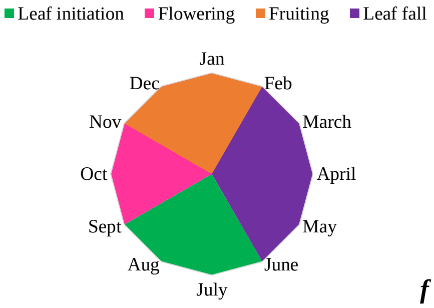

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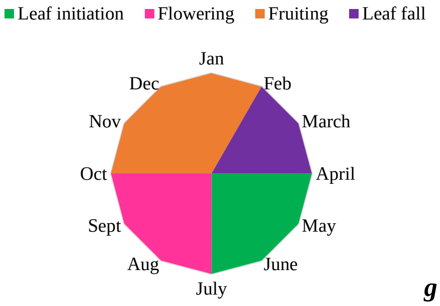

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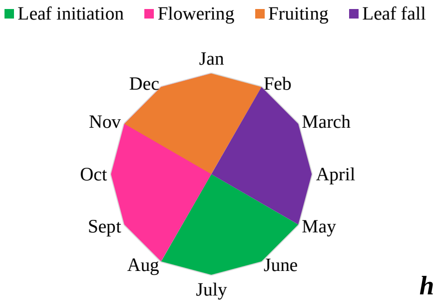

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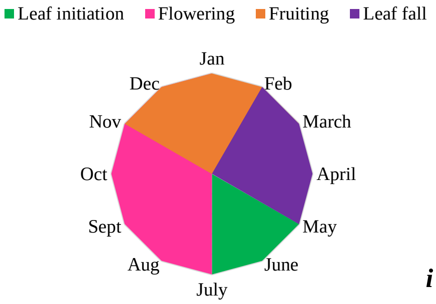
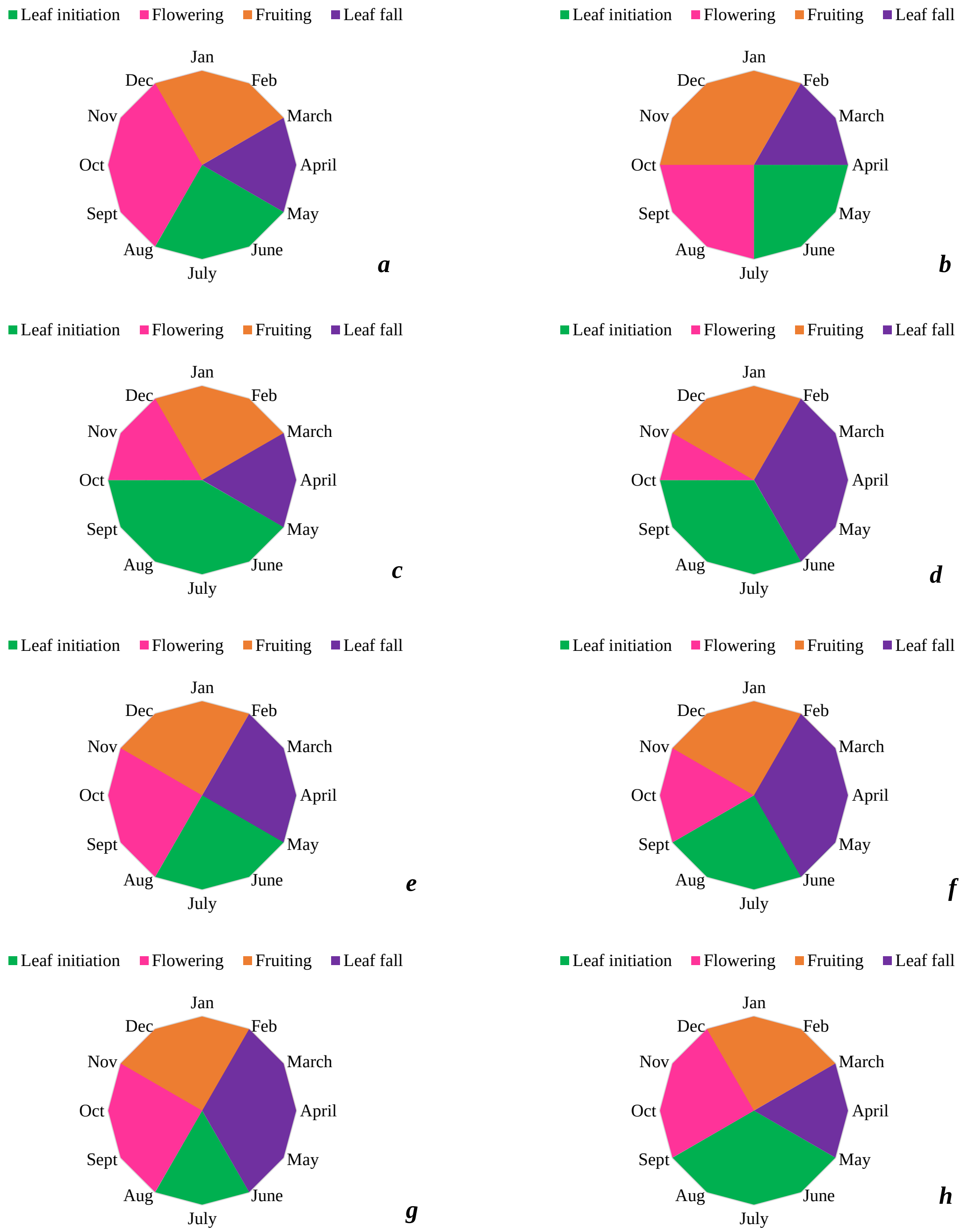
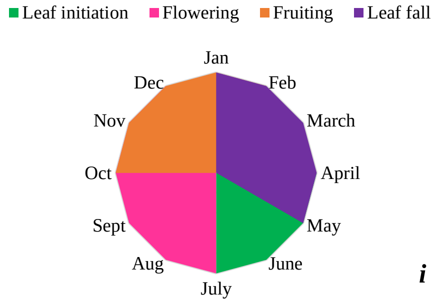


i

(D) Non- Forest





(E) Riverine Forest

Figure 5.2: Details phenograms of selected nine invasive alien plant species across the five forest types in study area- (A) Assam Alluvial Plain Semi Evergreen Forest, (B) East Himalayan Mixed Coniferous Forest (C) East Himalayan Sub-Tropical Wet Hill, (D) Non-Forest and (E) Riverine Forest. * (a) *Ageratum conyzoides*, (b) *Bidens pilosa*, (c) *Chromolaena odorata*, (d) *M. Micrantha*, (e) *Solanum torvum, (f) Solanum viarum, (g) Synedrella nodiflora, (h) Triumfetta rhomboidea*, and *(i) Urena lobata.*

In contrast, leaf initiation of *Bidens pilosa* lasted from late April to early July of the study period (Fig. 5.2), with the EHMCF (108±12.9days) having the longest duration while the lowest duration was found in EHSTWH (59±13.3 days). The average duration of *Bidens pilosa* was 79.6±13.3 days in a studied period. Moreover, range of period in a month was highest in AAPSEF, EHSTWH and NF (05-31 days) and lowest was in EHMCF and RF (03-31 days) during the study period. Further, coefficient of variation was found highest in NF (68.7%) and the lowest was in RF (8%) with an average of 52.4% indicating their asynchronous leaf initiation in the study sites.

However, coefficient of variation of *Chromolaena odorata* was found maximum in RF (86%) and minimum in EHMCF (31%) and NF (31%) having an average of 54.5% indicating asynchronous degree of phenology across the sites. Leaf initiation range in a month was highest in NF (12-31 days) and lowest in (03-31 days) in the study region (Table 5.2). Moreover, the longest duration was found in EHMCF (165±12.9 days) and NF (165±8.3 days) having an average of 109.8±12.4 days in a year. These species phenology start from late April to early October of the study period. However, phenology of *Mikania micrantha* starts from late June to early October during the study period (Fig. 5.2). Duration of these phenophases were found highest in NF (152±0.4 days) and lowest in AAPSEF (109±2.8 days) with an average of 125.4±2.4 days in a year. The length of these phenophases during a month was higher in NF (29-31 days) and lower in EHMCF (11-31 days) in the study area (Table 5.2). Additionally, coefficient of variation was found maximum in EHMCF (39.08%) and minimum in NF (2.9%). The average value of these species was 19.8% across the forest types denoting homogenous degree of the phenophases in the study area.

In contrast, the average coefficient of variation of *Solanum torvum* (47.2%) was found to be diverse and different from each forest types as their initiation start from late April to early August. The maximum coefficient of variation was observed in EHSTWH (50%) and minimum in EHMCF (16.8%) in the study area (Fig 5.2). The highest and lowest range of these phenophases was observed in EHMCF (20-31 days) and RF (05-31 days) respectively. Moreover, duration of initiation phenology was highest in EHMCF (143±11.2 days) and minimum in NF (69±9.1 days) while their average was 89.8±10.8 days in the studied period.

However, *Solanum viarum* average duration (98.4±8.4 days) was greater than *Solanum torvum* in the studied year while their maximum and minimum was found in NF (131±10.1 days) and EHSTWH (61±14 days) respectively. During the month, the longest phenophases was observed in RF (28-31 days) and EHSTWH (04-31 days) as their initiation started from late April to early August in the study area. Moreover, the coefficient of variation was highest in EHSTWH (69.9%) and lowest in RF (5.7%) across the study area (Table 5.2). Likewise, the phenophase of *Solanum torvum*, show similar degree of heterogeneity with *Solanum viarum* (35.5%) in the entire study area. In the study area, *Synedrella nodiflora* had an average duration of 93.6±10.3 days in the whole period with maximum in AAPSEF (125±8.3 days) and minimum in EHSTWH (72±11.2 days; Fig. 5.2).

This species phenophases started from late April to early September (Fig. 5.2) and range of phenology during the month was 12-31 days (AAPSEF) to 05-31 days (RF). In addition, this species average coefficient of variation was 43.6% which describe the degree of heterogeneity across the study area. The maximum and minimum coefficient of variation was found in RF (52.8%) and AAPSEF (33.5%) in the study region.

In the present study, *Triumfetta rhomboidea* phenophases start from late April to early September (Fig. 5.2) and their ranges of phenophases during a month was 22-31 days (RF) to 04-31 days (AAPSEF) in the study region. The average phenophases duration was 127.4±8.04 days during the study period with RF (173±3.5 days) having the maximum and NF (75±9.5 days), the minimum duration. In addition, the maximum coefficient of variation was observed in AAPSEF (53%) and minimum in RF (12.2%) while their average was 33 % which clearly surpassed the degree of homogeneity across the study area (Table 5.2). The descriptive statistic of *Urena lobata* clearly shows that the range in a month was highest in AAPSEF as well as RF (26-31 days) and lowest in EHMCF (02-31 days). Moreover, average duration during the whole year was 112.4±6.3% with RF (179±1.9 days) having the longest duration and EHSTWH (81±4.3 days), the shortest duration. *Urena lobata* initiate their phenology from late April to early July (Fig. 5.2) and their average coefficient of variation (29.2%) describes the synchronous phenophases across the study area. The maximum coefficient of variation was observed in EHMCF (64%) and the minimum was found in RF (6.5%) in the present study (Table 5.2).

Table 5.2. Descriptive statistics of leaf initiation phenology across the forest types in Pakke Tiger Reserve.

	Forest	Descriptive Statisitcs	*AC*	*BP*	*CO*	*MM*	*SN*	*ST*	*SV*	*TR*	*UL*
Leaf initiation	AAPSEF	Mean± SD	21±12.1	21±12.1	21±12	27.2±2.8	25±8.3	23.7±11.2	26.3±6.4	23±12	28±2.1
		Range	5-31	5-31	5-31	19-31	12-31	7-31	19-31	4-31	26-31
		CV%	58	58.06	58	20.8	33.5	47.5	24.14	53	7.52
		Duration	84	84	84	109	125	95	79	138	144
	EHMCF	Mean± SD	21.6±12.9	21.6±12.9	21.6±12.9	24±4.1	23.4±10	28.6±4.8	26.2±10.1	23.8±10.7	21±137
		Range	3-31	3-31	5-31	11-31	11-31	20-31	8-31	9-31	2-31
		CV%	59.7	59.7	31	39.08	46.9	16.8	38.8	45	64
		Duration	108	108	165	120	72	143	131	143	84
	EHSTWH	Mean± SD	19.66±13.3	19±13.3	21.6±14	30±0.7	24±11.2	23.6±11.8	20.3±14	27±4.5	27±4.3
		Range	5-31	5-31	5-31	28-31	11-31	10-31	4-30	21-31	22-30
		CV%	67.7	67.7	66.7	4.71	46.9	50.05	69.9	16.8	16.14
		Duration	59	59	65	120	72	71	61	108	81
	NF	Mean± SD	26±9.3	19.2±13.2	27±8.3	30.4±0.4	25.7±9.8	23±9.1	26.2±10.1	25±9.5	18.5±9.6
		Range	12-31	5-31	12-31	29-31	11-31	13-31	8-31	14-31	8-31
		CV%	35.9	68.7	31	2.94	38.2	39.8	38.8	38.15	52.3
		Duration	104	77	135	152	103	69	131	75	74
	RF	Mean± SD	17.5±15	17.5±15	17±15	25.2±3.5	24±12.6	17.7±14.7	30±1.7	28.8±3.5	29.8±1.9
		Range	3-31	3-31	3-31	13-31	5-31	5-31	28-31	22-31	26-31
		CV%	86	8	86	31.7	52.8	82	5.7	12.29	6.5
		Duration	70	70	70	126	96	71	90	173	179

* *AC- Ageratum conyzoides, BP- Bidens pilosa, CO- Chromolaena odorata, MM- M. Micrantha, ST- Solanum torvum, SV- Solanum viarum, SN- Synedrella nodiflora, TR- Triumfetta rhomboidea, UL- Urena lobata* and LI- Leaf Initiation

5.5.1.2 Flowering

The duration of flowering phenophases in all the selected species are different in each forest types of Pakke Tiger Reserve. However, some of the species have synchronous flowering phase in the same months of the year. This might have been due to influence of climatic parameter on the particular species. All the selected species i.e., *Ageratum conyzoides, Bidens pilosa, Chromolaena odorata, Mikania micrantha, Solanum torvum, Solanum viarum, Synedrella nodiflora, Triumfetta rhomboidea* and *Urena lobata* have different flowering phase in the study area. Moreover, duration of the flowering phenophases in each forest was quite different in all selected species in each forest types (Table 5.3). The duration of *A. conyzoides* flowering ranges from 211±7.9 days (AAPSEF) to 112±9 days (NF) with an average mean of 165.8±9.1 days in the study area (Table 5.3). Regarding variability, the range of coefficient of variation (CV) was found highest in EHSTWH (47.8%) followed by EHMCF (37.7%), AAPSEF (30.2%), RF (19.3%) and NF (11.3%) with an average of 29.2% indicating the degree of homogeneity in flowering phenophases across the forest types. Moreover, the numbers of the flowering days in a month were longest in NF (24-31 days) while the least was observed in EHSTWH (5-31 days). In addition, flowering phenophases of *A. conyzoides* was detected from the month of July to February while the peak period was observed from early August to late November in all the forest types of the study area (Table 5.3). Moreover, coefficient of variation was highest in EHSTWH (52.1%) followed by AAPSEF (43.4%), EHMCF (26.8%), NF (22.2%) and RF (19.2%) respectively. The average value of coefficient of variation was 32.7 indicate the degree of heterogeneity in their phenophases with the longest number of flowerings was in RF (17-31 days) and the least was observed in EHSTWH (01-31 days).

In addition, *Chromolaena odorata* has a maximum flowering length of 154±7.9 days (EHSTWH) and the minimum were 76±4.5 days (EHMCF) with an average of 102.4±7.1 days across the different forest types. The synchronous peak flowering was observed from the month of October to December in all forest types (Fig. 5.2). The longest number of flowerings in a month was found in AAPSEF (23-31 days) while the least was RF (7-31 days) and the coefficient of variation was found highest in RF (48.2%) followed by NF (32.6%), EHSTWH (30.9%), EHMCF (17.7%) and AAPSEF (12.7%), respectively. Moreover, the average value of CV (28.4%) suggests synchronous flowering across the study area. In contrast, coefficient of variation of *Mikania micrantha* has an average value of 55.2% which indicate asynchronous degree of flowering across the forest types. The highest coefficient of variation was found in EHMCF (66%) and the least was NF (34%) in the study area. Moreover, duration of flowering was longest in AAPSEF (80±6.4 days) while the shortest was RF (66±8 days) with an average of 84±6.2 days in Pakke Tiger Reserve. The number of flowerings was highest in NF (15-31 days) followed by RF (6-31 days) and the shortest was in EHSTWH (03-31 days). The peak blooming time of *Mikania micrantha* was observed from November to December while the initiation started in October lasted till January in the study region (Fig. 5.2). Similarly, the peak flowering time of *Bidens pilosa* was also observed from early August to November with the phenophases starting from the month of July to January (Fig. 5.2). The length of flowering duration ranged from 219±12.6 days (EHSTWH) to 159±7.1 (NF) with an average of 156.6±8.6 days across the forest types (Table 5.3).

The species, *Solanum torvum* displayed degree of homogeneity with an average coefficient of variation value of 27.1% indicating synchronous flowering across the study area. The highest coefficient of variation was observed in EHMCF (54.5%) and smallest coefficient of variation was found in NF (5.3%) representing their degree of flowering phenophases. The flowering days in a month was highest in AAPSEF (26-31 days) while the least was found in EHMCF (04-31 days) Moreover, longest flowering duration was observed in AAPSEF (175±4.5 days) and the shortest was found in NF (93±5.8 days) with an average value of 125±7.8 days across the studied region. The peak blooming occurred from October to December though their initiation started lately in July tilled early January (Fig. 5.2).

However, *Solanum viarum* does not display degree of homogeneity as its average value (35.6%) exceeded the accepted range of coefficient of variation. The highest coefficient of variation was found in EHSTWH (54.2%) while the least was observed in EHMCF (6.04%). Duration of flowering was longest in EHMCF (176±1.9 days) and shortest duration was in RF (131±7 days) with an average of 147.8±8.5 days. In addition, EHMCF (26-31 days) have the longest flowering period in a month while the shortest period was in EHSTWH (02-31days). Moreover, the peak flowering was observed from September to November while their initiation was started from late in June and lasted till early January (Fig. 5.2).

Table 5.3. Descriptive statistics of flowering phenology across the forest types in Pakke Tiger Reserve.

	Forest	**Descriptive Statistics**	***AC***	***BP***	***CO***	***MM***	***SN***	***ST***	***SV***	***TR***	***UL***
Flowering	AAPSEF	Mean± SD	26.37±7.96	24.8±12.1	28.2±3.5	20±6.4	28.6±2.9	29±4.5	23.1±11	27±6.2	27.1±5.2
		Range	13-31	5-31	23-31	4-31	24-31	26-31	5-31	15-31	19-31
		CV%	30.19	43.4	12.7	64	10.2	7.32	51.3	23.3	19.39
		Duration	211	199	113	80	172	175	139	162	163
	EHMCF	Mean± SD	25.28±9.5	26.5±7.1	25.3±4.5	19.7±6.5	24±10	22.±12.2	29.3±1.9	22.6±11	25.4±10
		Range	7-31	13-31	21-30	4-31	10-31	4-31	26-31	5-31	4-31
		CV%	37.7	26.8	17.7	66	42	54.5	6.04	51.8	39.7
		Duration	117	159	76	79	144	136	176	113	178
	EHSTWH	Mean± SD	24.37±11.6	24±12.6	25.6±7.9	24±5.3	25±8.8	24±10.1	22.8±12.3	24.1±10	27±6.2
		Range	5-31	1-31	14-31	3-31	10-31	7-31	2-31	5-31	16-31
		CV%	47.8	52.1	30.9	50	35.3	42.3	54.2	44	23.12
		Duration	195	219	154	120	125	120	137	145	135
	NF	Mean± SD	28±3.1	27±6	25.6±8.3	25±5	24.5±7.8	23.2±5.8	26±10.3	28.8±4.3	27.42±5.5
		Range	24-31	16-31	19-31	15-31	14-31	6-31	5-31	21-31	18-31
		CV%	11.29	22.2	32.6	34	32	50.3	39.6	15.21	20.06
		Duration	112	162	77	75	98	93	156	144	192
	RF	Mean± SD	27.71±5.3	27.7±5.3	23±11.1	22±8	26.75±4.4	25.2±6.5	26.2±7	24.2±9.6	28.1±5.4
		Range	17-31	17-31	7-31	6-30	22-31	17-31	14-31	9-31	16-31
		CV%	19.28	19.2	48.2	62	16.54	25.9	27.04	39.9	19.4
		Duration	194	194	92	66	107	101	131	121	197

* *AC- Ageratum conyzoides, BP- Bidens pilosa, CO- Chromolaena odorata, MM- M. Micrantha, ST- Solanum torvum, SV- Solanum viarum, SN- Synedrella nodiflora, TR- Triumfetta rhomboidea, UL- Urena lobata* and FL- Flowering.

In contrast to *Solanum viarum*, coefficient of variation of *Synedrella nodiflora* (27.2%) indicated a homogenous degree of flowering across the forest types with the maximum in EHSTWH (35.3%) and the minimum in AAPSEF (10.2%). Flowering duration was longest in AAPSEF (172±10.2 days) and shortest in NF (98±7.8 days) and have an average of 129.2±6.78 days in the study region. Additionally, the longest and shortest flowering period in a month was observed in AAPSEF (24-31 days) and both in EHMCF and EHSTWH (10-31 days). The peak flowering period was observed from September to December while their blossoming start in late August and finished their flowering period in early January (Fig. 5.2).

Triumfetta rhomboidea flowering duration was longest in AAPSEF (162±6.2 days) and shortest in EHMCF (113±11 days) with an average of 137±6.4 days across the study area. These species flowering period in month was longest in NF (21-31 days) and shortest in both EHCMF and EHSTWH (05-31 days). In case of variability, coefficient of variation (34.8%) was found to be asynchronous as the maximum coefficient of variation was found in EHMCF (51.8%) and the minimum in NF (15.2%), respectively. The flowering of these species started in late August continued till early January while its peak was observed in September to late November in the study area (Fig. 5.2).

Lastly, *Urena lobata* (24.3%) show synchronous flowering degree in all the study sites where the maximum and minimum coefficient of variation were found in EHCMF (39.7%) and AAPSEF (19.3%). These species have the longest flowering duration in RF (197±5.4 days) and the shortest in EHSTWH (135±6.2 days) and have an average of 173±6.4 days in a year. During the months, the longest and shortest period of flowering were found in AAPSEF (19-31 days) and EHMCF (05-31 days). *Urena lobata* started to flower in late July and continued till early January, with its peak session in between August- November in the study period (Fig. 5.2).

5.5.1.3 Fruiting

After flowering, the most important stage in a plant life is fruiting. In the present study, fruiting of selected invasive species were more or less similar in all the forest types and these might be due to influence of climatic parameters or biotic factors (Table 5.4). The fruiting phenology of *Ageratum conyzoides* across the study sites was different as their duration was longest in AAPSEF (107±3.4 days) and shortest in NF (67±9 days) with an average of 90.8±8.04 days in a year. Moreover, average coefficient of variation was 36.5%

which indicate their different fruiting phenophases across the forest types. Their fruiting period in a month was highest in AAPSEF (24-31 days) while the least was EHMCF (01-31 days). These species started fruiting phenophases from late November to early March and its peak period was between Decembers to January (Fig. 5.2). In the present study, duration of *Bidens pilosa* fruiting was longest in RF (99±7.4 days) and lowest was observed in EHSTWH (74±6 days) with a mean average value of 82.6±8.3 days in a study period.

During the month, the longest fruiting period was observed in EHSTWH (19-31 days) while the shortest was in EHMCF (01-31 days) across the forest types. However, the average coefficient of variation was 50% suggesting diverse fruiting phenology in all the study sites. The fruiting of these species was started in late October and lasted up to early March of the study period (Fig. 5.2).

Similarly, fruiting phenology of *Chromolaena odorata* also start from late October though it lasted up to early April, little longer than *Bidens pilosa* (Fig. 5.2). The duration of fruiting was found longest in EHSTWH (172±10.3 days) and shortest in NF (74±7 days) while their average duration of fruiting in the study period was 104.4±8.6 days. Moreover, longest flowering range in a month was observed in NF (17-31 days) and the lowest in EHSTWH (04-31 days) and average coefficient of variation was 37.06% indicating degree of heterogeneity in this phenology. The highest coefficient of variation was found in AAPSEF (49%) and the least was in NF (28%) in the study area (Table 5.4).

However, *Mikania micrantha* average coefficient of variation was 25.5% indicating the degree of homogeneity in fruiting across the study area while NF (33.9%) have the maximum and EHMCF (12.8%) the least value (Table 5.4). Moreover, the duration of fruiting was found highest in RF (107±3.3 days) and lowest in NF (67±4.3 days) and have an average fruiting of 85.8±3.4 days in a year. The range of fruiting in a month was higher in EHMCF (24-31 days) and lowest in EHSTWH (15-31 days). Their phenology starts from late November to early March of the study period.

However, *Solanum torvum* phenology start from late October to early March with maximum duration in EHSTWH (157±6.6 days) and minimum in EHMCF (86±11.4 days) in the study area. These species average duration during the study period was 115.2±8.3 days.

Table 5.4. Descriptive statistics of fruiting phenology across the forest types in Pakke Tiger Reserve.

	Forest	Descriptive Statisitcs	*AC*	*BP*	*CO*	*MM*	*SN*	*ST*	*SV*	*TR*	*UL*
Fruiting	AAPSEF	Mean± SD	26.75±3.4	20.5±12	20.7±10.1	25.3±3.8	24.6±6.3	24.7±3.8	24.4±8.6	27.5±2.9	24±8.5
		Range	24-31	4-31	12-31	18-31	17-31	17-31	14-31	24-31	12-31
		CV%	12.7	59.1	49	26.2	24.6	27.6	35.3	10.9	35.7
		Duration	107	82	83	76	128	99	122	109	120
	EHMCF	Mean± SD	19±14.6	19±14	23.5±8.6	28±2	24.6±5.5	21.5±11.4	21.5±12.3	25±8.4	21.2±11
		Range	1-31	1-31	16-31	24-31	21-31	8-31	5-31	13-31	6-31
		CV%	77.35	77.3	36	12.8	22.3	53.2	57.5	33.9	55.3
		Duration	76	76	94	84	74	86	86	100	85
	EHSTWH	Mean± SD	26.25±5.8	24±6	24.5±10.3	23.7±3.5	22±12.7	26±6.6	27.2±3.3	28.7±2.6	26.2±5.8
		Range	19-31	19-31	4-31	15-31	4-31	14-31	23-31	26-31	19-31
		CV%	22.2	24.4	42.2	30.2	57.8	25.5	12.12	9.14	22.2
		Duration	105	74	172	95	88	157	109	115	105
	NF	Mean± SD	22.33±9	205±12.1	24.6±7	22.3±4.3	25.4±7.1	22.8±11.6	26±6.2	25±9.5	26.2±5.8
		Range	13-31	6-31	17-31	17-31	14-31	3-31	18-31	11-31	17-31
		CV%	40.3	59.1	28	33.9	28.1	51.1	24.12	38.08	22.4
		Duration	67	82	74	67	127	137	104	100	131
	RF	Mean± SD	24.75±7.4	24.7±7.4	24.7±7.4	26.7±3.3	21.2±11	24.2±8.3	25±6.9	20.6±11	25.6±6.8
		Range	14-31	4-31	14-31	17-31	9-31	14-31	19-31	9-31	16-31
		CV%	30.1	30.1	30.1	24.8	53.8	34.2	27.12	53.5	26.6
		Duration	99	99	99	107	85	97	100	62	154

* *AC- Ageratum conyzoides, BP- Bidens pilosa, CO- Chromolaena odorata, MM- M. Micrantha, ST- Solanum torvum, SV- Solanum viarum, SN- Synedrella nodiflora, TR- Triumfetta rhomboidea, UL- Urena lobata* and FR- Fruiting.

Moreover, the highest coefficient of variation was observed in EHMCF (53.2%) and least in EHSTWH (25.5%). Across the study area, the average coefficient of variation was 41% which define the diverse and different phenophases in each forest types during the study period and the range of phenophases was higher in AAPSEF (17-31 days) and lower in NF (03-31 days).

Similarly, *Solanum viarum* also has a diverse and different phenophases across the study sites as the average coefficient of variation (31.2%) was higher than 30%. The coefficient of variation was maximum in EHMCF (57.5%) and minimum in EHSTWH (12.1%) in the study area. Alternatively, the range of these phenophases was highest in EHSTWH (23-31 days) and lowest in EHMCF (5-31 days) as their phenology was found similar with *Solanum torvum* (October-March). The species phenophases duration was longest in AAPSEF (122±8.6 days) while the shortest was observed in EHMCF (86±12.3 days) and their average phenophases in the year was 104.2±7.4 days. Likewise, *Synedrella nodiflora* phenology also had same fruiting months (October-March) as *Solanum viarum* and average duration during the whole period was 100.4±8.5 days. The longest duration was observed in AAPSEF (128±6.3 days) and shortest in EHMCF (74±5.5 days) in the study area. Moreover, the range of the phenophases was highest in EHMCF (21-31 days) and EHSTWH (04-31 days) with an average coefficient of variation of 37.3% across the study sites describing their diverse and distinct phenology in each forest types. The maximum coefficient of variation was found in RF (53.8%) and minimum was in EHMCF (22.3%) in the study region. In the study area, *Triumfetta rhomboidea* had the longest duration in EHSTWH (115±2.6 days) and shortest in RF (62±11 days) while their average was 97.2±6.8 days in the whole study period. These species phenology started from late November to early March in the study region and had the range from 26-31 days (EHSTWH) to 09-31 days (RF). Additionally, their average coefficient of variation was 29.1% which describe the degree of homogeneity in their phenology across the study sites. The maximum and minimum coefficient of variation was observed in RF (53.5%) and EHSTWH (9.1%) respectively. The phenology of *Urena lobata* started from late October to early March and had the range from 19-31 days (EHSTWH) to 06-31 days (EHMCF) in the study period. The average duration of these species was 119±7.5 days in the whole year across the study sites while the longest duration was in RF (154±6.8 days) and the shortest was EHMCF (85±11 days). In addition, the maximum coefficient of variation was found in

EHMCF (55.3%) and minimum was in EHSTWH (22.2%) while their average was 32.4% which follows the degree of heterogeneity across the forest types.

5.5.1.4 Leaf Fall

In the present study, *Ageratum conyzoides* duration of leaf fall was maximum in NF (82±11.3 days) and minimum in AAPSEF (40±1.4 days) in a whole year (Table 5.5). The average mean value of leaf fall across the forest types was 63.2±6.2 days in a year. The length of leaf fall duration in a month was longest in EHSTWH (24-31 days) and shortest in NF (06-31 days). In addition, the average coefficient of variation was 28.05% suggesting their synchronous leaf fall across the study area. In the study area, the highest coefficient of variation was found in NF (55.5%) and smallest was in AAPSEF (7.07%) and the species started its leaf fall from late February and lasted up to early April across the forest types. Similar pattern of leaf fall (late February to early April) was also observed in *Bidens pilosa* in the study area (Fig. 5.2). Leaf fall duration was found maximum in NF (95±7.1 days) and minimum in AAPSEF (40±1.4 days) while their average value was 67.8±6.6 days in the study period (Table 5.5). However, their range of leaf fall during a month was longest in AAPSEF (19-31 days) while the lowest was in EHSTWH (13-31 days). Moreover, average coefficient of variation was 30% across the forest types of the study describing their synchronous phenology in all the study sites. The highest coefficient of variation was found highest in EHSTWH (52.3%) followed by RF (31.7%), NF (30.2%) and EHMCF (28.8%) while the lowest was in AAPSEF (7.07%), respectively. However, the leaf fall of *Chromolaena odorata* started early January and finished in early May across the study sites (Fig. 5.2) and longest phenophases happened in EHSTWH (91±10 days) and RF (91±0.5 days) while the shortest was observed in AAPSEF (40±1.4 days) in the study area (Table 5.5). Additionally, the average duration was 72.4±5.6 days across the different forest types. Leaf fall ranges during a month was found highest in EHMCF (22-31 days) and lowest in NF (06-31 days) with an average coefficient of variation value of 26.2% describing their degree of homogeneity across the study sites. The highest coefficient of variation was observed in EHMCF (44.8%) while the least was observed in RF (1.09%) in the study area. However, leaf fall duration of *Mikania micrantha* was found highest in EHMCF (126±3.4 days) and lowest in EHSTWH (67±8.1 days) in the study area.

Their average duration in a year across the forest types was 100.6±5.1 days (Table 5.5). Moreover, the maximum and minimum coefficient of variation was found in EHSTWH (63.3%) and EHMCF (30.2%) with an average value of 43.9% in the entire studied region.

This value describes their asynchronous nature and diverse phenology pattern across the forest types. The range of leaf fall was found highest in EHMCF (15-31 days) and lowest in EHSTWH (06-31 days) as phenophases start from late February to early June of the study period (Fig. 5.2). However, the range of *Solanum torvum* was highest in AAPSEF (17-31 days) and the lowest in NF (02-31 days) as phenology start from late February to early June of the study period (Fig. 5.2). The phenophase duration of these species was longest in RF (124±9.8 days) and shortest in EHMCF (76±6.5 days) as their average was 96±10 days in a year (Table 5.5).

In addition, average coefficient of variation was 59.9% indicating their degree of heterogeneity in these species' phenology. The NF (77.4%) had the higher coefficient of variation and the AAPSEF (27.4%) had the lowest value in the study area. Equally, the average coefficient of variation of *Solanum viarum* (53.7%) also indicates the diverse phenophases in the study area. The maximum and minimum coefficient of variation was found in RF (72%) and AAPSEF (15.1%) respectively (Table 5.5). Moreover, the range of these phenophases was highest in AAPSEF (23-31 days) and lowest in NF as well as RF (01-31 days) with their phenology occurred between late February and early May during the study period (Fig. 5.2). The duration of these species was found maximum in RF (101±14.5 days) and minimum in EHSTWH (61±14.2 days) while their average duration in whole year was 73.4±10.7 days. Similarly, the phenology of *Synedrella nodiflora* also start from late February to early May with the longest and shorter range of phenophases was observed in AAPSEF (15-31 days) and NF (02-31 days) in the study area (Table 5.5).

The average coefficient of variation (50.6%) indicates different phenology and diverse nature in each forest types. The highest and lowest coefficient of variation was observed in NF (67.9%) and EHMCF (31.1%) in the study area. Moreover, the longest phenophases duration was observed in AAPSEF (125±8.3 days) and shortest duration in RF (44.4±12 days) having an average of 86.4±10.9 days in the whole study period. The phenology of *Triumfetta rhomboidea* start from late February to early May (Fig. 5.2) and their phenophases range was highest in RF (30-31 days) and lowest in EHSTWH (04-30 days) during the month (Table 5.5). The average duration of these species was 74.4±8.2 days in whole year with the longest in RF (92±0.5 days) and shortest in EHMCF (61±10 days) in the study region.

Table 5.5. Descriptive statistics of leaf fall phenology across the forest types in Pakke Tiger Reserve.

	Forest	Descriptive Statisitcs	*AC*	*BP*	*CO*	*MM*	*SN*	*ST*	*SV*	*TR*	*UL*
Leaf Fall	**AAPSEF**	Mean± SD	20±1.4	20±1.4	20±1.4	23±4.4	24.6±8.3	24±6.5	26.6±4	25±3.3	20.2±12
		Range	19-21	19-21	19-21	13-31	15-31	17-31	23-31	21-31	4-30
		CV%	7.07	7.07	7.07	38.7	33.5	27.4	15.15	20.25	61.4
		Duration	40	40	40	92	125	96	80	76	81
	EHMCF	Mean± SD	24.33±7	25±7.2	26.3±4.5	25.2±3.4	25±7.7	19±6.1	24±6	20.3±10	22±11.5
		Range	17-31	17-31	22-31	15-31	14-31	7-31	20-31	11-31	9-31
		CV%	28.8	28.8	17.1	30.2	31.15	64.4	25.3	49.5	52
		Duration	73	75	79	126	100	76	72	61	66
	EHSTWH	Mean± SD	26.5±3.5	19.3±10.1	22.7±10	22.3±8.1	20.7±13	21.5±12.5	20.3±14.2	21±14.7	21±15
		Range	24-29	13-31	9-31	6-31	3-31	4-31	4-30	4-30	3-30
		CV%	13.3	52.3	44.8	63.3	62.7	58.1	69.9	70.14	74
		Duration	53	58	91	67	83	86	61	63	63
	NF	Mean± SD	20.5±11.3	23.7±7.1	19.5±11.9	25.5±5.2	20±13.5	19.6±15.1	17.6±15.2	20±12.9	21.2±12.9
		Range	6-31	15-31	6-31	9-31	2-31	2-31	1-31	4-31	3-31
		CV%	55.5	30.2	61	42.9	67.9	77.4	86.4	64.6	61
		Duration	82	95	78	101	80	98	53	80	85
	RF	Mean± SD	22.33±8	23.6±7.5	30.3±0.5	23.4±4.6	22±12	24.8±9.8	20.2±14.5	30.6±0.5	29.2±2.1
		Range	14-30	16-31	30-31	8-31	13-31	8-31	1-31	30-31	26-31
		CV%	35.6	31.7	1.09	44.6	57.8	39.6	72.02	1.88	7.4
		Duration	68	71	91	117	44	124	101	92	146

** AC- Ageratum conyzoides, BP- Bidens pilosa, CO- Chromolaena odorata, MM- M. Micrantha, ST- Solanum torvum, SV- Solanum viarum, SN- Synedrella nodiflora, TR- Triumfetta rhomboidea, UL- Urena lobata* and LF- Leaf fall.

The coefficient of variation was found highest and lowest in EHSTWH (70.1%) and RF (1.8%) while their average was 41.2% describing their degree of heterogeneity across the forest types. In the study of *Urena lobata* phenology, the range of phenophases during the month was highest in RF (26-31 days) (Table 5.5) and lowest in EHSTWH (03-30 days) as their phenophases initiated start from late January to early May (Fig. 5.2). Moreover, the average duration of these species was 88.2±10.7 days in the studied period with RF (146±2.1 days) having the maximum and EHSTWH (63±15 days), the minimum during the study. The coefficient of variation was maximum in AAPSEF (61.4%) and minimum in RF (7.4%) while their average was 51.1% which indicate the diverse and distinct phenophases in each forest types.

5.5.2 Variation (ANOVA) Between Phenophases

5.5.2.1 Leaf Initiation

In the leaf initiation phenology, significant differences were observed in *Chromolaena odorata* (0.04: $P<0.05$) while the rest of invasive alien plant species (*Ageratum conyzoides, Bidens pilosa, Mikania micrantha, Solanum torvum, Solanum viarum, Synedrella nodiflora, Triumfetta rhomboidea* and *Urena lobata*) displays no variation ($P<0.05$) indicating their similar occurrence of phenophases in the study sites (Table 5.8). This shows that *Chromolaena odorata* leaf initiation during the study period was different from the remaining species phenology (Table 5.6). According to Post-hoc Tukey's analysis, phenology of *Chromolaena odorata* in EHSTWH and EHMCF were found different from the rest of the forest types in Pakke Tiger Reserve (Table 5.7). This shows that there was distinct phenology pattern among the species in the study sites.

Table 5.6. Variation of *Chromolaena odorata* leaf initiation phenophases among the forest types in Pakke Tiger Reserve.

ANOVA	**Sum of square**	**df**	**Mean square**	**F**	***p (same)***
Between groups	0.766	4	0.191	2.608	**0.04**
Within groups	14.16	55	0.257		
Error	3.233	44	0.073		
Between subject	10.93	11	0.993		
Total	14.93	59			

Table 5.7. Post-hoc Tukey Pairwise Test of *Chromolaena odorata* phenophases among the forest types in Pakke Tiger Reserve.

Post Hoc Tukey Pairwise Test	**AAPSEF**	**EHMCF**	**EHSTWH**	**NF**	**RF**
AAPSEF		0.94	0.17	1	1
EHMCF	1.06		**0.03**	0.94	0.94
EHSTWH	3.19	4.26		0.17	0.17
NF	0	1.06	3.19		1
RF	0	1.06	3.19	0	

Table 5.8. Variation of remaining invasive alien plant species leaf initiation phenophases among the forest types in Pakke Tiger Reserve.

Species	**F**	***p* (same)**
Ageratum conyzoides	1.238	0.3
Bidens pilosa	0.6044	0.66
Mikania micrantha	1.692	0.16
Solanum torvum	0.6962	0.59
Solanum viarum	0.7857	0.54
Synedrella nodiflora	0.4074	0.80
Triumfetta rhomboidea	1.723	0.16
Urena lobata	1.453	0.23

5.5.2.2 Flowering

In the present study, the variation of flowering phenophases was measured in order to find the distinct phenophases among the forest types. The variation was checked in all the selected invasive alien plant species flowering phenology to understand their significant phenology difference in species between the forest types of Pakke Tiger Reserve. All the species (*Bidens pilosa, Chromolaena odorata, Mikania micrantha, Solanum torvum, Solanum viarum, Synedrella nodiflora, Triumfetta rhomboidea* and *Urena lobata*) have similar phenophases i.e., no significant differences (P>0.05) among the study sites (Table 5.11), except for *Ageratum conyzoides* (0.02: P<0.05) which differ significantly among the forest types in the study region (Table 5.9). By using Post-hoc Tukey pairwise test, it was clearly understandable that *A. conyzoides* phenophases in NF was quite different from the phenology of AAPSEF and EHSTWH in the study area (Table 5.10).

Table 5.9. Variation of *A. conyzoides* flowering phenophases among the forest types in Pakke Tiger Reserve.

ANOVA	Sum of square	df	Mean square	F	*p* (same)
Between groups	0.9	4	0.22	3.194	**0.02**
Within groups	13.83	55	0.25		
Error	3.1	44	0.070		
Between subjects	10.73	11	0.97		
Total	14.73	59			

Table 5.10. Post-Hoc Tukey Pairwise Test of *A. conyzoides* phenophases among the forest types in Pakke Tiger Reserve.

Post-hoc Tukey Pairwise Test	AAPSEF	EHMCF	EHSTWH	NF	RF
AAPSEF		0.9	1	**0.02**	0.93
EHMCF	1.08		0.93	0.16	1
EHSTWH	0	1.08		**0.02**	0.93
NF	4.35	3.26	4.35		0.16
RF	1.08	0	1.08	3.26	

5.5.2.3 Fruiting

In the study area, there were no significant differences in fruiting phenophases (*P*>0.05) among the species in different forest types of Pakke Tiger Reserve (Table 5.12). All the selected invasive alien plant species species (*Ageratum conyzoides, Bidens pilosa, Chromolaena odorata, Mikania micrantha, Solanum torvum, Solanum viarum, Synedrella nodiflora, Triumfetta rhomboidea* and *Urena lobata*) shows similar phenology and no difference was observed in the study area. These clearly indicate that the fruiting phenophases in all the forest types occur at the same time during the same month of the year.

Table 5.11. Variation of remaining invasive alien plant species flowering phenophases among the forest types in Pakke Tiger Reserve.

Species	**F**	***p* (same)**
Bidens pilosa	2.2	0.08
Chromolaena odorata	2.391	0.06
Mikania micrantha	1.185	0.33
Solanum torvum	1.0	0.41
Solanum viarum	0.3143	0.8
Synedrella nodiflora	1.453	0.2
Triumfetta rhomboidea	1.0	0.4
Urena lobata	0.8209	0.5

Table 5.12. Variation of all the selected invasive alien plant species fruiting phenophases among the forest types in Pakke Tiger Reserve.

Species	**F**	***p* (same)**
Ageratum conyzoides	0.3548	0.83
Bidens pilosa	0.3143	0.86
Chromolaena odorata	2.321	0.07
Mikania micrantha	0.7333	0.57
Solanum torvum	0.8652	0.49
Solanum viarum	0.1864	0.94
Synedrella nodiflora	0.5789	0.67
Triumfetta rhomboidea	0.3143	0.86
Urena lobata	0	1.0

5.5.2.4 Leaf Fall

The result of ANOVA analysis shows that there was no variation in the leaf fall phenology among the species in different forest types of Pakke Tiger Reserve (Table 5.13). During the same month of the year, leaf fall phenophases occurrence were similar in the selected invasive alien plant species (*Ageratum conyzoides, Bidens pilosa, Chromolaena odorata, Mikania micrantha, Solanum torvum, Solanum viarum, Synedrella nodiflora, Triumfetta rhomboidea* and *Urena lobata*) and there was no significant difference (P>0.05) in their phenology.

Table 5.13. Variation of all the selected invasive alien plant species leaf fall phenophases among the forest types in Pakke Tiger Reserve.

Species	F	*p* (same)
Ageratum conyzoides	0.4162	0.79
Bidens pilosa	1.453	0.23
Chromolaena odorata	0.234	0.91
Mikania micrantha	2.2	0.08
Solanum torvum	0.8652	0.49
Solanum viarum	1.158	0.34
Synedrella nodiflora	1.0	0.41
Triumfetta rhomboidea	0.3548	0.83
Urena lobata	1.453	0.23

5.5.3 Influence of Phenophases by Climatic Parameters

The climatic parameters such as maximum temperature, minimum temperature, precipitation and relative humidity have certain effect on the species phenology. In the present study, these climatic parameters effects on the phenophases of selected invasive alien plant species were measured using linear regression, correlation and canonical correspondence analysis (CCA) to understand their influences and relationship on their phenology.

5.5.3.1 Linear Regression Between the Climatic Parameters and Phenophases

To test the relationship between the environmental variables and phenophases phases, linear regression analyses were studied for the selected species. Moreover, linear regression indicates that the climatic parameters indices that were best related with specific phenophases differed with invasive alien plant species and phenophases. In terms of data interpretation, all the selected species shows excellent significances in relations of R^2 and slopes.

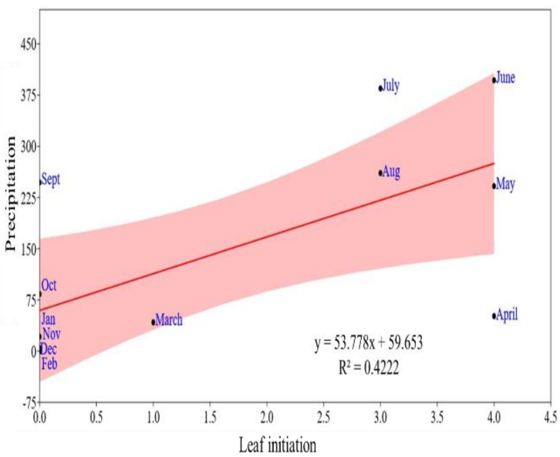
5.5.3.2 Leaf Initiation and Climatic Parameters

The results of the present study indicate that precipitation was the main factor for early initiation of leaf in the study area (Table 5.14). *Ageratum conyzoides, Chromolaena odorata, Mikania micrantha, Solanum torvum, Solanum viarum, Synedrella nodiflora, Triumfetta rhomboidea* and *Urena lobata* were influenced by the precipitation (Fig. 5.3) while *Chromolaena odorata, Mikania micrantha, Solanum torvum, Solanum viarum,* *Synedrella nodiflora* and *Triumfetta rhomboidea* were influenced by minimum temperature of the study area (Fig. 5.4). Furthermore, *Ageratum conyzoides, Chromolaena odorata, Solanum torvum, Solanum viarum, Synedrella nodiflora* and *Triumfetta rhomboidea* were found significant ($P<0.05$) with maximum temperature (Fig. 5.5) while *Mikania micrantha* was only significantly related with relative humidity of Pakke Tiger Reserve (Fig. 5.6). All the species were related with either of the climatic parameters except *Bidens pilosa* who does not have any relation with the environmental variables.

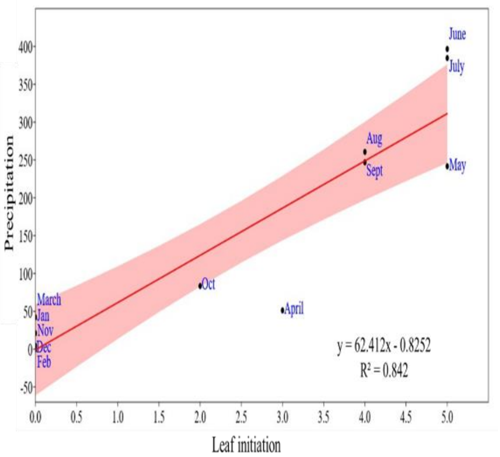
Table 5.14. Correlation coefficient (R^2) and slopes for linear regression describing the relationship of leaf initiation phenophases and climatic parameters for the selected invasive alien plant species in Pakke Tiger Reserve during the study period.

	Leaf initiation							
Species	**PreC**		**RelH**		**MinT**		**MaxT**	
	R^2	*Slope*	R^2	*Slope*	R^2	*Slope*	R^2	*Slope*
Ageratum conyzoides	0.422	**53.778***	0.0002	-0.078	0.358	1.831	0.472	**1.639***
Bidens pilosa	0.111	26.443	0.132	-1.68	0.07	0.787	0.293	1.2385
Chromolaena odorata	0.841	**62.412***	0.18	1.678	0.86	**2.3397***	0.668	**1.603***
Mikania micrantha	0.706	**54.683***	0.538	**2.777***	0.760	**2.100***	0.270	0.976
Solanum torvum	0.783	**71.541***	0.077	1.303	0.624	**2.364***	0.423	**1.516***
Solanum viarum	0.601	**59.471***	0.028	0.743	0.433	**1.867***	0.359	**1.325***
Synedrella nodiflora	0.724	**56.714***	0.087	1.143	0.611	**1.928***	0.450	**1.289***
Triumfetta rhomboidea	0.844	**62.343***	0.161	1.583	0.731	**2.147***	0.445	**1.304***
Urena lobata	0.514	**51.694***	0.015	0.517	0.308	1.481	0.248	1.035

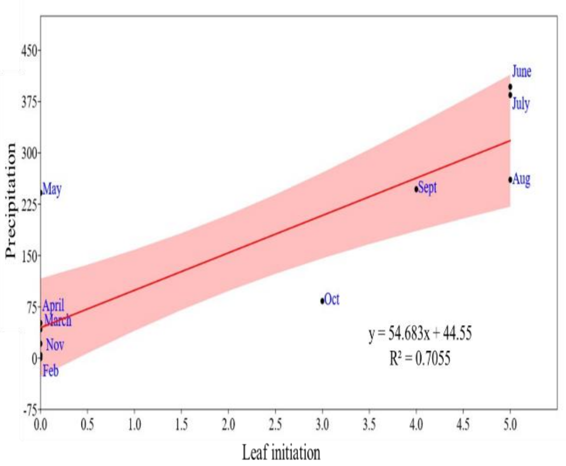
*****-Significant relationship, PreC- Precipitation, RelH- Relative Humidity, MinT- Minimum Temperature, MaxT- Maximum Temperature


Precipitation
450
375
300
225
150
75
0
-75
July
June
Aug
May
Sept
Oct
Jan
Nov
Dec
Feb
March
April
y = 53.778x + 59.653
R² = 0.4222
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5
4.0
4.5
Leaf initiation

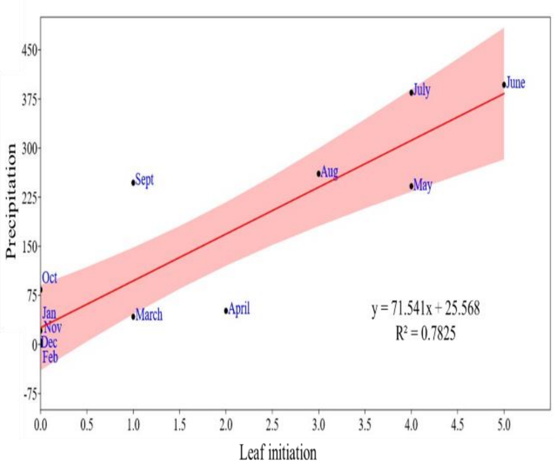
A


Precipitation
400
350
300
250
200
150
100
50
0
-50
June
July
Aug
Sept
May
Oct
April
March
Jan
Nov
Dec
Feb
y = 62.412x - 0.8252
R² = 0.842
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5
4.0
4.5
5.0
Leaf initiation

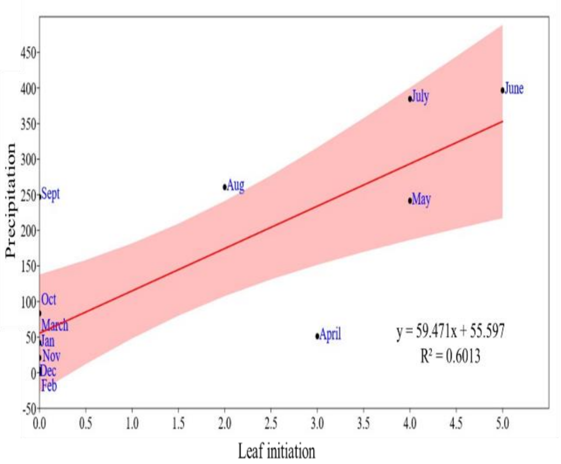
B


Precipitation
450
375
300
225
150
75
0
-75
June
July
Aug
Sept
May
Oct
April
March
Nov
Feb
y = 54.683x + 44.55
R² = 0.7055
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5
4.0
4.5
5.0
Leaf initiation

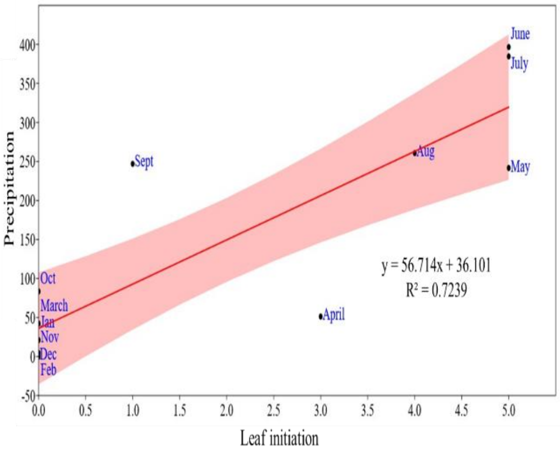
C


Precipitation
450
375
300
225
150
75
0
-75
June
July
Aug
May
Sept
Oct
Jan
Nov
Dec
Feb
March
April
y = 71.541x + 25.568
R² = 0.7825
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5
4.0
4.5
5.0
Leaf initiation

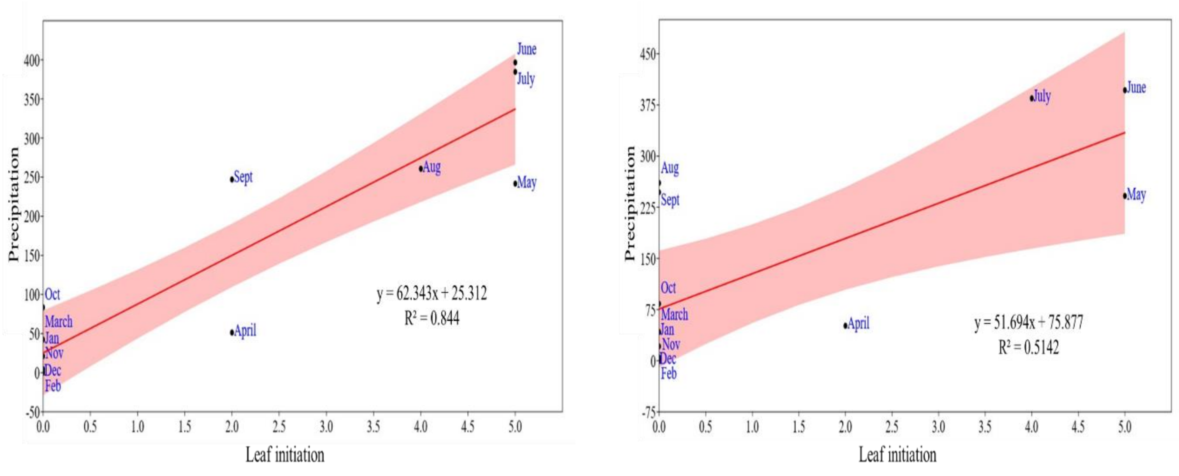
D


Precipitation
450
400
350
300
250
200
150
100
50
0
-50
June
July
Aug
May
Sept
Oct
March
Jan
Nov
Dec
Feb
April
y = 59.471x + 55.597
R² = 0.6013
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5
4.0
4.5
5.0
Leaf initiation

E

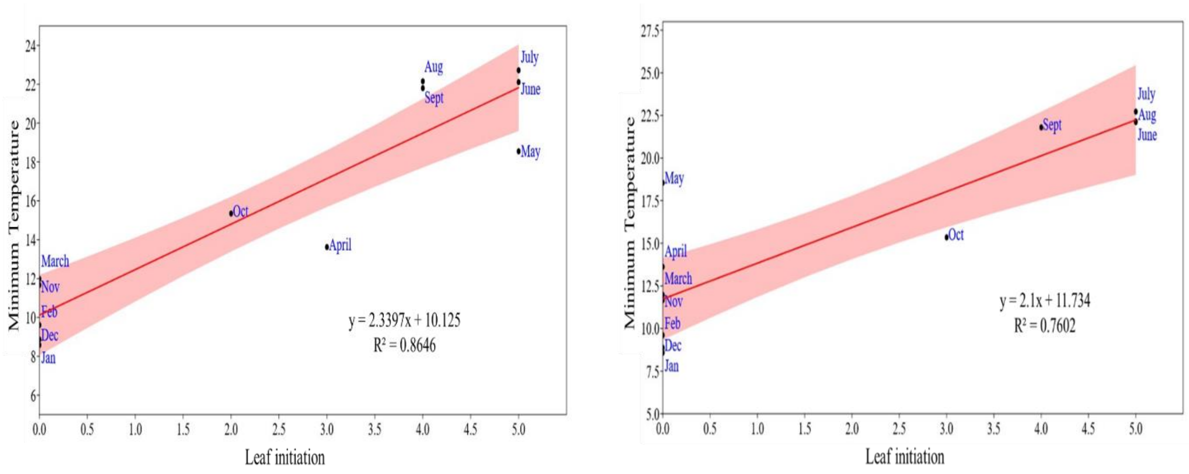

Precipitation
400
350
300
250
200
150
100
50
0
-50
June
July
Aug
May
Sept
Oct
March
Jan
Nov
Dec
Feb
April
y = 56.714x + 36.101
R² = 0.7239
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5
4.0
4.5
5.0
Leaf initiation

F

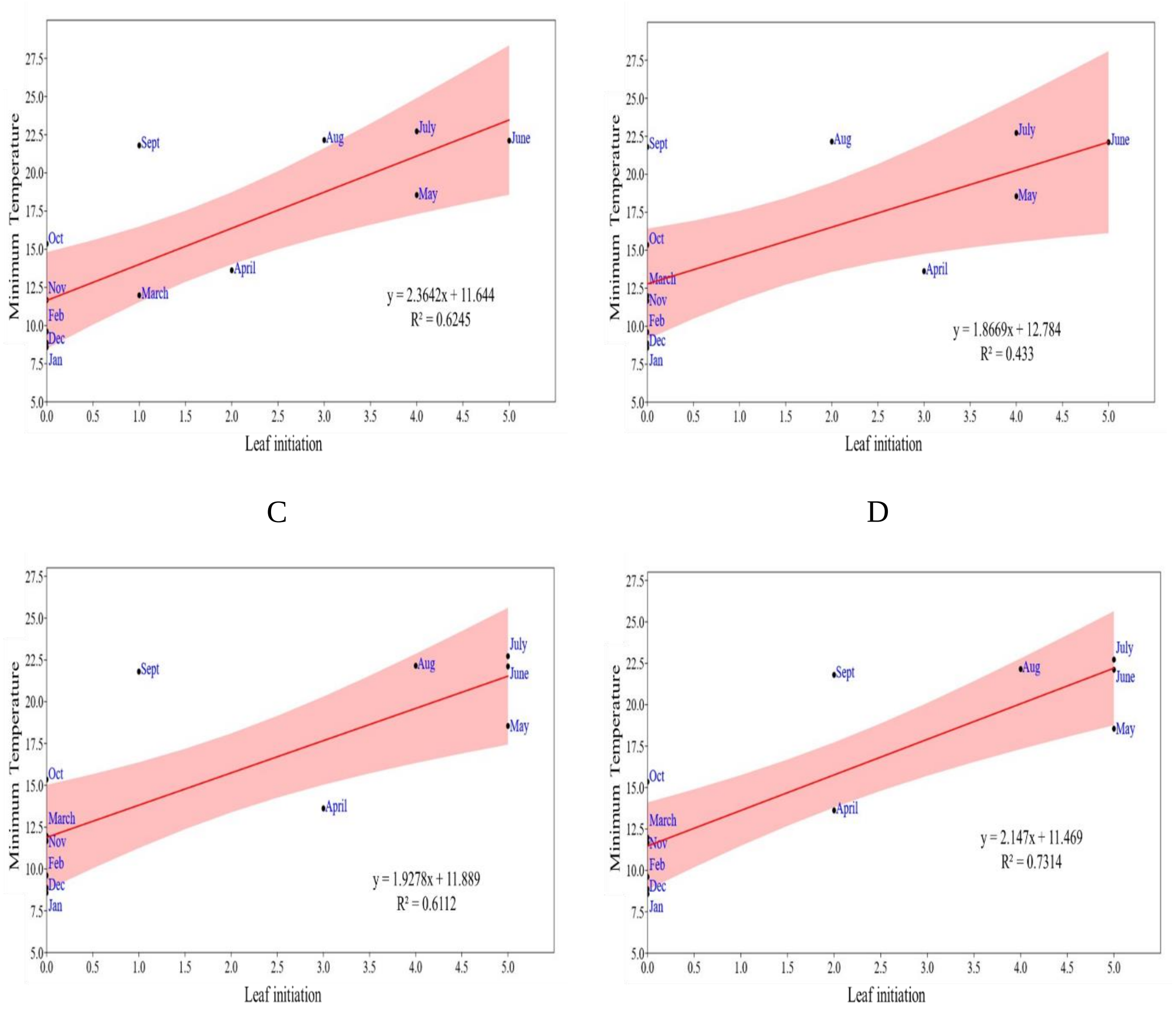


G H

Figure 5.3: Regression of leaf initiation phenology of eight invasive species- *A-Ageratum conyzoides*, *B-Chromolaena odorata*, *C-Mikania micrantha*, *D-Solanum torvum*, *E-Solanum viarum*, *F-Synedrella nodiflora*, *G-Triumfetta rhomboidea* and *H-Urena lobata* as functions of precipitation of the months in Pakke Tiger Reserve.



A B



C

D

E

F

Figure 5.4: Regression of leaf initiation phenology of six invasive species- *A-Chromolaena odorata*, *B-Mikania micrantha*, *C-Solanum torvum*, *D-Solanum viarum*, *E-Synedrella nodiflora* and *F-Triumfetta rhomboidea* as functions of minimum temperature of the months in Pakke Tiger Reserve.

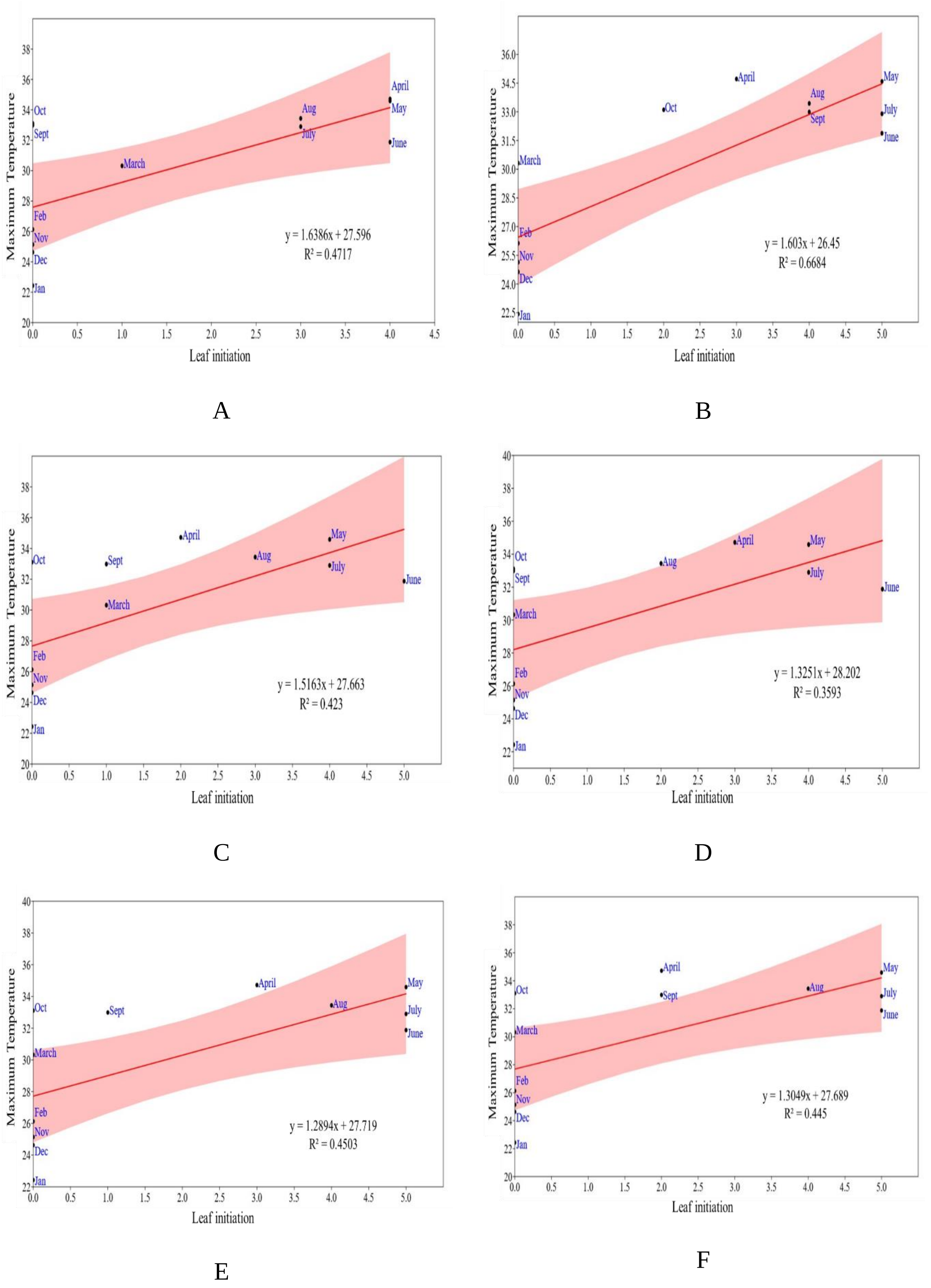


Figure 5.5: Regression of leaf initiation phenology of six invasive species- A-*Ageratum conyzoides*, B-*Chromolaena odorata*, C-*Solanum torvum*, D-*Solanum viarum*, E-*Synedrella nodiflora* and F-*Triumfetta rhomboidea* as functions of maximum temperature of the months in Pakke Tiger Reserve.

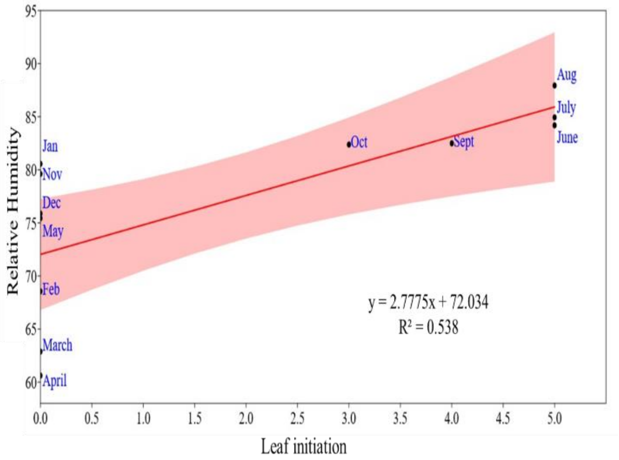


Figure 5.6: Regression of leaf initiation phenology of *Mikania micrantha* and as functions of relative humidity of the months in Pakke Tiger Reserve.

5.5.3.3 Flowering and Climatic Parameters

In the present study, five invasive alien plant species (*Ageratum conyzoides*, *Bidens pilosa*, *Chromolaena odorata, Mikania micrantha, Solanum viarum* and *Urena lobata*) were found having significant relationship (*P*<0.05) with the climatic parameters. *A. conyzoides*, *Bidens pilosa*, *Solanum viarum* and *Urena lobata* flowering phenophases was found significantly related with relative humidity (Fig. 5.8) while the precipitation, minimum and maximum temperature were found to be influencing the flowering phenology of *Mikania micrantha* (Fig. 5.9). However, the flowering phenology of *Chromolaena odorata* display only relation with precipitation (Fig. 5.7) while the rest of remaining invasive alien plant species (*Bidens pilosa, Solanum torvum*, *Synedrella nodiflora* and *Triumfetta rhomboidea*) does not seem to have any significant relation (*P*>0.05) with climatic parameters (Table 5.15). Moreover, the climatic parameters such as precipitation, minimum and maximum temperature have influenced the early flowering in *Chromolaena odorata* and *Mikania micrantha* in the study area. Similarly, relative humidity also influences earlier flowering in *A. conyzoides*, *Solanum viarum* and *Urena lobata* in Pakke Tiger Reserve.

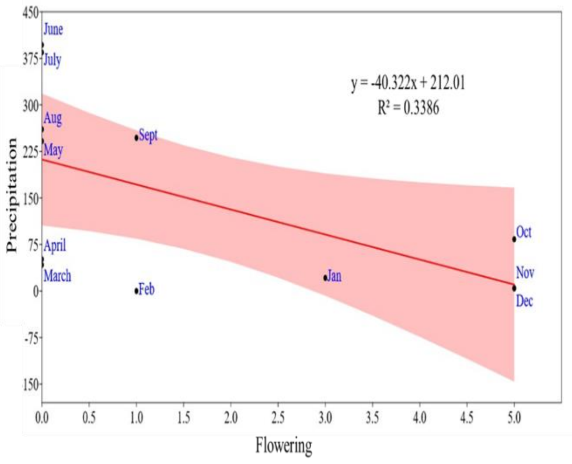
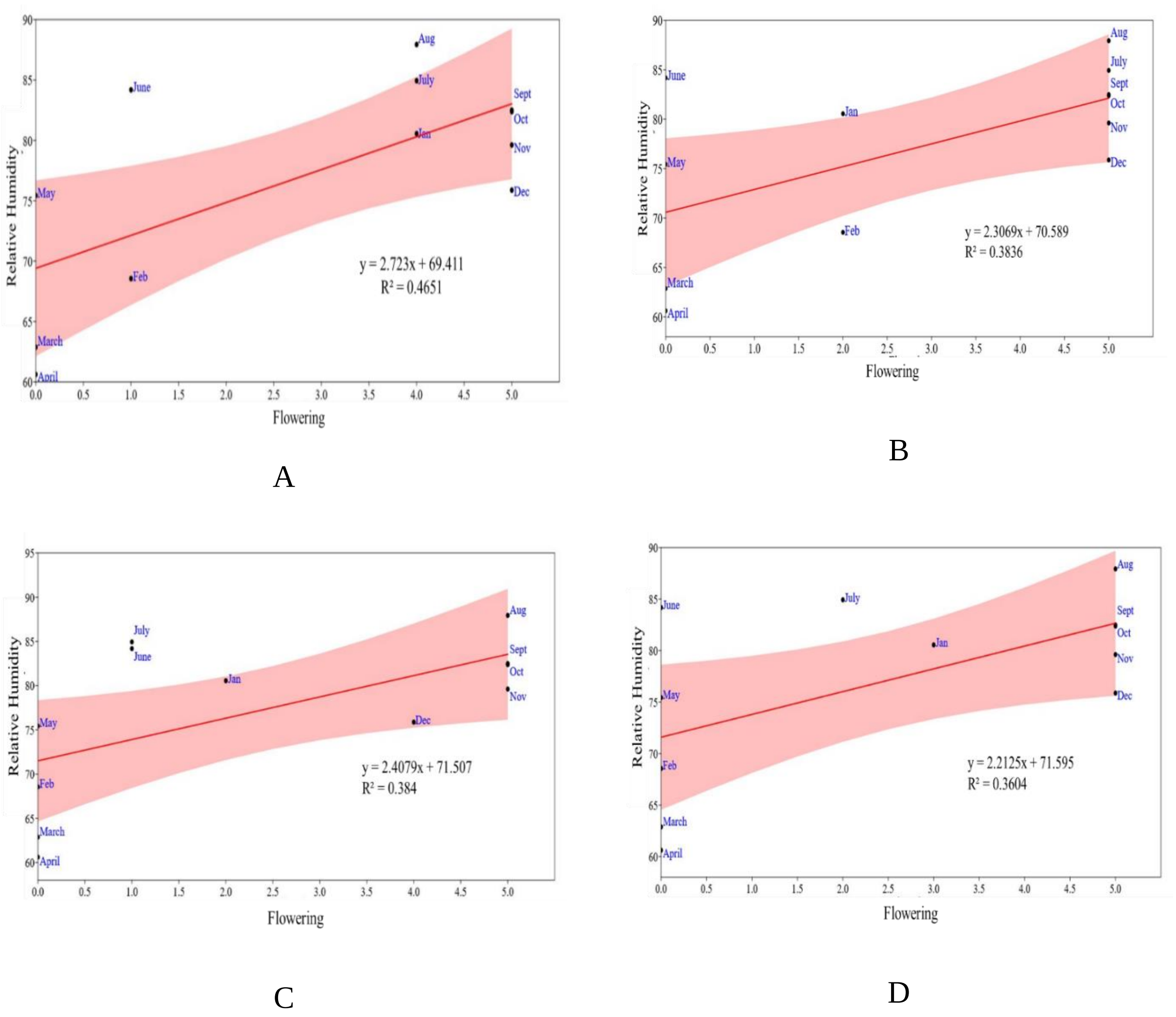


Figure 5.7: Regression of flowering phenology of *Chromolaena odorata* as functions of precipitation of the months in Pakke Tiger Reserve.



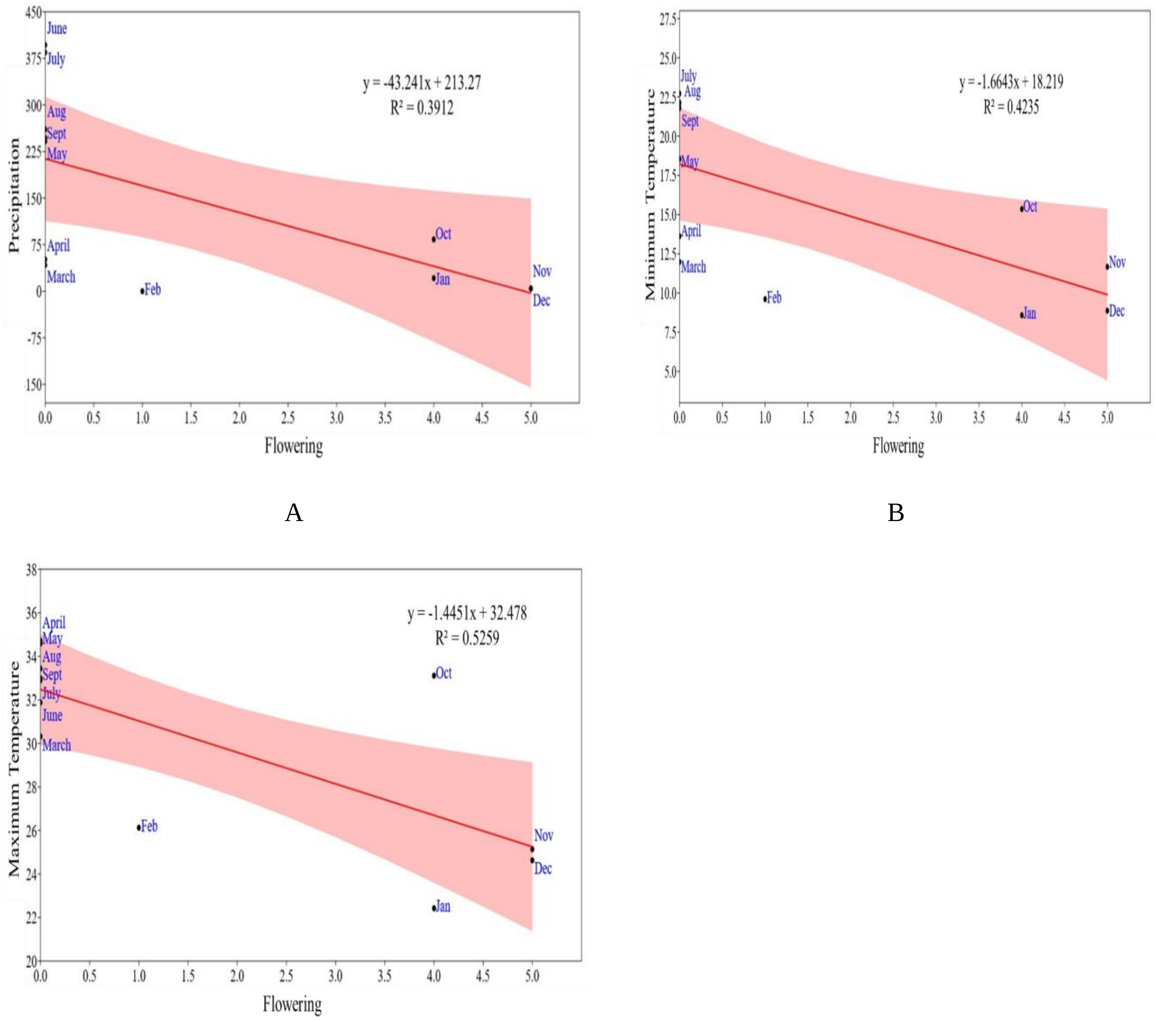
A

B

C

D

Figure 5.8: Regression of flowering phenology of (A) *Ageratum conyzoides* (B*)* *Bidens pilosa* (C) *Solanum viarum* and (D) *Urena lobata* and as functions of relative humidity of the months in Pakke Tiger Reserve.



A

B

C

Figure 5.9: Regression of flowering phenology of *Mikania micrantha* as functions of (A) Precipitation, (B) minimum temperature and (C) maximum temperature of the months in Pakke Tiger Reserve.

Table 5.15. Correlation coefficient (R^2) and slopes for linear regression describing the relationship of flowering phenophases and climatic parameters for the selected invasive alien plant species in Pakke Tiger Reserve during the study period. *****-Significant relationship, PreC- Precipitation, RelH- Relative Humidity, MinT- Minimum Temperature, MaxT- Maximum Temperature.

Species	**Flowering**							
	PreC		**RelH**		**MinT**		**MaxT**	
	R^2	*Slope*	R^2	*Slope*	R^2	*Slope*	R^2	*Slope*
Ageratum conyzoides	0.002	-3.0508	0.465	**2.723***	0.0009	0.0766	0.102	-0.63391
Bidens pilosa	0.0001	0.695	0.384	**2.307***	0.017	0.313	0.026	-0.297
Chromolaena odorata	0.339	**-40.322***	0.035	0.752	0.309	-1.426	0.359	-1.196
Mikania micrantha	0.391	**-43.241***	0.023	0.6128	0.423	**-1.6643***	0.525	**-1.4451***
Solanum torvum	0.049	-14.673	0.171	1.5907	0.04	-0.1719	0.043	-0.39655
Solanum viarum	0.003	-4.1434	0.384	**2.4079***	0.012	0.2817	0.015	-0.24282
Synedrella nodiflora	0.059	-15.436	0.22	1.72	0.005	-0.1724	0.54	-0.42361
Triumfetta rhomboidea	0.124	-21.405	0.202	1.5873	0.062	-0.5615	0.198	-0.77913
Urena lobata	0.016	-8.1085	0.36	**2.2125***	0.0003	0.0453	0.051	-0.41365

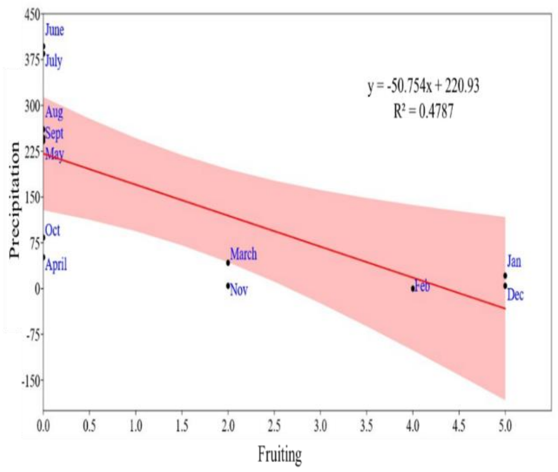
5.5.3.4 Fruiting and Climatic Parameters

In the fruiting phenophases, precipitation seems to play an important role in comparison to other climatic parameters (Table 5.16). All the selected invasive alien plant species i.e., *Ageratum conyzoides, Bidens pilosa, Chromolaena odorata, Mikania micrantha, Solanum torvum, Solanum viarum, Synedrella nodiflora, Triumfetta rhomboidea* and *Urena lobata* were significantly related with precipitation in the study area (Fig. 5.10). Moreover, minimum temperature was also responsible for the fruiting phenophases of *Ageratum conyzoides, Bidens pilosa, Chromolaena odorata, Mikania micrantha, Solanum torvum, Synedrella nodiflora, Triumfetta rhomboidea* and *Urena lobata* in the study area (Fig. 5.11). While, *Bidens pilosa, Chromolaena odorata, Mikania micrantha, Solanum torvum, Solanum viarum, Synedrella nodiflora, Triumfetta rhomboidea* and *Urena lobata* were influenced by the maximum temperature of the study area (Fig. 5.12).

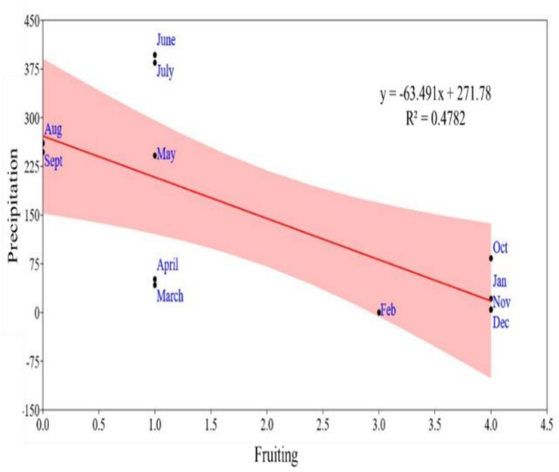
Table 5.16. Correlation coefficient (R^2) and slopes for linear regression describing the relationship of fruiting phenophases and climatic parameters for the selected invasive alien plant species in Pakke Tiger Reserve during the study period. *****-Significant relationship, PreC- Precipitation, RelH- Relative Humidity, MinT- Minimum Temperature, MaxT- Maximum Temperature.

	Fruiting							
Species	**PreC**		**RelH**		**MinT**		**MaxT**	
	R^2	*Slope*	R^2	*Slope*	R^2	*Slope*	R^2	*Slope*
Ageratum conyzoides	0.479	**-50.754***	0.055	-0.996	0.711	**-2.2878***	0.870	-1.972
Bidens pilosa	0.478	**-63.491***	0.0004	-0.117	0.591	**-2.611***	0.581	**-2.017***
Chromolaena odorata	0.691	**-59.764***	0.108	-1.377	0.867	**-2.477***	0.882	**-1.946***
Mikania micrantha	0.391	**-43.225***	0.042	-0.825	0.613	**-2.0028***	0.758	**-1.735***
Solanum torvum	0.649	**-54.135***	0.039	-0.769	0.785	**-2.2025***	0.882	**-1.819***
Solanum viarum	0.697	**-58.211***	0.071	-1.085	0.852	-2.380	0.852	**-1.855***
Synedrella nodiflora	0.589	**-49.452***	0.035	-0.700	0.747	**-2.059***	0.928	**-1.789***
Triumfetta rhomboidea	0.541	**-49.300***	0.094	-1.194	0.752	**-2.151***	0.874	**-1.807***
Urena lobata	0.638	**-51.464***	0.020	-0.528	0.752	**-2.068***	0.831	**-1.693***

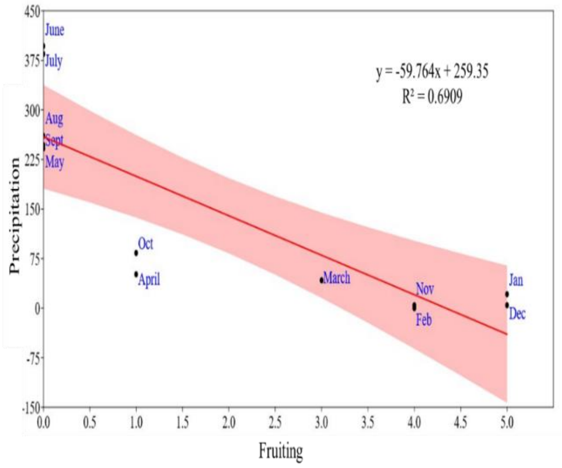
However, relative humidity does not seem to have any influences on the selected invasive species in Pakke Tiger Reserve (Table 5.16). The result of the study clearly indicates that the fruiting phenophases of the selected invasive species have been influenced and initiated early as well as longer fruiting period by precipitation, minimum and maximum temperature in the study region.



A



B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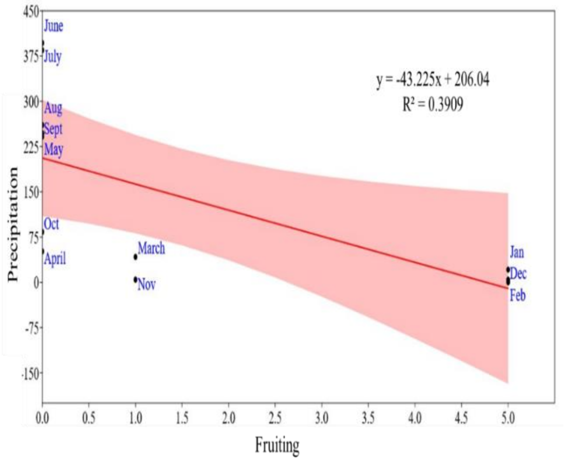
R² = 0.6909
Precipitation
Fruiting
June
July
Aug
Sept
May
Oct
April
March
Nov
Feb
Jan
Dec
450
375
300
225
150
75
0
-75
-150
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5
4.0
4.5
5.0

C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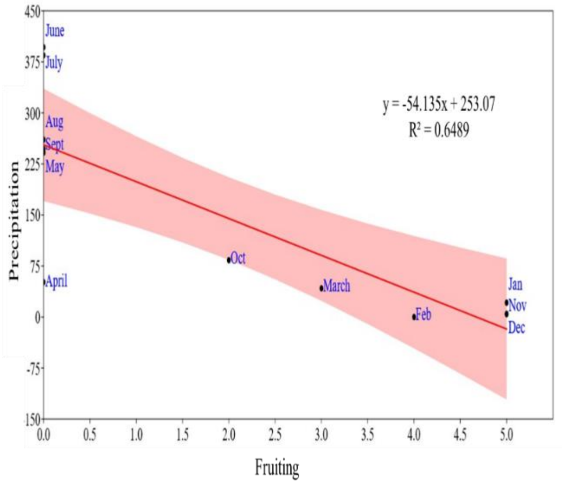
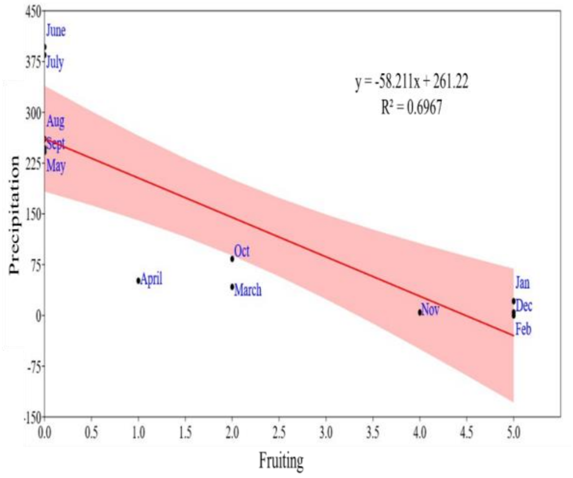
R² = 0.3909
Precipitation
Fruiting
June
July
Aug
Sept
May
Oct
April
March
Nov
Jan
Dec
Feb
450
375
300
225
150
75
0
-75
-150
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5
4.0
4.5
5.0

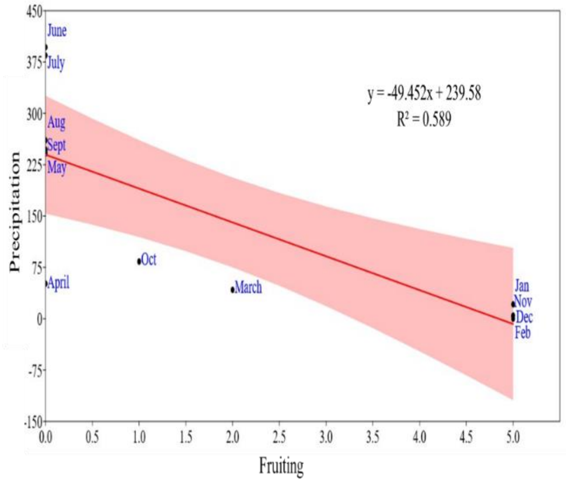
D


y = -54.135x + 253.07
R² = 0.6489
Precipitation
Fruiting
June
July
Aug
Sept
May
April
Oct
March
Feb
Jan
Nov
Dec
450
375
300
225
150
75
0
-75
150
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5
4.0
4.5
5.0

E


y = -58.211x + 261.22
R² = 0.6967
Precipitation
Fruiting
June
July
Aug
Sept
May
Oct
April
March
Nov
Jan
Dec
Feb
450
375
300
225
150
75
0
-75
-150
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5
4.0
4.5
5.0

F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R² = 0.589
Precipitation
Fruiting
June
July
Aug
Sept
May
Oct
April
March
Jan
Nov
Dec
Feb
450
375
300
225
150
75
0
-75
-150
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5
4.0
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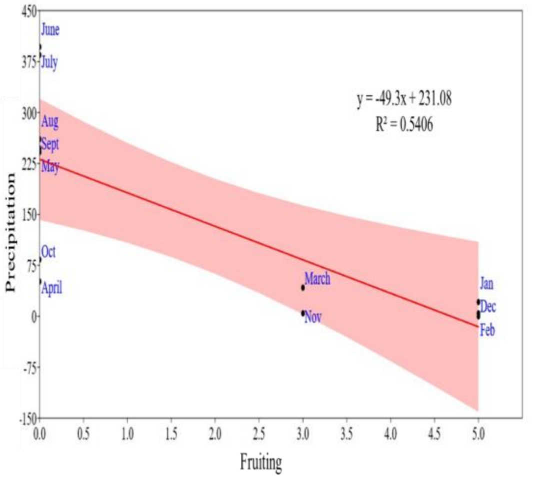
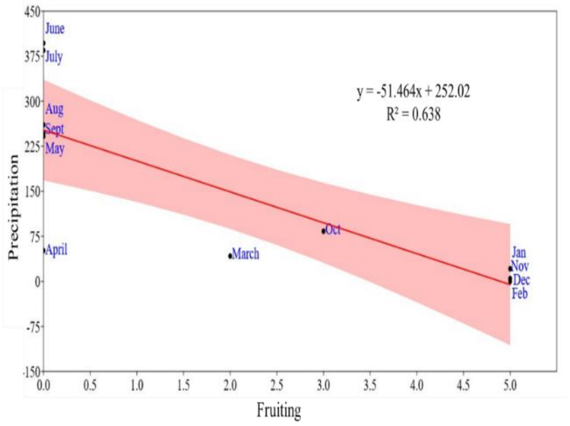
5.0

G


y = -49.3x + 231.08
R² = 0.5406
Precipitation
Fruiting
June
July
Aug
Sept
May
Oct
April
March
Nov
Jan
Dec
Feb
450
375
300
225
150
75
0
-75
-150
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5
4.0
4.5
5.0

H



I

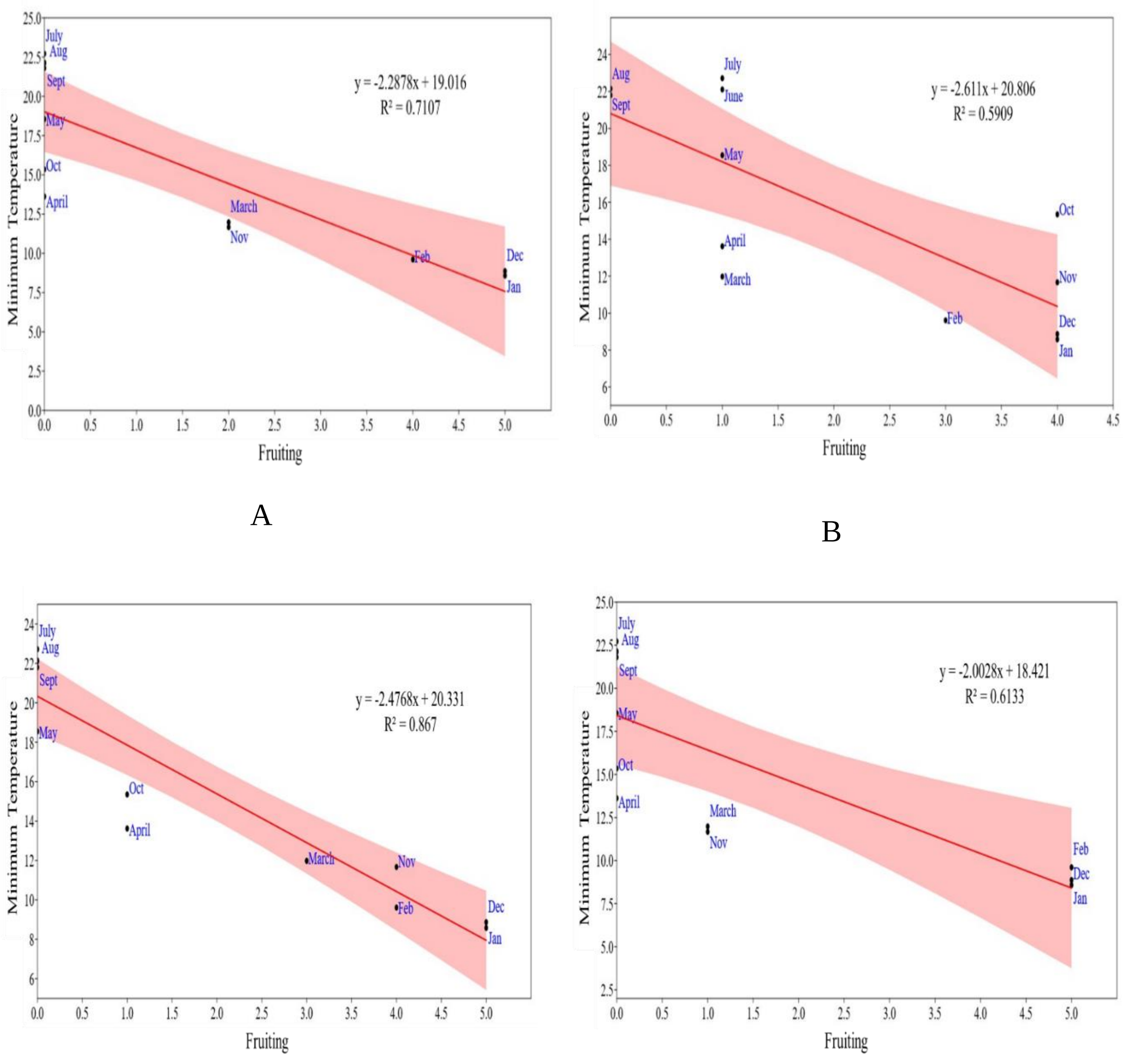
Figure 5.10: Regression of fruiting phenology of eight invasive species- *A-Ageratum conyzoides*, *B-Bidens pilosa*, *C-Chromolaena odorata*, *D-Mikania micrantha*, *E-Solanum torvum*, *F-Solanum viarum*, *G-Synedrella nodiflora*, *H-Triumfetta rhomboidea* and *-Urena lobata* as functions of precipitation of the months in Pakke Tiger Reserve.

Table 5.17. Correlation coefficient (R^2) and slopes for linear regression describing the relationship of leaf fall phenophases and climatic parameters for the selected invasive alien plant species in Pakke Tiger Reserve during the study period. *****-Significant relationship, PreC- Precipitation, RelH- Relative Humidity, MinT- Minimum Temperature, MaxT- Maximum Temperature

	Leaf fall							
Species	**PreC**		**RelH**		**MinT**		**MaxT**	
	R^2	*Slope*	R^2	*Slope*	R^2	*Slope*	R^2	*Slope*
Ageratum conyzoides	0.00003	-0.531	0.58642	**-3.9041***	0.0042	-0.2117	0.124	0.889
Bidens pilosa	0.320	-41.515	0.829	**-3.8877***	0.323	**-1.545***	0.020	-0.299
Chromolaena odorata	0.186	-31.008	0.829	**-3.807***	0.191	-1.162	0.002	0.097
Mikania micrantha	0.0002	-1.086	0.526	**-2.673***	0.005	-0.172	0.097	0.569
Solanum torvum	0.081	-19.794	0.758	**-3.523***	0.101	-0.819	0.013	0.226
Solanum viarum	0.105	-23.753	0.846	**-3.929***	0.095	-0.837	0.031	0.374
Synedrella nodiflora	0.075	-17.658	0.740	**-3.224***	0.065	-0.606	0.066	0.477
Triumfetta rhomboidea	0.092	-22.762	0.835	**-3.988***	0.060	-0.682	0.078	0.604
Urena lobata	0.166	-26.862	0.774	**-3.373***	0.178	-1.028	0.002	0.087

5.5.3.5 Leaf Fall and Climatic Parameters

Relative humidity had major influences on all the selected invasive alien plant species in the leaf fall phenology. *Ageratum conyzoides*, *Bidens pilosa*, *Chromolaena odorata*, *Mikania micrantha*, *Solanum torvum*, *Solanum viarum*, *Synedrella nodiflora*, *Triumfetta rhomboidea* and *Urena lobata* were found to be influence by relative humidity (Fig. 5.13 and 5.14) and significantly related (Table 5.17). However, *Bidens pilosa* was also significantly ($P<0.05$) related with minimum temperature in the study area (Fig. 5.13). In this phenophases, falling of leaf was mostly governed by the increase or decrease of relative humidity as the other climatic parameters does not have much influence on falling of leaf in the selected invasive species. All the selected invasive species shows no relation with maximum temperatures in the present study.

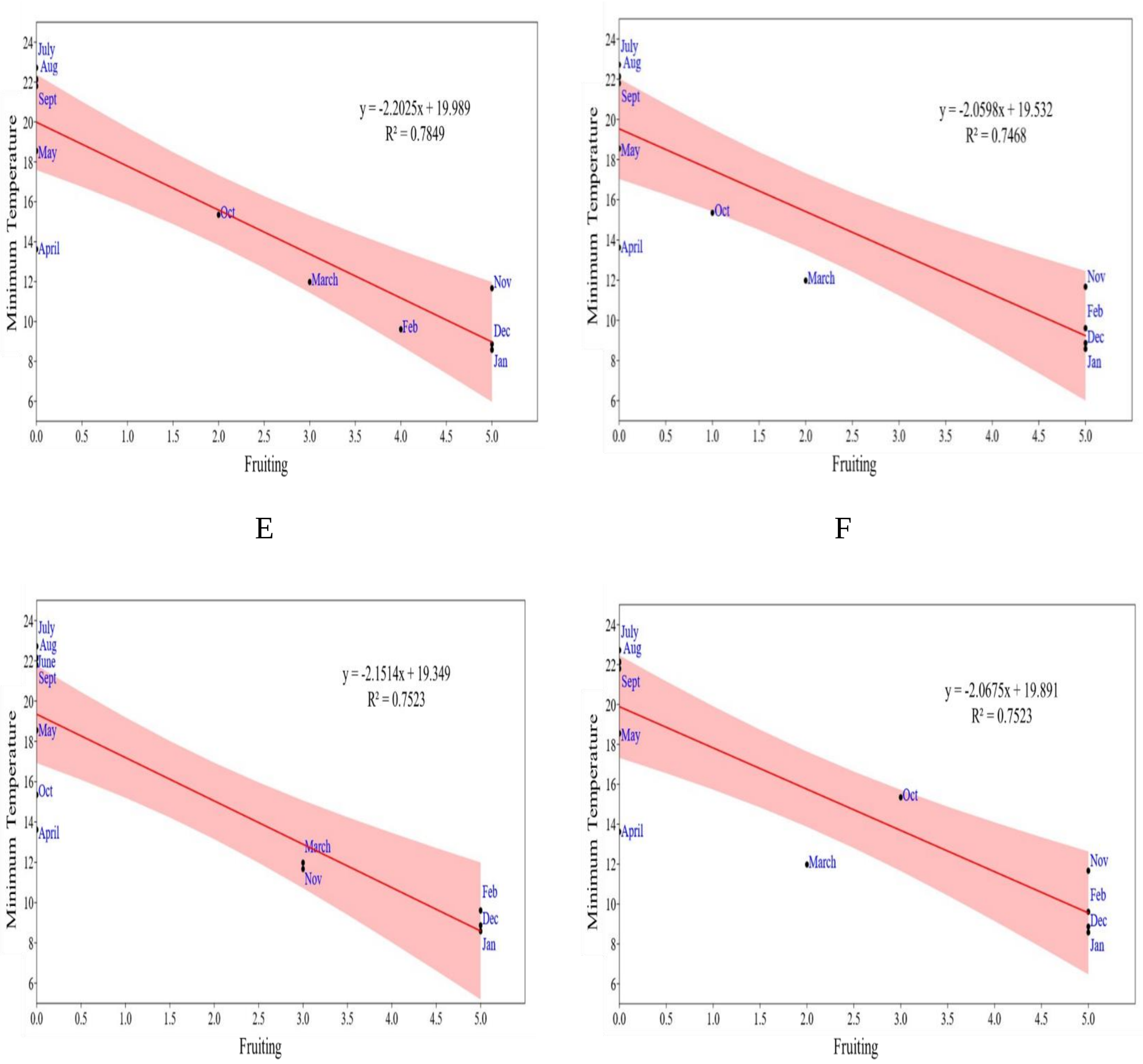


A

B

C

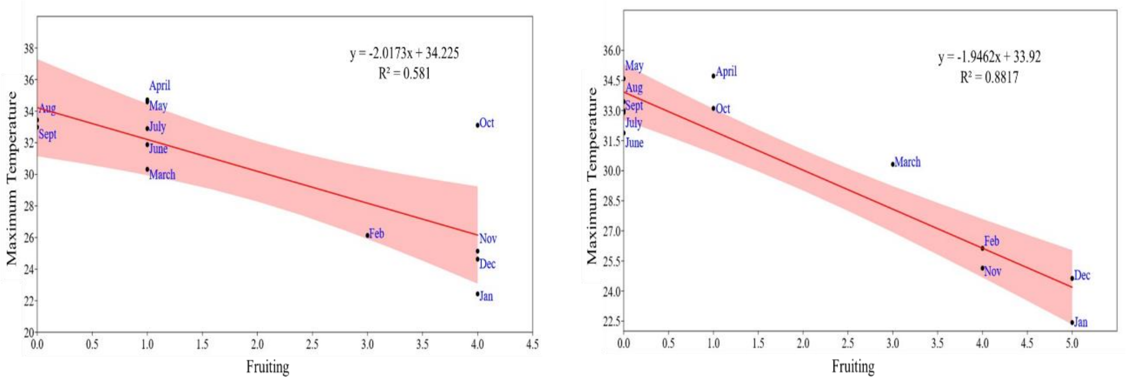
D



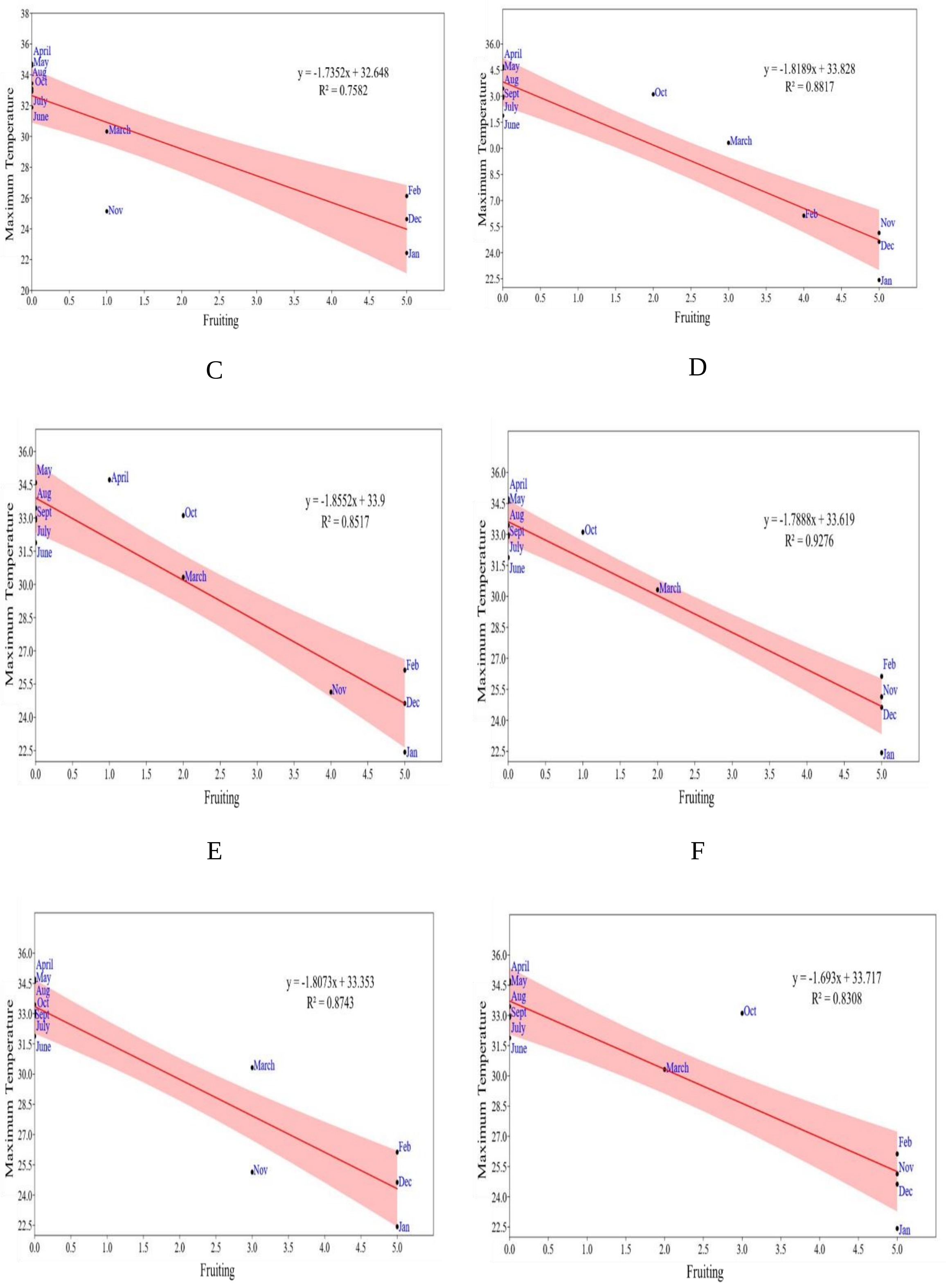
E F

G H

Figure 5.11: Regression of fruiting phenology of eight invasive species- *A-Ageratum conyzoides*, *B-Bidens pilosa*, *C-Chromolaena odorata*, *D-Mikania micrantha*, *E-Solanum torvum*, *F-Synedrella nodiflora*, *G-Triumfetta rhomboidea* and *H-Urena lobata* as functions of minimum temperature of the months in Pakke Tiger Reserve.



A B



C

D

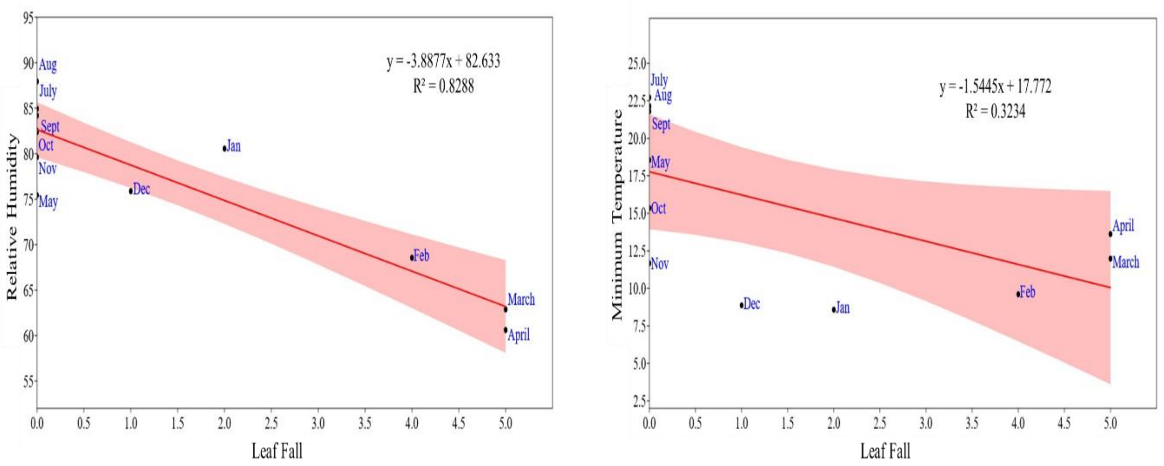
E

F

G

H

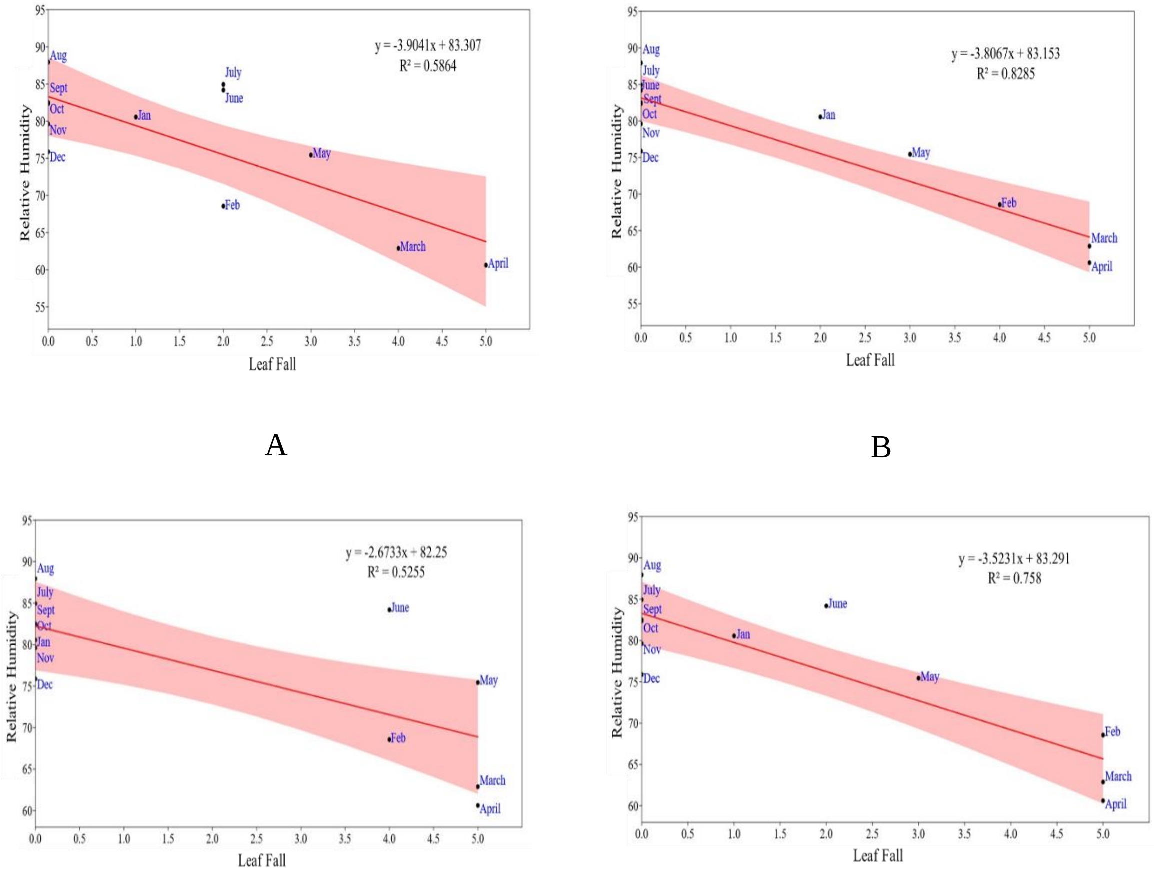
Figure 5.12: Regression of fruiting phenology of eight invasive species- *A-Bidens pilosa*, *B-Chromolaena odorata*, *C-Mikania micrantha, D-Solanum torvum, E-Solanum viarum, F-Synedrella nodiflora, G-Triumfetta rhomboidea* and *H-Urena lobata* as functions of maximum temperature of the months in Pakke Tiger Reserve.



A

B

Figure 5.13: Regression of leaf fall phenology of *Bidens pilosa* as functions of (A) relative humidity and (B) minimum temperature of the months in Pakke Tiger Reserve.

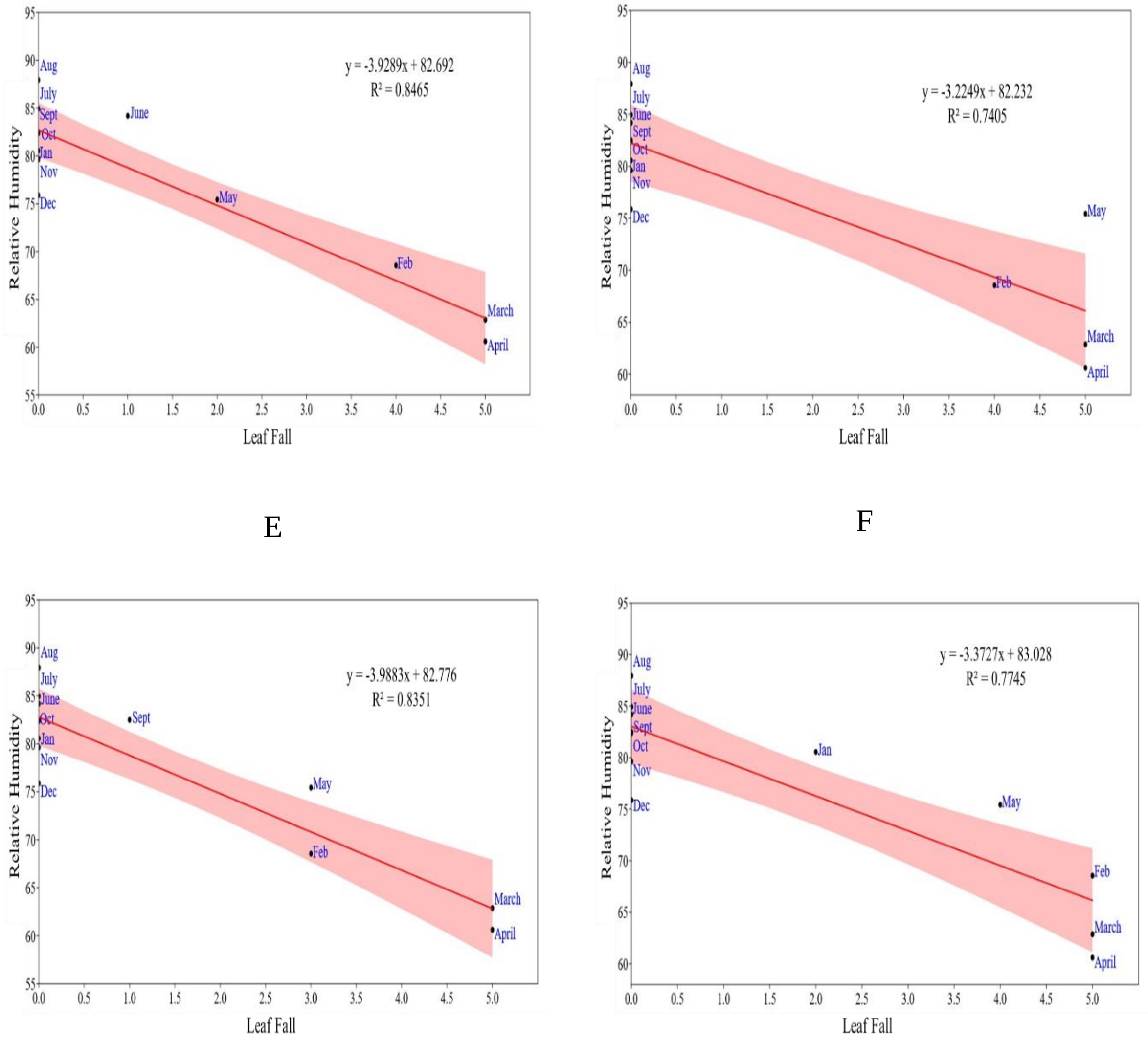


A

B

C

D



E

F

G

H

Figure 5.14: Regression of leaf fall phenology of all the eight invasive alien plant species- *A-Ageratum conyzoides, B-Chromolaena odorata, C-Mikania micrantha, D-Solanum torvum, E-Solanum viarum, F-Synedrella nodiflora, G-Triumfetta rhomboidea* and *H-Urena lobata* as functions of relative humidity of the months in Pakke Tiger Reserve.

5.5.4 Correlation Between the Climatic Parameters and Phenophases

Pearson correlation was used to determine the significance between the phenophases of selected invasive alien plant species (IAPs) and the environmental variables, i.e., minimum temperature, maximum temperature, precipitation and relative humidity (Table 5.18).

5.5.4.1 Leaf Initiation and Environmental Variables

The phenophases of leaf initiation also display correlation with the environmental variables. In minimum temperature and precipitation, *Ageratum conyzoides* and *Bidens pilosa* show non significance ($p>0.05$) while the remaining species display only positive correlation, i.e., $r = 0.82$, 0.82, 0.57, 0.27, 0.86, 0.68 and 0.73; $p<0.05$, 0.001 and 0.0001: $r = 0.82$, 0.82, 0.63, 0.34, 0.86, 0.69 and 0.74; $p<0.05$, 0.001 and 0.0001, respectively. However, correlation with maximum temperature and humidity expressed both positive and negative association. In maximum temperature, only *Ageratum conyzoides* ($r = -0.18$, $p<0.05$) relate negative association while six of them, i.e., *Chromolaena odorata* ($r = 0.26$, $p < 0.05$), *Mikania micrantha* ($r = 0.26$, $p < 0.05$), *Solanum viarum* ($r = 0.26$, $p < 0.05$), *Synedrela nodiflora* ($r = 0.26$, $p<0.05$), *Triumfetta rhomboidea* ($r = 0.26$, $p < 0.05$) and *Urena lobata* ($r = 0.26$, $p < 0.05$), show positive correlation. With humidity, *Bidens pilosa* only show non-significant ($p > 0.05$) while *Ageratum conyzoides* ($r = -0.27$, $p < 0.05$) and *Solanum viarum* ($r = -0.26$, $p < 0.05$) show negative correlation and the remaining species display positive association, i.e., *Chromolaena odorata* ($r = 0.82$, $p < 0.0001$), *Mikania micrantha* ($r = 0.82$, $p < 0.0001$), *Solanum torvum* ($r = 0.35$, $p < 0.05$),), *Synedrela nodiflora* ($r = 0.66$, $p < 0.001$), *Triumfetta rhomboidea* ($r = 0.27$, $p<0.05$) and *Urena lobata* ($r = 0.51$, $p<0.001$), respectively.

5.5.4.2 Flowering and Environmental Variables

The correlation between the flowering and minimum temperature shows both negative and positive correlation in five species, i.e., *Ageratum conyzoides* ($r = -0.18$; $p<0.05$), *Bidens pilosa* ($r = -0.22$; $p<0.05$), *Chromolaena odorata* ($r = -70$; $p<0.001$) and *Mikania micrantha* ($r = -0.61$; $p<0.001$) while *Solanum viarum* show positive relationship ($r = 0.77$; $p<0.001$). Similarly, the relation between the flowering and maximum temperature shows both in which *Ageratum conyzoides* ($r = -0.36$; $p<0.05$), *Bidens pilosa* ($r = -0.33$; $p<0.05$), *Chromolaena odorata* ($r = -64$; $p<0.001$), *Mikania micrantha* ($r = -0.56$; $p<0.001$) and *Urena lobata* ($r = -0.29$; $p<0.05$) display negative correlation though *Solanum viarum* show positive relationship ($r = 0.30$; $p<0.05$). In case of flowering with relative humidity, correlation show only positive relationship in seven species (*Ageratum conyzoides*: $r = 0.28$; $p<0.05$, *Bidens pilosa*: $r = 0.27$; $p<0.05$, *Solanum torvum*: $r = 0.55$; $p<0.001$, *Solanum viarum*: $r = 0.79$; $p<0.001$, *Synedrella nodiflora:* $r = 0.38$; $p<0.05$, *Triumfetta rhomboidea:* $r = 0.42$; $p<0.05$ and *Urena lobata:* $r = 0.73$; $p<0.001$). Moreover, precipitation (rainfall) also displays negative (*Ageratum conyzoides*: $r = -0.22$; $p<0.05$, *Bidens pilosa*: $r = -0.24$; $p<0.05$, *Chromolaena odorata*: $r = -70$; $p<0.001$ and *Mikania micrantha*: $r = -0.51$; $p<0.001$) and positive correlation (*Solanum viarum:* $r = 0.75$; $p<0.001$), respectively.

Table 5.18: Correlation between the environmental variables and phenophases (flowering, fruiting, leaf initiation and leaf fall) of the selected invasive alien plant species. *Correlation is significant at p<0.05, **<0.001, ***<0.0001, ns- not significant.

Correlation	*Ageratum conyzoides*	*Bidens pilosa*	*Chromolaena odorata*	*Mikania micrantha*	*Solanum torvum*	*Solanum viarum*	*Synedrella nodiflora*	*Triumfetta rhomboidea*	*Urena lobata*
Flowering x Minimum Temperature	-0.18*	-0.22*	-0.7**	-0.61*	ns	0.77**	ns	ns	ns
Flowering x Maximum Temperature	-0.36*	-0.33*	-0.64**	-0.56**	ns	0.3*	ns	ns	-0.29*
Flowering x Humidity	0.28*	0.27*	ns	ns	0.55*	0.79**	0.38*	0.42*	0.73**
Flowering x Precipitation	-0.22*	-0.24*	-0.7**	-0.51**	ns	0.75**	ns	ns	ns
Fruiting x Minimum Temperature.	-0.48*	-0.5**	-0.76**	-0.75**	-0.79*	-0.35*	-0.85***	-0.86*	-0.82***
Fruiting x Maximum Temperature.	-0.48*	-0.55**	-0.42*	-0.7**	-0.7**	-0.37*	-0.83***	-0.86*	-0.72**
Fruiting x Humidity	ns	ns	-0.71**	ns	-0.3*	0.26*	-0.45*	-0.47*	-0.39*
Fruiting x Precipitation	-0.49*	-0.51**	-0.71**	-0.7**	-0.79**	-0.29*	-0.85***	-0.86***	-0.82***
Leaf initiation x Minimum Temperature	ns	ns	0.82***	0.82***	0.57**	0.27*	0.86***	0.68**	0.73**
Leaf initiation x Maximum Temperature	-0.18*	ns	0.26*	0.26*	ns	0.52**	0.47*	0.58**	0.38*
Leaf initiation x Humidity	-0.27*	ns	0.82***	0.82***	0.35*	-0.26*	0.66**	0.27*	0.51*
Leaf initiation x Precipitation	ns	ns	0.82***	0.82***	0.63**	0.34*	0.86***	0.69**	0.74**
Leaf fall x Minimum Temperature	-0.22*	-0.23*	ns	ns	ns	-0.6*	ns	-0.28*	ns
Leaf fall x Maximum Temperature	ns	-0.15*	0.48*	0.28*	0.28*	-0.36*	0.42*	ns	ns
Leaf fall x Humidity	ns	ns	-0.34*	-0.65**	-0.71**	-0.54**	-0.67**	-0.76**	-0.79**
Leaf fall x Precipitation	-0.23*	-0.25*	0.32*	ns	ns	-0.55**	ns	-0.39*	-0.35*

5.5.4.3 Fruiting and Environmental Variables

The fruiting phenophases expressed both positive and negative correlation in the study. In relation with minimum temperature, maximum temperature and precipitation, it shows only negative correlation in all the selected species, i.e., *Ageratum conyzoides, Bidens pilosa, Chromolaena odorata, Mikania micrantha, Solanum torvum, Solanum viarum, Synedrella nodiflora*, *Triumfetta rhomboidea* and *Urena lobata* (r = -0.48, -0.50, -0.76, -0.75, -0.79, -0.35, -0.85, -0.86 and -0.82: r = -0.48, -0.55, -0.42, -0.70, -0.70, -0.37, -0.83, -0.86 and -0.72: r = -0.49, -0.51, -0.71, -0.70, -0.79, -0.29, -0.85, -0.86 and -0.82, $p<0.05$, 0.001 and 0.0001, respectively). However, in association with relative humidity, six species were correlated with the climatic parameters, which showed both positive correlation (*Solanum viarum:* $r=0.26$; $p<0.05$) and negative (*Chromolaena odorata*: $r=-0.71$; $p<0.001$, *Solanum torvum*: $r=-0.30$; $p<0.05$, *Synedrella nodiflora*: $r=-0.45$; $p<0.05$, *Triumfetta rhomboidea:* $r=-0.47$; $p<0.05$ and *Urena lobata:* $r=-0.39$; $p<0.05$) while *Ageratum conyzoides*, *Bidens pilosa* and *Mikania micrantha* shows non-significant ($p>0.05$), respectively.

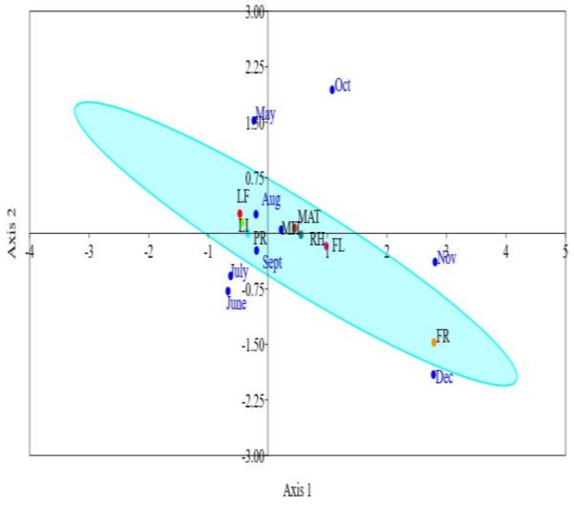
5.5.4.4 Leaf Fall and Environmental Variables

The leaf fall phenophases show both positive and negative relationship with the climatic conditions. The relation of leaf fall with minimum temperature displays only negative correlation in four species i.e., *Ageratum conyzoides* ($r=-0.22$, $p<0.05$), *Bidens pilosa* ($r=-0.23$, $p<0.05$), *Solanum viarum* ($r=-0.60$, $p<0.001$) and *Triumfetta rhomboidea* ($r=-0.28$, $p<0.05$) while the rest shows non-significant ($p>0.05$). With maximum temperature, *Bidens pilosa* ($r=-0.15$, $p<0.05$) and *Solanum viarum* ($r=-0.36$, $p<0.05$) shows negative correlation while *Chromolaena odorata* ($r=0.48$, $p<0.05$), *Mikania micrantha* ($r=0.28$, $p<0.05$), *Solanum torvum* ($r=0.28$, $p<0.05$) and *Synedrella nodiflora* ($r=0.42$, $p<0.05$). In association with humidity, *Ageratum conyzoides* and *Bidens pilosa* indicate non-significant while the remaining seven species, i.e., Chromolaena *odorata* ($r=-0.34$, $p<0.05$), *Mikania micrantha* ($r=-0.65$, $p<0.001$), *Solanum torvum* ($r=-0.71$, $p<0.001$) *Solanum viarum* ($r=-0.54$, $p<0.001$), *Synedrella nodiflora* ($r=-0.67$, $p<0.001$), *Triumfetta rhomboidea* ($r=-0.78$, $p<0.001$) and *Urena lobata* ($r=-0.79$, $p<0.001$), shows only negative correlation. In addition, precipitation had both negative (*Ageratum conyzoides,* $r=-0.23$, $p<0.05$; *Bidens pilosa,* $r=-0.25$, $p<0.05$; *Solanum viarum,* $r=-55$, $p<0.001$; *Triumfetta rhomboidea*, $r=-0.39$, $p<0.05$; *Urena lobata*, $r=-0.35$, $p<0.05$) and positive correlation (*Chromolaena odorata,* $r=0.32$, $p<0.05$), respectively.

5.5.5 Canonical Correspondence Analysis (CCA)

In the present study, canonical correspondence analysis (CCA) was employed to investigate the relationships between the phenophases and climatic parameters i.e., environmental variables (precipitation, relative humidity, minimum and maximum temperature). All the selected nine species phenophases were measured to find the seasonal pattern and their association between certain phenological events and environmental variables (Fig. 5.15). The CCA result of *Ageratum conyzoides* reveals that the primary environmental drivers that influenced the timing of phenological events were relative humidity, minimum and maximum temperature. The separation between the phenophases of flowering and fruiting indicate the distinct environmental requirements for these species while the leaf fall and initiation are closely positioned suggesting similar climatic conditions.

However, the month of November falls outside the ellipse describing unique climatic conditions or phenophases that differ significantly from the seasonal patterns. In addition, precipitation is located near the centre of the plot implying a weaker correlation with other environmental variables but might play a supporting role in shaping the phenological events. Moreover, it does not strongly influence or correlate with the primary axes of variation (Axis 1 and Axis 2).



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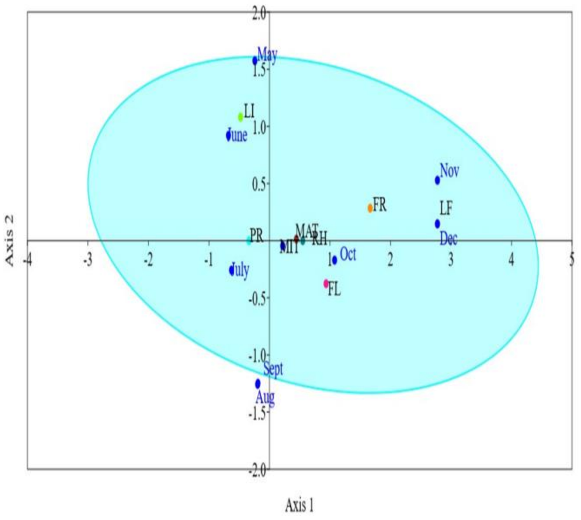

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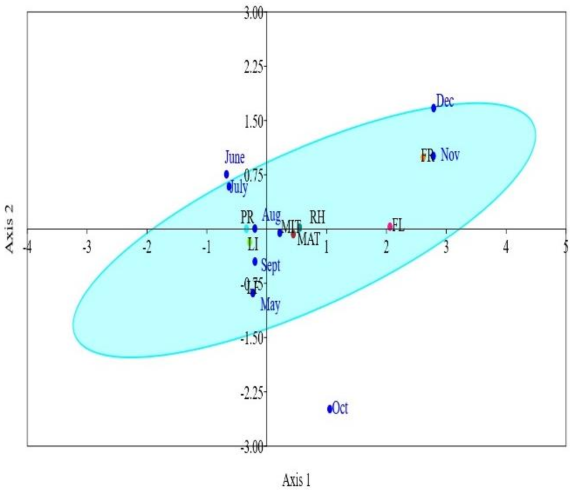
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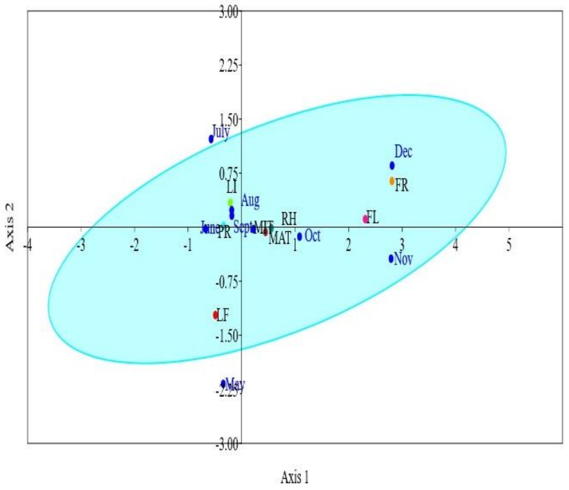
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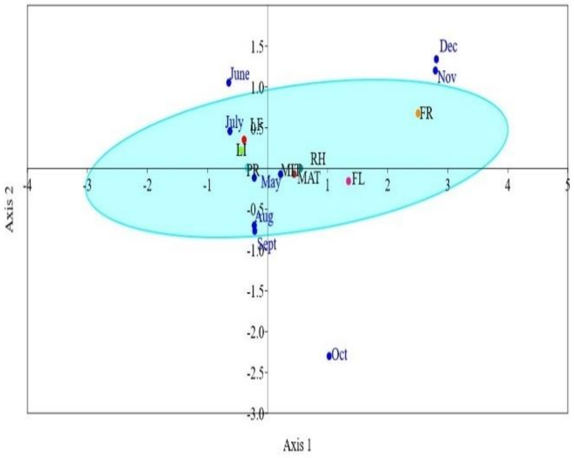
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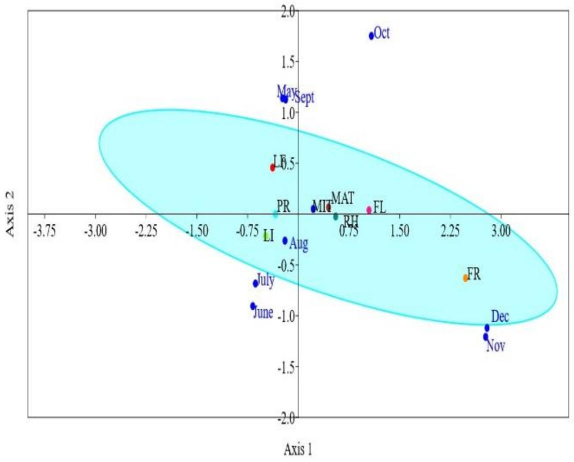
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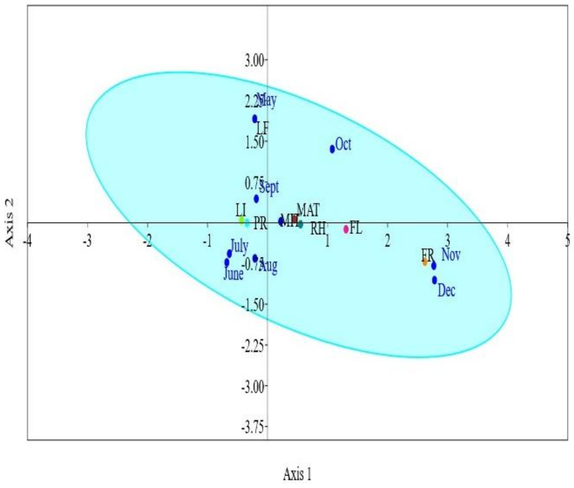
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1.50
0.75
-0.75
-1.50
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-4
-3
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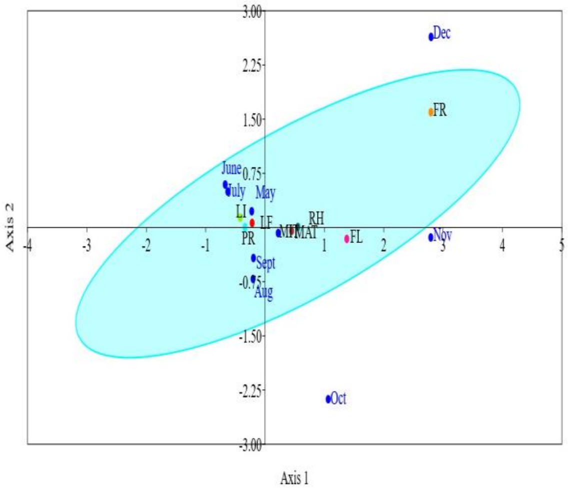
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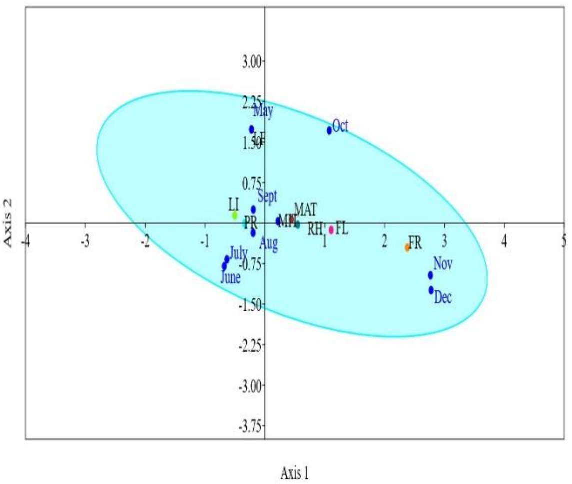

Figure 5.15: Temporal pattern of nine IAPs on the relationships between climatic parameter and phenophases. LI- leaf initiation, LF- leaf fall, FR- fruiting, FL- flowering, PR- precipitation, MIT- minimum temperature, MAT- maximum temperature and RH-relative humidity.

I

However, precipitation and maximum temperature significantly influence the timing and occurrence of phenological events in *Bidens Pilosa*. In addition, both minimum temperature and relative humidity positioned near the centre indicating their moderate influence on the phenophases. Maximum temperature is strongly associated with the phenophases of fruiting and flowering while precipitation influences leaf initiation and flowering. Moreover, relative humidity and precipitation had an association with the overall phenophases pattern particularly during the growing season and in the timing of leaf fall.

The CCA plot of *Chromolaena odorata* highlighted the strong influence of maximum temperature and relative humidity on phenophases. The phenology of flowering is driven by warmer conditions while leaf initiation and fall are influenced by precipitation and humidity. Fruiting occurs during the cooler, more humid months reflecting seasonal environmental changes. However, the month of October is an outlier in lower portion suggesting unique climatic parameters than the rest of the variables. The data points form an elliptical distribution suggesting a gradient or continuum of variation along the two axes.

In *Mikania micrantha,* precipitation and humidity were strongly linked to all phenophases, particularly leaf initiation while the temperature variables (maximum and minimum) gain prominence during transitional and cooler months influencing leaf fall. Additionally, a strong positive correlation is observed between flowering and fruiting as well as separation between leaf initiation and reproductive phades i.e., flowering and fruiting. Environmental variables also show varying degree of influence on plant phenology in *Mikania micrantha*.

The CCA plot of *Solanum torvum* provides insight into the relationship between phenophases and environmental factors across the different months. Flowering and fruiting were associated with precipitation, minimum and maximum temperature in axis 1 while the relative humidity representing the secondary gradients in axis 2 influenced leaf fall and leaf initiation. Moreover, the ellipse encompasses most of the data points that majority of variation is explained by the two axes with October month being different from the other months. The clustering of months and phenophases indicate the temporal pattern in the plant life cycle with specific events occurring during distinct environmental conditions.

However, the relative humidity and maximum temperature were closely associated with flowering phenophases of *Solanum viarum*. In addition, minimum temperature positioning near the precipitation indicates a potential relationship between cooler temperature and rainfall. The elliptical region encloses most of the points suggesting variation in the data. In *Synedrella nodiflora*, environmental variables i.e., relative humidity, minimum and maximum temperature were closely aligned with flowering phenophases while the precipitation play a more general influence on the phenophases. Moreover, the phenophases like leaf fall and fruiting were tied to specific months indicating their timing is influenced by environmental conditions. In axis 1, the primary gradient of variation is likely driven by temperature variables and its seasonal fluctuation with flowering and fruiting phenophases influenced by temperature dependent process while the secondary gradient influences the variables such as relative humidity and precipitation with leaf fall and initiation were positioned along the axis 2 indicating their dependence on moisture related factors.

The plot of *Triumfetta rhomboidea* displayed the close association between precipitation and minimum temperature and leaf fall and initiation indicating strong influence in these phenophases. Flowering phenology is linked with relative humidity and maximum temperature while the fruiting is positioned further along axis 1 indicating a distinct relationship with climatic parameters compared to other phenophases. Moreover, the clustering of months and phenophases suggest the seasonal cycle with specific events occurring under particular environmental conditions. The elliptical region encloses most of the points likely representing the main cluster of the data points. Furthermore, the CCA analysis of *Urena lobata* shows that leaf initiation is closely associated with precipitation and minimum temperature while flowering and fruiting were strongly associated with maximum temperature and humidity. Also, the elliptical region encompasses most of the data points indicating the spread or variability of the data. However, minimum temperature plays a crucial role in influencing in different phenophases particularly during the transition from leaf initiation to flowering phenophases.

5.6 Discussion

5.6.1 Phenophases

The phenological patterns i.e., flowering, fruiting, leaf fall and leaf initiation of the selected invasive species (*Ageratum conyzoides, Bidens pilosa, Chromolaena odorata, Mikania micrantha, Solanum torvum, Solanum viarum, Synedrella nodiflora, Triumfetta rhomboidea* and *Urena lobata*) were studied. The present study found differences and variations between phenology across the different forest types of Pakke Tiger Reserve. Some of the species have shown similar pattern of phenophases while other display distinct phenophases in the study area. In invasive species, flowering and fruiting phenophases were key traits that have been recurrently studied (Moravcova et al., 2010; Hornoy et al., 2011; Lediuk et al., 2014) in comparison to leaf phenology i.e., leaf fall and initiation (Piao et al., 2019). However, the present study studied all the key traits of phenophases in invasive species. The flowering phenophases of *Ageratum conyzoides* observed in the present study coincides with the findings of Sarma and Bhattacharjya (2016) and Kumar et al. (2023). Longer period of flowering with violet colour retaining was also observed in the study area which was parallel to the findings of Kaur et al. (2012). Correspondingly, long flowering duration of *Bidens pilosa* was also observed in the study region which was also in accordance with the results of Kumar et al. (2023) and Kato-Noguchi and Kurniadie (2024). The peak flowering period of *Chromolaena odorata* in all forest types was from October to December in Pakke Tiger Reserve. A similar pattern of phenophases was observed by Rathnayake and Wijetunga (2016). The timing of *Mikania micrantha* blooming was from October to January and their average duration were found almost similar with the results of Shen et al. (2016). The flowering pattern of *Solanum torvum* observed in the present study was in accordance with the study of Paul and Kumar (2023) and Knapp (2024a). Moreover, the blooming period of *Solanum viarum* in Pakke Tiger Reserve was also similar with the study of Mullahey et al. (1993), Knapp (2024b) and CABI (2024). *Synedrella nodiflora* flowering phenophases across the forest types were found similar with the findings of Mbatudde et al. (2007) and Win and Hnin (2022). The phenology of *Triumfetta rhomboidea* was also concurred with the findings of Bosch (2012), Raju and Rani (2017) and Prakash (2023). The blooming period of *Urena lobata* was from July to January across the study area. This observation was similar with the study of Mukherjee (1969), Naidu et al. (2005) and Buragohain (2011).

The result of the present study shows that the fruiting phenophases of the selected invasive species (*Ageratum conyzoides, Bidens pilosa, Chromolaena odorata, Mikania micrantha, Solanum torvum, Solanum viarum, Synedrella nodiflora, Triumfetta rhomboidea and Urena lobata*) across different forest types were in line with the findings of Mullahey et al. (1993), Welman (2003), Shen et al. (2016). Raju and Rani (2017), Sankara et al. (2019), Ghosh et al. (2022), Meitei et al. (2022) and Inuthai (2023).

In addition, the leaf fall phenology of the selected species (*Ageratum conyzoides, Bidens pilosa, Chromolaena odorata, Mikania micrantha, Solanum torvum, Solanum viarum, Synedrella nodiflora, Triumfetta rhomboidea and Urena lobata*) were coincides with the works of Call (1998), Das and Agarwala (2011), Mullahey (2012), Lalmuanpuii et al. (2020) and Ghosh et al. (2022).

The leaf initiation phenophases of the selected species (*Ageratum conyzoides, Bidens pilosa, Chromolaena odorata, Mikania micrantha, Solanum torvum, Solanum viarum, Synedrella nodiflora, Triumfetta rhomboidea and Urena lobata*) were corresponded with the study of Call (1998), Cao et al. (2003), Ekeleme et al. (2005), Ghosh (2006), Das and Agarwala (2011), Mullahey (2012), Girija et al. (2015), Chauhan et al. (2019), Lalmuanpuii et al. (2020).

5.6.2 Phenological Variations Between the Forests

Variation of phenophases on the selected invasive species across the study site had been observed in study area which is in accordance with the findings of Beniwal (1987) in Arunachal Pradesh. This might be due to phenological phenomenon which varies depending upon humidity and temperature. In addition, influence of high relative humidity and low temperature might have played a role in difference of phenology (Beniwal, 1987). Moreover, variation between the phenophases i.e., *Ageratum conyzoides* in flowering and *Chromolaena odorata* in leaf initiation across the forest types was also observed in Pakke Tiger Reserve which concurred with the results of Mohanta et al. (2020). The difference between phenophases is the outcome of phenological behaviour of different forest types differed from species to species, elevation to elevation and forest type to forest type as well as the effect of precipitation and maximum temperature might also be the reason for the variation between the forest types (Beniwal, 1987; Mohanta et al., 2020).

5.6.3 Phenology in Relation to Climatic Parameter

5.6.3.1 Linear Regression

The relationship of phenological patterns to climatic parameters is mirrored by linear regression. The results of the study clearly show that climatic parameters were related with the phenological phenomena of the selected invasive species. Precipitation, relative humidity, minimum and maximum temperature were significantly associated with the plant phenophases across the different forest types. In flowering phenology, all climatic parameters influenced the flowering in some of the invasive species. Similar climatic influence on flowering phenophases were also found in the study of Williams and Abberton (2004), Lesica nad Kittelson (2010) and Fernandes et al. (2015). These influences led to earlier onset of flowering in invasive in comparison to native species (Godoy et al., 2009; Colautti et al., 2017).

However, in fruiting phenology, relative humidity does not have influence over the invasive species but precipitation and temperature (minimum and maximum) do in Pakke Tiger Reserve. These findings correlate with the report of Parry et al. (2007) and Ullah et al. (2022). The climatic parameters i.e., precipitation and temperature help in formation and development of fruit in the species (Tutin and Fernandez, 1993; Hatfield and Prueger, 2015). Conversely, relative humidity regulates the pace at which seeds spread in fruiting phenology (Van der pijl, 1972), rather than influencing the phenophases. The phenophases of leaf fall display their association with relative humidity and minimum temperature in Pakke Tiger Reserve. This finding is also supported by other lines of evidence (Zhang et al., 2015; da Silva et al., 2018; Camarero and Rubio-Cuadrado, 2024; Lang et al., 2024). Both of this climatic parameter delayed leaf fall, increasing the duration of phenophases (Taylor et al., 2008; Lee et al., 2022). Consistent with other studies (Frankie et al., 1974; Hegarty, 1990), the present study also found that precipitation and maximum temperature does not have significant influence in the phenology of invasive species. In contrast to present findings, some studies have reported their influence in plant phenology (Barrett and Brown, 2021; Negi et al., 2022; Tang et al., 2023; Nasja et al., 2024). In addition, leaf initiation phenophases were closely related with all the climatic parameters across the different forest types of Pakke Tiger Reserve. These significant associations which were observed in present study were also reported by O'Hare (2002), Jaroenkit et al. (2012) and Zhang et al. (2015). Temperature (minimum and maximum) and humidity initiate early leaf flushed in the invasive species by increasing the length of phenophases (Koptur et al., 1988; Fogelstrom et al., 2021). Likewise, precipitation also helps in early flushing of leaf in the study area which coincides with the study of Jolly et al. (2004) and Jaroenkit et al. (2012).

5.6.3.2 Correlation

The present study had shown significant correlation between the phenophases and environmental variables.

5.6.3.3 Correlation between Flowering Phenophases and Environmental Variables

The correlation between flowering and minimum temperature displays significant correlation in five species (*Ageratum conyzoides, Bidens pilosa, Chromolaena odorata, Mikania micrantha* and *Solanum viarum*). Negative correlation and positive correlation with minimum temperature, maximum temperature and precipitation were observed in the present study, which is in accordance with the study of William and Aberton (2004), Von Holle et al. (2010), Tiwari and Chandra (2017) and Mohandass et al. (2018). Minimum temperature had a significant correlation with the flowering phenophases because it is main climatic parameter for inducing and accelerating the flowering in many plants, with the regulation of "florigen" which is present in the leaves (Wang et al., 2022). When florigen reaches the shoot apical meristem, it is synthesized in the leaves and activate the floral transition (Tsuji, 2017). Later, through the vernalisation process, plant changes from sprouting to sexual growth under the influence of minimum temperature (Khodorova and Boitel-Conti, 2013). Maximum temperature also had significant association with the flowering of invasive alien plant species in the present study. It leads to early flowering and faster growth in plants which might due to gene expression and pigment synthesis such as Chalcone synthade (Khodorova and Boitel-Conti, 2013; Hedge et al., 2020). Furthermore, precipitation (rainfall) plays an important role in the flowering phenophases (Godoy et al., 2009). Precipitation triggers early flowering due to their interaction with edaphic factors leading to more resource assimilation of higher nutrient levels (Lesica and Kittelson, 2010). Moreover, it interacts with the timing of vernalisation to initiate the allele and transcription factors, which ultimately lead to flowering (Caicedo et al., 2004). However, the relation with relative humidity displays only positive correlation in the study which is also similar with the findings of Mortensen (2000). Humidity increases the number of flower and flower buds by directly influencing the plant water relation, accelerating photosynthetic activity and rate of assimilation leading to shorter vegetative growth and early flowering in plant (Mortensen, 1986).

5.6.3.4 Correlation Between Fruiting Phenophases and Environmental Variables

The fruiting phenophases display negative correlation with minimum temperature, maximum temperature and precipitation in the present study. These findings were similar with the result of Tutin and Fernandez (1993), Cortes-Flores et al. (2013), Sobrinho et al. (2013), Nakar and Jadeja (2015) and Khound and Barua (2016). Moreover, minimum temperature influences the formation of fruit size (Tutin and Fernandez, 1993) while the higher temperature increases the fruit size and soluble solid in the plant (Hatfield and Prueger, 2015). In addition, rainfall also influences fruiting seasonality (Van Schaik et al., 1993) and germination success (Rathche and Lacey, 1985). However, humidity shows both positive and negative correlation in the present study which were in line with the findings of Tiwari and Chandra (2017). Relative humidity assists the initiation of fruiting (Stamets, 2011) and impact on the fruit body by increasing the seed set without affecting fruit set (Moore et al., 2008).

5.6.3.5 Correlation between Leaf Initiation Phenophases and Environmental Variables

In the present study, leaf initiation shows positive correlation with minimum temperature and precipitation. The derived result of this study is consistent with the result of Singh et al. (2023). This might be due to invasive species having weaker winter chilling requirement or strong response to spring warming or photoperiod process (Polgar et al., 2014). Moreover, winter chilling, spring warming and photoperiod deemed essential prerequisites for early leaf initiation in invasive species and often leaf initiate earlier than native vegetation (Xu et al., 2007; Polgar et al., 2014). Moreover, precipitation triggers leaf initiation by breaking of vegetative buds in some of the species (Kushwaha et al., 2010).

In addition, maximum temperature and humidity display both positive and negative correlation. These findings were concurrent with the results of Williams-Linera (1997), Jaroenkit et al. (2012), Singh et al. (2023) and Nasja et al. (2024). With the higher temperature, the invasive plant might have additional advantages over the native flora (Polgar et al., 2014) though leaf initiation to maximum temperature is species specific (Cleland et al., 2007). It also increases the length of growing season in a plant and the capacity to compete for nearby resources (Fridley, 2012), which might lead to alteration in species composition at the community level ((Polgar et al., 2014) and forces early bursting of bud and leaf initiation in the plant (Vitasse et al., 2009). Furthermore, humidity also activates the bud break and initiates early leaf out in the plant (Laube et al., 2015).

5.6.3.6 Correlation between Leaf Fall Phenophases and Environmental Variables

Leaf fall is regarded as significant yearly event in forest (Zalamea and González, 2008), which help in determining the plant functional types (Chapin et al., 1996). Leaf fall is negatively correlated with minimum temperature and humidity. The results of the present study is in accordance with the study of Williams-Linera and Tolome (1996) and Williams-Linera (1997), Nanda et al. (2014) and da Silva et al. (2018). Under the minimum temperature, leaf fall is delayed due to post harvest inhibition by declining antioxidant enzyme and transcriptional activities (Lee et al., 2022;). Moreover, minimum temperature helps in maintaining prolong shelf life and slowing down of leaf aging (An et al., 2020). Similarly, humidity also delays leaf fall because of phytohormones, leaf sugar content and source–sink status of the plant (Taylor et al., 2008). However, positive and negative correlation is found in maximum temperature and precipitation which concur with the report of Williams-Linera and Tolome (1996), Williams-Linera (1997) and Nanda et al. (2014). The maximum temperature induces early leaf fall in some plants due to accumulation of soluble sugars (De la Haba et al., 2014). Additionally, precipitation also induce early leaf fall as they were directly regulated (Yu et al., 2022).

5.6.3.7 Canonical Correspondence Analysis and Climatic Parameters

CCA is an efficient ordination technique that produces a multivariate gradient analysis and helps in visualizing the relation between the species and environmental variables (Ter Braak, 1986, 1987). Moreover, multicollinearity in the species data does not interfere with CCA, more species can be evaluated than sites (Ter Braak, 1986). This analysis displays an ordination diagram which had both species and environmental variables explaining their relationship with each other. Therefore, CCA had been used in the present study to describe the relation between the species phenophases (flowering, fruiting, leaf fall and leaf initiation) and environmental variables (precipitation, relative humidity, minimum and maximum temperature). According to CCA results, all the climatic parameters i.e., precipitation, relative humidity, minimum and maximum temperature influenced the various phenophases (flowering, fruiting, leaf fall and initiation) occurrences of invasive species. Abbad and Benchaabane (2004), Inouye et al. (2003), Lesica and Kittelson (2010), Hegazy et al. (2012) and Khan et al. (2018) studied the phenological patterns and conclude that phenological behaviours of plants were significantly influenced by climatic parameters. Similar findings have been observed in the present study in which precipitation, relative humidity and temperature (minimum and maximum) together tended to accelerate the flowering, fruiting, leaf fall and initiation in these invasive plants (Morais and Freitas, 2015; Ullah et al., 2022).

5.7 Conclusion

The results obtained represent important information to understand the phenological behavior of invasive alien plant species in the eastern Himalayan region. This research reveals that the eastern Himalayan region is more vulnerable to invasion due to climate change. The selected invasive species modify phenological patterns to increase their geographical range under the influence of climate conditions in the study area. Moreover, precipitation, relative humidity, minimum and maximum temperature had a vital effect on their distribution in Pakke Tiger Reserve. The study had demonstrated that flowering, fruiting, leaf initiation and leaf fall phenophases of invasive species modify their phenology plasticity. The findings give a vast avenue for future research and help to anticipate the behavior and proliferation of invasive species in eastern Himalayan region. This study will be highly significant and valuable to widen the knowledge of phenological response in the context of observed and anticipated climate change. The outcomes of present study revealed only the annual response of invasive phenophases to climatic parameters. To gain a better understanding of more phenological response, long term monitoring of phenology should be undertaken in in future studies. This will offer a more thorough knowledge of the consequences of climate change on invasive species.

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Chapter 6:

Ecological Impact of Invasive Alien Plant Species on Native Flora

6.0 Introduction

IAPs have a significant impact on the ecosystem by altering the diversity of native population, productivity and fluxes of soil properties and nutrients pools (Alvarez and Cushman, 2002; Blank and Young, 2002; Wilsey and Polley, 2006; Pysek et al., 2012; Barney et al., 2015; Tekiela and Barney, 2017) and can changes the key biological process discretely (Ehrenfeld, 2003; Levine et al., 2003; Vardien et al., 2012). Furthermore, impacts of IAPs on the native vegetation have been widely recognized worldwide but lack the thorough knowledge through which they proliferate successfully (Li et al., 2006). In order to address this matter, it is crucial to comprehend the approaches by which the IAPs affect the populations they invade (Parker et al., 1999; Levine et al., 2003).

Majority of the IAPs studies focused on the impact of the above ground terrestrial plant communities (Macdonald, 1991; Levine et al., 2003; Hejda and Pysek, 2006; Brown et al., 2006; Hejda et al., 2009; Meffin et al., 2010) as compared to below ground structures which might be due to methodological limitation (Belnap and Philips, 2001). However, IAPs often had more potential for expansion which resulted in ecological impact on both above and below ground storage and nutrient exchanges (Liao et al., 2008; Te Beest et al., 2015). In recent years, few studies of IAPs impact on below ground ecosystem have garnered significant global interest among the invasion biologist (Simba et al., 2013; Baranova et al., 2017; Cuda et al., 2017). Such studies need urgent prominence as the plant and soil are tethered to each other and any changes due to IAPs in native vegetation would causes further changes in soil properties of the native vegetation (Vitousek, 1990; Timsina et al., 2011). IAPs also change the soil conditions in such a way that facilitate further incursions in invaded ecosystem (Simberloff and Von holle, 1999; Richardson et al., 2000; Callaway et al., 2001). Moreover, they not only alter the physicochemical properties but also the displayed profound impact on abundance, composition and soil microbial communities creating a favorable condition for IAPs over native plant (Klironomos, 2002; Duda et al., 2003; Callaway et al., 2004). Some IAPs positively influenced soil physicochemical properties resulting in notable changes in the characteristics of the soil (Windham and Lathrop, 1999; Stefanowicz et al., 2018; Kumar et al., 2021).

Moreover, alteration of soil is often associated with the changes in decomposition and litter fall which connect the above and below processes (Ehrenfeld, 2003; Pandey et al., 2014). Though IAPs invasion have led to changes in litter quality and quantity as well as decomposer communities (Kourtev et al., 2002; Liao et al., 2008; Kone et al., 2012) resulting in negative effects on many environments (Wei et al., 2017). According to Ehrenfeld (2010) and Simberloff et al. (2013), IAPs invasion may be due to site- specific and effects might depend on the soil properties, especially fertility in the incursion region (Funk and Vitousek, 2007). Several studies have shown that IAPs have many mechanisms which help in alteration of soil properties (Tererai et al., 2015) by shifting the species composition (Jeddi et al., 2009), favouring more nutrient uptake than native in the invaded soil (Yelenik et al., 2004). Also, the IAPs impacts on the dynamics of soil system were varied and included changes in key elements such as carbon, nitrogen and other nutrient pools, as well as physicochemical properties modification (Chen et al., 2015; Soti and Jayachandra, 2016; Afreen et al., 2018). Under the influence of IAPs, some studies reveal that biotic and abiotic characteristics of soil changes such as faunal and microbial communities, nutrients availability, moisture and pH (Chapin et al., 2002; Elgersma and Ehrenfeld, 2011; Suding et al., 2013) resulting in increased of organic carbon, total nitrogen, phosphorus, soil pH, biomass, rates of decomposition and repellence of soil water (Fan et al., 2010; Vila et al., 2011; Ruwanza et al., 2013; Sardans et al., 2017)., Some studies shows negative effect in soil properties, when IAPs comes in contact with the recipient environment (Christian and Wilson, 1999; Leary et al., 2006; Tererai et al., 2015). Furthermore, the effects of IAPs vary greatly depending on particular region (Osunkoya and Perrett, 2011). Therefore, it had been recommended to study the impact of IAPs conducted using condition unique to particular species and region.

However, in eastern Himalayan region of Arunachal Pradesh, no comparative studies of invasive alien plant species on physicochemical properties of soil have so far investigated from this region in regional or global scale. Therefore, the present study is undertaken to understand the impact of invasive alien plant species on the dynamics system of physicochemical properties of the soil in the eastern Himalayan of Indian Himalayan region.

6.1 Methodology

6.1.1 Study Area

The Pakke Tiger Reserve (PTR; 26^{o}55’ to 27^{o}15’E: 92^{o}35’ to 93^{o}10’N; Figure 6.1), Arunachal Pradesh lies in the north eastern region of eastern Himalaya. The eastern Himalaya owing to the beautiful realms of the oriental and the Indo-Malayan region have been considered as one of the biodiversity hotspot regions and one of the ecoregions in the world (Myers et al., 2000; Brooks et al., 2006). Pakke Tiger Reserve is located in the western part of sloping and rugged mountains region of Arunachal Pradesh in the Eastern Himalayas, specifically in the newly declared district of Pakke Kessang. PTR covers a total geographical area of 861.95 km^2, which accounts for 9.04% of the total protected areas of this eastern Himalayan state. Moreover, it covers 45% of the total areas of the newly formed Pakke Kessang district (1932 km^2) of the Arunachal Pradesh state. The PTR is completely circled from all sides-Papum Reserve Forest on the east, Doimara Reserve Forest and Eaglenest Wildlife Sanctuary on the west, Tenga Reserve Forest on the north and Nameri Tiger Reserve on the south, along with two rivers, Bhareli or Kameng River on the west and Pakke River on the east. The reserve is governed by three Range offices namely Seijosa Range office, Tippi Range office and Riloh Range office, with their own area demarcated for jurisdiction. The terrain of the reserve is very steep and continuously rugged on the sides of Tippi range and Riloh range as compared to Seijosa range which have plain and moderate sloping hills. The reserve spans altitudes ranging from 150 to 2050 m above the sea level. The reserve is home to different varieties of birds (296), butterflies (284), snakes (46), mammals (60) and many other small species (Datta and Goyal, 1997; Datta et al., 1998; Sondhi and Kunte, 2014; Rambia and Rathore, 2022).

According to Champion and Seth (1968) the reserve vegetation is classified as the Assam valley tropical evergreen forest with the average annual rainfall of 2500 mm, predominantly falling in the month of May to September. The temperature varies from 12° to 36° C with cold weather starting from December to February and May to June is the hottest month. In the south eastern side of the reserve, more than 20 villages and small settlements adjacent to the Pakke River with a population of more than 4000 adults’ villagers were located (Datta and Goyal, 1997). Because of its location in the oriental and Indo Malayan region, the area had a rich biological diversity in both flora and fauna. Moreover, the steeped terrain of the reserve restricts the easy accessibility inside the core area.

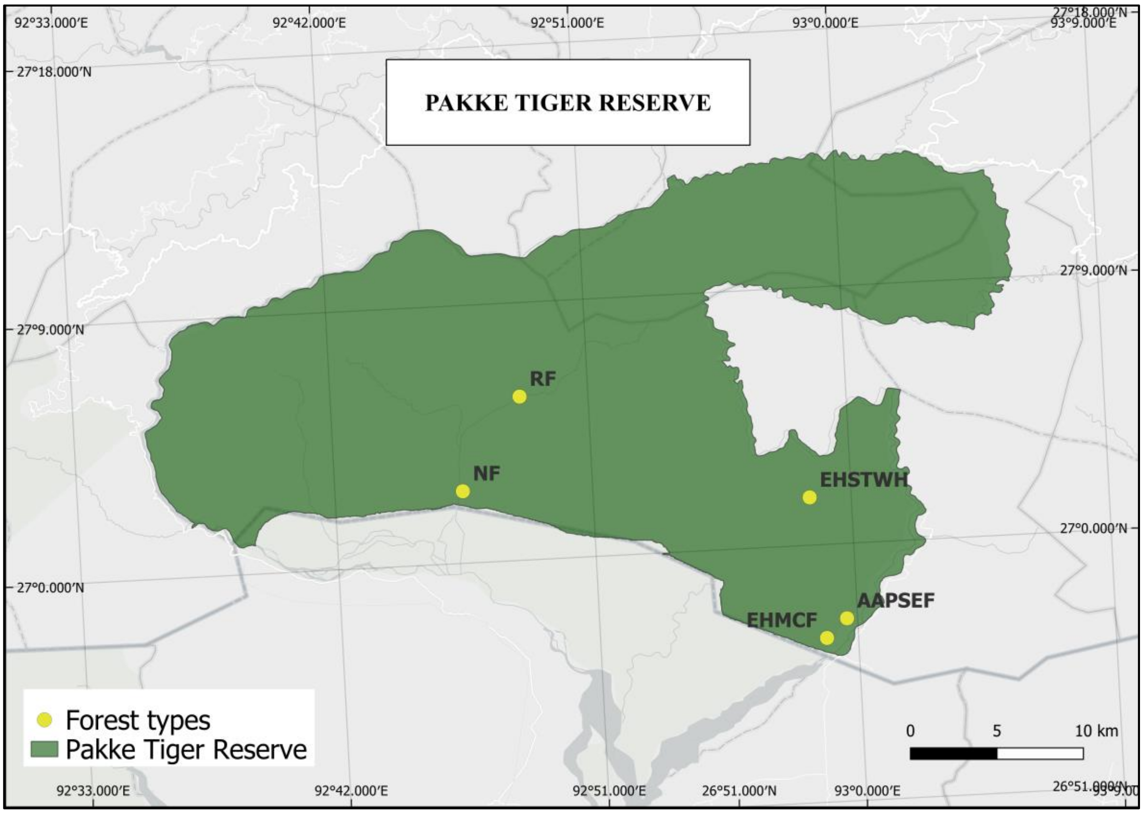


Figure 6.1: Map of the study areas showing the location point of sampling sites in Pakke Tiger Reserve.

Table 6.1. Detail information about the sampling sites with status, location and vegetation in PTR. *IN- Invasive, UN- Un-invasive, AAPSEF- Assam Alluvial Plain Semi Evergreen Forest, EHMCF- East Himalayan Mixed Coniferous Forest, EHSTWH- the East Himalayan Subtropical Wet Hill, NF- Non-Forest and RF- Riverine Forest.

Forest types	**Status (IN /UN)**	**Coordinates**		**Altitude (m)**	**Location**
	Invaded	26.954056	92.993444	149	West Bank
AAPSEF	Uninvaded	26.956389	92.993889	162	West Bank
	Invaded	26.950778	92.981222	234	West Bank
EHMCF	Uninvaded	26.945556	92.981667	163	West Bank
	Invaded	27.025722	92.981778	857	Laling camp
EHSTWH	Uninvaded	27.027833	92.975833	930	Laling camp
	Invaded	27.056222	92.776222	144	Bhalu camp
NF	Uninvaded	27.041944	92.774444	137	Bhalu camp
	Invaded	27.088056	92.796389	187	Kachuwa Matha
RF	Uninvaded	27.095278	92.810278	211	Kachuwa Matha

6.1.2 Species Selection

For the present study, the dominantly distributed species was considered for observation i.e, *Chromolaena odorata*, as these species was distributed in five forests at contagious level in the study area (Photo plate 6.1). Moreover, the selected species was more densely distributed and their proliferation was more than the rest of selected species in chapter 4 and 5. Considering their invasiveness and the results observed from the chapter 4, these species was selected for the present chapter.

Chromolaena odorata (L.) R.M. King & H. Rob. (hereafter *C. odorata*) (Asteraceae), commonly known as “Siam weed”, “Christmas weed” or “bitter brush” is a globally known invasive scrambling shrub (King and Robinson, 1970**;** Howard 1989; Lioger, 1997), native to tropical and subtropical America (Koutika and Rainey, 2010) and second most abundant invasive alien plant species (Robertson et al., 2003). *C. odorata* is reported as one of the global worst weed invader (Lowe et al., 2000), which have now invaded five continents out of seven continents in the world (Kriticos et al., 2005**;** Mandal and Joshi, 2014a**)**. With the ability to withstand different soil and climatic conditions in recipient environment (Muniappan et al., 2009; Uyi et al., 2014), this species can quickly be established and overwhelm the native vegetation (McFayden and Skarratt, 1996**;** Ohtsuka, 1999). These species have been reported as one of the worst terrestrial invasive plants in the humid tropics and subtropics regions of the world (Holm et al., 1977; Gautier, 1992) and well known for invading the secondary habitats i.e., agricultural lands, forest clearings and along the trails, roadsides and forest margins (Rao, 1977; Azmi, 2002; CABI, 2024).

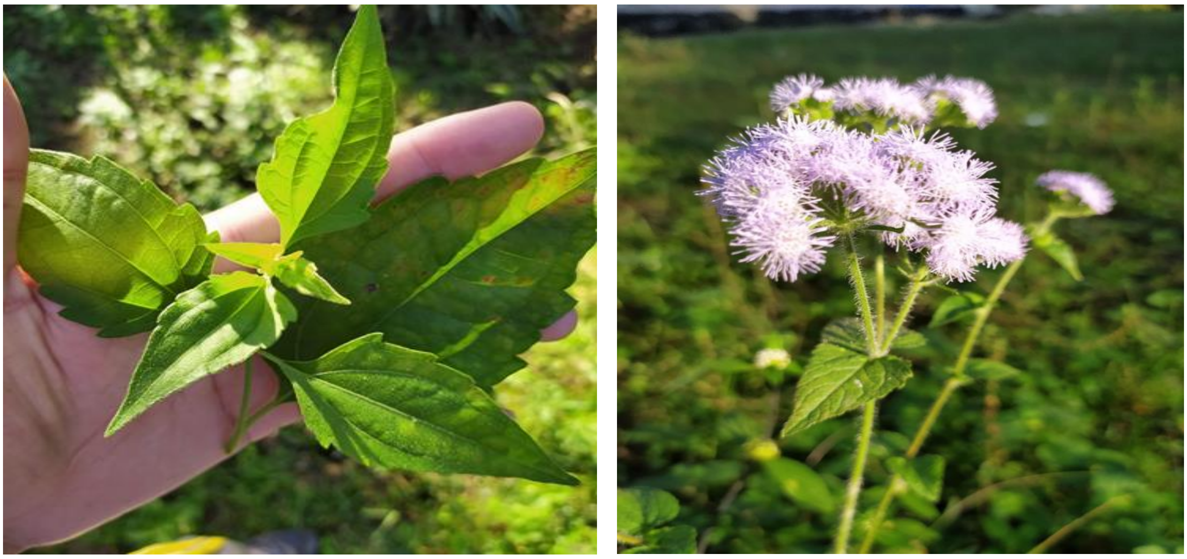


Photo plate 6.1: Floral and leaf initiation of *Chromolaena odorata*

6.1.3 Sampling Site and Soil Collection

The present study was conducted in the core area of the Pakke Tiger Reserve and the study site is located on five different forest types i.e., Assam Alluvial Plain Semi Evergreen Forest (AAPSEF), the East Himalayan Mixed Coniferous Forest (EHMCF), the East Himalayan Subtropical Wet Hill Forest (EHSTWH), the Non-Forest (NF) and the Riverine Forest (RF; Champion and Seth, 1968; FSI, 2009; Table 6.1). The study is rugged mountainous in the north and narrow plain and steeping sides in the south with little bit of slope southward toward the river valley area. The elevational of the study site varies from one forest types to another and soil of the different forest is described as loamy to sandy loamy soil. In these study area, structure of the forest is a typical layered with *Tetrameles nudiflora* and *Ailanthus integrifolia* forming the dominant species in the study area (Singh, 1991). Moreover, different types of tree species dominate at each level such as *Pterospermum acerifolium, Sterculia villosa, Monoon simiarum, Duabanga grandiflora* and *Stereospermum chelonoides* dominating the lower plain and middle level of the forest whereas the species like *Castanopsis tribuloides, Castanopsis indica* and *Mesua ferrea* dominate the upper level of the forest region. With high diversity of trees, shrubs, herbs, lianas and climbers, the vegetation of the forest is very dense. Additionally, native species such as *Clerodendrum colebrookeanum, Melastoma malabathricum, Phlogacanthus curviflorus, Strobilanthes cusia, Osbeckia nutans, Hellenia speciosa, Lepidagathis incurva, Phanera vahlii* and *Poikilospermum suaveolens* covers the ground layers of the study area. Detailed information regarding about the location sites is given in Table 6.1.

Extensive field study was conducted in each of selected five different forest types to know about the distribution and peak growing season of *Chromolaena odarata* before the collection of soil from the study area. During the month of September, the collection of soil samples following Lawrence et al. (2016) was carried out in all selected site (invaded and uninvaded) of five different forest types of the Pakke Tiger Reserve. In each forest types, six sites were selected randomly for soil collection; one with high invasion of *Chromolaena odorata* (>70% cover percentage targeted) and other adjacent to the uninvaded sites with same cover percentage of native population were selected for the study (Ruwanza and Shackleton, 2016; Fig. 6.1). At each soil sampling sites, six 1x1m in invaded and another six 1x1m in uninvaded vegetation were laid. Thus, twelve soil samples per sites were collected to a depth of 0-15 cm using a soil auger after the removal of overlaying debris from the forest floor (Photo Plate 6.2**)**. Hence, a total of 60 samples from the five forest types (invaded and uninvaded sites) resulting six soil samples per site both in invaded and uninvaded were collected.

Photo Plate 6.2: Collection of soil from *C. odorata* invaded and uninvaded area.

All the soil samples were thoroughly mixed and pooled in the zip-locked plastic bag and kept in cold temperature. In each zip-locked polyethene bag, all the necessary details were patched on the cover such as site location with GPS point, habitats types, date, time and elevation (Photo plate 6.3). After the collection of soil from the sites of each forest types, unwanted debris, stones, plant materials and macro fauna were removed from the soil. In lab, the soil samples were dried and sieved by 2 mm mesh screen for the detailed physicochemical analysis, were then stored in a refrigerator at – 4° C until further analysis.

6.1.4 Soil Properties and Analysis

The soil pH was determined using a digital pH meter (model LT 50, Labtronics) with soil distilled water (1:2.5 w:v) at 25° C (FAO, 2021) and electrical conductivity (EC) was measured by a digital conductivity meter (model CM 183 EC-TDS, Elico) at 25°C respectively. Water holding capacity (WHC) was determined by following Kirkham (2005) and soil texture was measured using the hygrometer method following Okalebo et al. (2002). The organic carbon (OC) was determined according to volumetric method following Walkley and Black (1934), nitrogen (N) by Kjeldahl method (Subbiah and Asija, 1956), available phosphorus (P) by Olsen method (Olsen et al., 1954). Available potassium is determined by using flame photometer following Hanway and Heidal (1952), available sulphur was done by extraction of mono-calcium phosphate following William and Steinberg (1959) and available boron was determined by using hot water extraction and colorimetric method following John et al. (1975). Furthermore, micronutrients such as Copper (Cu), Iron (Fe), Zinc (Zn) and Manganese (Mn) were determined by using double beam atomic absorption spectrophotometer (model AAS4141, Ecil).

The data were analysed by using Microsoft excel, Paleontological Statistics (PAST) software, version 5.1 (Hammer et al., 2001) and R-studio software, version 4.2.2. All the data of the samples of the soil physicochemical properties were taken in a single month of September in order to avoid repeated measure analysis (Debnath et al., 2018). The comparison between invaded plots and uninvaded plots of the five different forest types in Pakke Tiger Reserve were analysed by using *t*- test. Also, the data was subjected to Analysis of Variance (ANOVA) to understand the differences in soil properties between invaded and uninvaded across all the sites in five different forest types. Moreover, similarities among the invaded and uninvaded sites were measured through Bray-Curtis similarity index following Simba et al. (2013) and Testoni et al. (2022). Pearson correlation Varimax rotation was used to standardize the data in order to optimize the association between soil parameters and principal components (PC; Simba et al., 2013), as the soil properties with loading greater than 0.33 were considered to be related with the principal components (PCs) according to Kothari (2004). Also, the number of principal components (PCs) was chosen on the basis of Broken Stick which retains only those components whose eigenvalue value is larger than the value provided by the distribution of broken stick value (MacArthur, 1957).

6.2 Results

The present study shows the noteworthy effect of *C. odorata* on soil properties in the study area. There have been significant differences in the soil properties of invaded and uninvaded across the sampling sites in five different forest types.

6.2.1 Soil Physical Properties

The water holding capacity (WHC) differ significantly between the invaded and uninvaded plots in all three forest types namely, AAPSEF, EHMCF and NF (*P=0.04*, *0.02* and *0.02*), while EHSTWH and RF shows non-significant (Table 6.2), with higher percentage in uninvaded sites of AAPSEF, EHSTWH, NF and RF except for the EHMCF, which have high percentage in invaded site. Correspondingly, clay, silt and sand also varied across the forest types. The percentage of clay show significant between the EHMCF, EHSTWH and NF (*P=0.02*, *0.03* and *0.05*), in contrast to AAPSEF and RF which exhibit no significant. Clay percentages were high in all invaded sites as compare to uninvaded sites across the forest types. The percentage of silt are not significantly differed in invaded and uninvaded site of EHMCF and EHSTWH but displayed significance in AAPSEF, NF and RF (*P=0.04, 0.02* and *0.04*) and have high percentages in invaded sites of four forest types except for EHMCF, which have high percentage in uninvaded (Table 6.2). Moreover, the percentage of sand show non-significant in EHMCF, EHSTWH and NF, though AAPSEF and RF revealed their significant in the sampling site (*P=0.03* and *0.03).* However, the sand percentages were high in uninvaded sites of four forest types in comparison to EHMCF, with high percentage in invaded site (Table 6.2).

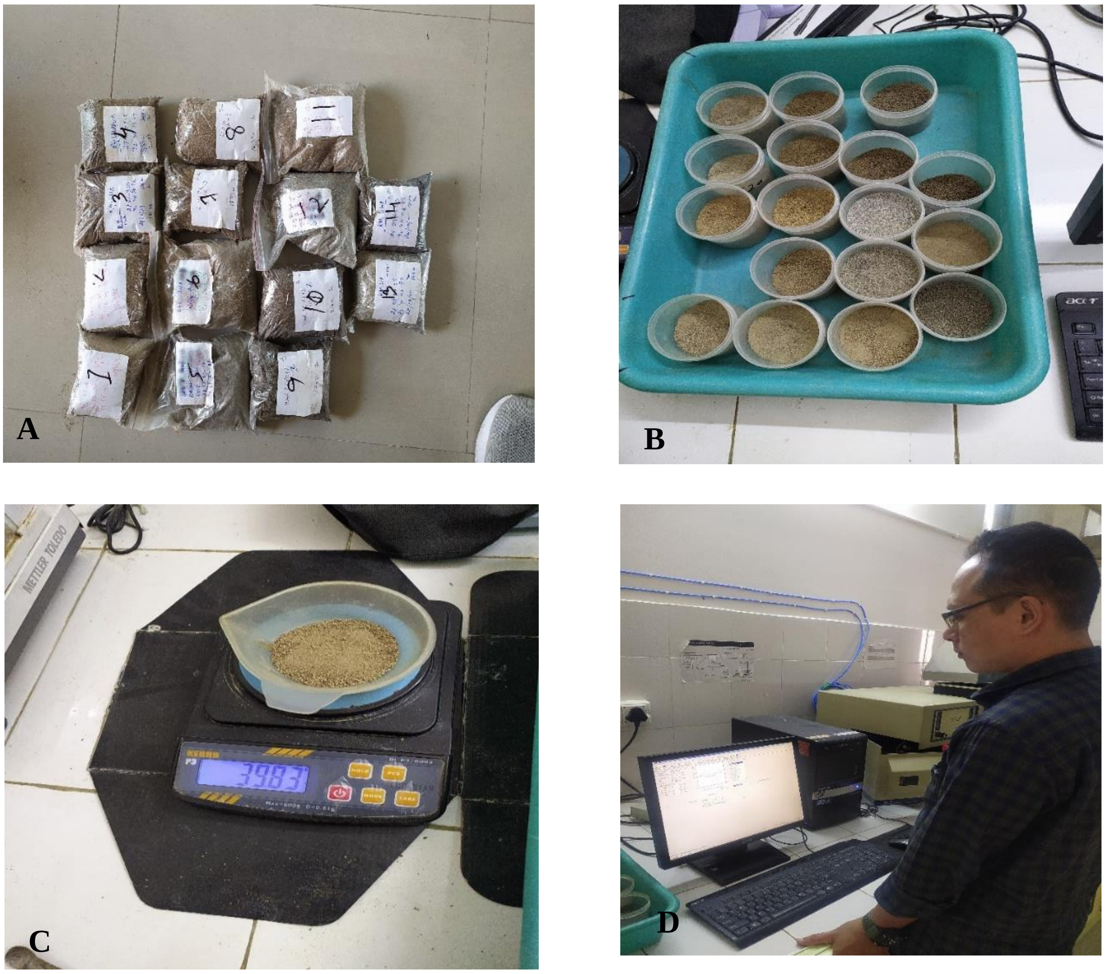


Photo Plate 6.3: (A) Ziplock polythene for keeping soil sample (B) After sieving the soil sample (C) Weighing of soil sample and (D) Atomic Absorption Spectrophotometer

6.2.2 Soil Chemical Properties

The soil pH across the sampling sites were quite acidic in all invaded and uninvaded plots with an average of pH 4**.** Moreover, there is not significant difference in AAPSEF, EHSTWH and RF while EHMCF and NF (*P=0.02* and *0.02)* displayed significant between the invaded and uninvaded in the study area **(**Table 6.2). Moreover, the average value of uninvaded site was high in AAPSEF, EHMCF and EHSTWH while NF and RF have high average value in invaded sites across the study sites. The electrical conductivity (EC) of the soil was high in uninvaded sites of AAPSEF and EHSTWH while EHMCF and RF show higher in invaded sites. The soil electrical conductivity significantly differs in AAPSEF, EHMCF, EHSTWH and RF *(P=0.007, 0.03, 0.02* and *0.003*) except for NF showing non-significant in the sampling site. At four sites (AAPSEF, EHMCF, NF and RF), soil organic carbon (OC) and available nitrogen (N) were higher in the soil invaded by IAPs, except for the EHSTWH. There was significant difference in four sites of soil OC (*P=0.02, 0.05, 0.04* and *0.008*) with no significant difference in AAPSEF. However, available N (*P=0.02, 0.006, 0.03, 0.02* and *0.03*) have shown significant differences in all sites. Similarly, available potassium (K; *P=0.04,0.02, 0.05, 0.05* and *0.04*) and available boron (B; *P=0.01, 0.04, 0.001, 0.01* and *0.01*)) exhibit significant difference in all sites. However, soil K was higher in invaded sites of AAPSEF, NF and RF but remaining sites display opposite trend. But in soil B, only AAPSEF was high in uninvaded site while the other shows higher in invaded sites of EHMCF, EHSTWH, NF and RF. Additionally, available phosphorus (P) was high in invaded sites of AAPSEF, EHMCF and RF but in EHSTWH and NF where it shows higher in uninvaded site with significance difference in four sites (*P=0.02, 0.04, 0.03* and *0.03*), except for EHMCF revealing non-significant. The available Sulphur (S) in the soil also revealed the significant difference in four sites of AAPSEF, EHMCF, EHSTWH and RF (*P=0.03, 0.03, 0.04 and 0.04*) while the NF display non-significant, though EHMCF, EHSTWH AND NF sites have high values in invaded as compare to AAPSEF and RF which were high in uninvaded sites. Furthermore, the soil micronutrients (Cu, Fe, Zn and Mn) show differences in site specific. The soil micronutrients, viz., Fe (*P=0.03, 0.02, 0.03, 0.02* and *0.04*) and Mn (*P=0.04, 0.007, 00.02, 0.04* and *0.03*) differ significantly between the invaded and uninvaded in all sites. However, Fe was higher in four invaded sites (AAPSEF, EHMCF, NF and RF) while EHSTWH show the opposite.

Table 6.2: Results of soil properties in the invaded and uninvaded plot across the five forest sites in the Pakke Tiger Reserve. Significance levels: *$p<0.05$= Significance; $p<0.001$=moderately significant; $p<0.0001$=strong significant and ns= not significant.

Soil parameters	AAPSEF			EHMCF			EHSTWH			NF			RF		
	Invaded	**Uninvaded**	***p- value***	**Invaded**	**Uninvaded**	***p- value***	**Invaded**	**Uninvaded**	***p- value***	**Invaded**	**Uninvaded**	***p- value***	**Invaded**	**Uninvaded**	***p- value***
Soil pH	4.13±0.28	4.49±0.16	ns	4.06±0.14	4.91±0.27	0.02	4.50±0.45	4.97±0.15	ns	4.24±0.29	4.07±0.32	0.02	4.18±0.38	3.98±0.14	ns
Electrical Conductivity (dS/m)	0.31±0.16	0.65±0.05	0.007	0.43±0.04	0.32±0.02	0.03	0.53±0.03	0.82±0.09	0.02	0.47±0.06	0.47±0.09	ns	0.60±0.03	0.39±0.04	0.003
Organic Carbon (%)	0.74±0.04	0.69±0.10	ns	1.56±0.29	0.79±0.17	0.02	1.15±0.38	1.96±0.39	0.05	0.88±0.04	0.45±0.15	0.04	0.96±0.05	0.47±0.12	0.008
Phosphorus (Kg/h)	26.59±4.78	13.08±2.27	0.02	33.74±10.35	29.61±9.62	ns	14.42±4.56	27.83±3.24	0.04	16.16±2.93	26.30±3.02	0.03	37.78±5.95	19.13±4.85	0.03
Potassium (Kg/h)	242.11±22.41	174.06±19.84	0.04	164.27±21.23	253.50±27.06	0.02	81.48±6.66	188.98±44.01	0.05	143.17±27.60	75±6.75	0.05	172.82±28.04	103.29±16.08	0.04
Sulphur (ppm)	20.30±3.11	32.91±4.07	0.03	23.19±3.51	12.56±2.63	0.03	26.57±3.03	15.29±3.72	0.04	27.83±11.99	19.67±6.71	ns	11.79±1.81	23.27±4.16	0.04
Zinc (ppm)	1.76±0.16	1.10±0.24	0.05	1.46±0.21	0.75±0.22	0.04	0.82±0.14	1.81±0.39	0.05	1.73±0.23	0.83±0.31	0.04	1.29±0.29	0.88±0.29	ns
Boron (ppm)	1.47±0.06	1.02±0.13	0.01	0.93±0.13	1.65±0.26	0.04	1.05±0.10	2.11±0.19	0.001	0.94±0.10	1.32±0.05	0.01	1.17±0.16	1.88±0.15	0.01
Iron (ppm)	19.82±1.00	14.84±1.75	0.03	21.83±0.94	16.28±1.80	0.02	13.69±1.11	19.27±1.89	0.03	20.55±1.65	14.31±1.80	0.02	21.18±2.26	14.71±1.51	0.04
Manganese (ppm)	10.09±0.82	15.13±1.91	0.04	16.42±3.68	12.15±3.14	0.007	14.79±3.22	13.85±2.74	0.02	14.97±3.001	7.04±0.99	0.04	23.10±2.45	14.58±2.58	0.03
Copper (ppm)	0.93±0.04	0.68±0.09	0.04	1.34±0.42	0.95±0.11	ns	2.12±0.39	0.91±0.18	0.02	1.01±0.04	0.88±0.08	0.03	1.07±0.06	0.86±0.06	0.04
Available Nitrogen (Kg/h)	374.52±42.34	231.68±29.52	0.02	537.37±67.81	250.80±25.14	0.006	207.69±16.70	494.29±98.82	0.03	477.96±79.61	212.48±13.26	0.02	565.99±96.80	280.34±37.41	0.03
WHC (%)	51.66±5.32	66.63±3.57	0.04	57.7±8.54	48.48±9.39	0.02	57.18±9.41	70.58±2.87	ns	61.09±2.20	69.38±2.18	0.02	60.42±6.59	70.5±2.43	ns
Clay (%)	3.185±0.29	2.56±0.47	ns	3.25±0.26	1.80±0.55	0.04	2.95±0.39	1.71±0.33	0.03	2.61±0.16	1.40±0.47	0.05	3.055±0.47	1.82±0.48	ns
Silt (%)	21.46±1.30	15.56±2.10	0.04	11.27±1.14	13.58±2.73	ns	13.68±2.60	9.22±3.23	ns	18.55±0.99	12.71±1.85	0.02	18.08±1.88	11.42±2.26	0.04
Sand (%)	75.35±1.24	81±2.21	0.03	86.35±1.24	84.65±3.01	ns	83.35±2.59	89.02±3.13	ns	84.26±1.77	85.91±2.05	ns	78.86±1.80	86.74±2.66	0.03

The Mn in the soil also show difference between thee invaded and uninvaded sites, in which EHMCF, NF and RF were high in invaded sites though AAPSEF and EHSTWH exhibit high in uninvaded site. While the remaining micronutrients Cu ($P=0.04, 0.02, 0.03$ and 0.04) and Zn ($P=0.05, 0.04, 0.05$ and 0.04) differ significantly in four sites in AAPSEF, EHSTWH, NF and RF and the later in AAPSEF, EHMCF, EHSTWH and NF, respectively. However, Zn were higher in all invaded sites of the sampling plots while Cu were high in four sites i.e., AAPSEF, EHMCF, NF and RF, except for EHSTWH which show high in uninvaded site (Table 6.2).

6.2.3 Analysis of Variance (ANOVA) Of Soil Properties in Invaded and Uninvaded Sites

The results of the ANOVA have shown significant differences in the physicochemical properties between the invaded and uninvaded plots in the study sites (Table 6.3 and 6.4). The analysis of variance of soil pH showed significance difference among the uninvaded sites ($F=4.14; P= 0.01; P<0.05$) while non significance ($P>0.05$) among the invaded sites. In soil EC, both invaded ($F=5.606, P= 0.002; P<0.05$) and uninvaded sites ($F=7.279, P=0.0004; P<0.05$) have shown significant differences. However, OC have shown no significant variation in invaded sites ($P>0.05$) but significant differences in uninvaded sites ($F=8.148, P=0.0002; P<0.05$). The sites of K ($F= 6.563, P=0.0009; F=7.504, P=0.0004$), B ($F= 3.49, P=0.02; F=6.017, P=0.001$) and N ($F= 4.735, P=0.005; F=5.128, P=0.003$) have shown significant variation among the invaded and uninvaded site in the study area. The available P and S have a different result among the invaded and uninvaded sites, in which P ($F=2.752, P=0.05$) show only significant in invaded sites while S ($F=3.154, P=0.03$) in uninvaded sites. Moreover, the texture of soil i.e., silt ($F=5.796, P=0.001$) and sand ($F= 6.086, P=0.001$) have shown significant variation among the invaded sites but non-significant in uninvaded sites while clay represent non-significant ($P>0.05$) in all invaded and uninvaded sites. Similarly, the soil micronutrients, viz., Cu ($F=3.441, P=0.02$), Fe ($F=4.881, P=0.004$), Zn ($F=3.167, P=0.03$) and Mn ($F=2.77, P=0.04$) also showed significant differences in all invaded sites but non-significant ($P>0.05$) variation in uninvaded sites. Furthermore, the water holding capacity ($F=3.741, P=0.01$) display significant variation in uninvaded sites but non-significant in invaded sites (Table 6.3 and 6.4).

Table 6.3: Results of the analysis of variance (ANOVA) test for soil properties in the invaded (IN) plots across all the sampling sites in Pakke Tiger Reserve. Significance levels: p<0.05= Significance; p<0.001=moderately significant; p<0.0001=strong significant and ns= not significant.

Soil parameters	**Forest Types**					***F-value***	***p-value***
	AAPSEF	**EHMCF**	**EHSTWH**	**NF**	**RF**		
PH	4.13±0.28	4.06±0.14	4.50±0.45	4.24±0.29	4.18±0.38	0.259	ns
EC (dS/m)	0.31±0.16	0.43±0.04	0.53±0.03	0.47±0.06	0.60±0.03	5.606	0.002
OC (%)	0.74±0.04	1.56±0.29	1.15±0.38	0.88±0.04	0.96±0.05	2.082	ns
P (Kg/h)	26.59±4.78	33.74±10.35	14.42±4.56	16.16±2.93	37.78±5.95	2.752	0.05
K (Kg/h)	242.11±22.41	164.27±21.23	81.48±6.66	143.17±27.60	172.82±28.04	6.563	0.0009
S (ppm)	20.30±3.11	23.19±3.51	26.57±3.03	27.83±11.99	11.79±1.81	1.144	ns
Zn (ppm)	1.76±0.16	1.46±0.21	0.82±0.14	1.73±0.23	1.29±0.29	3.167	0.03
B (ppm)	1.47±0.06	0.93±0.13	1.05±0.10	0.94±0.10	1.17±0.16	3.49	0.02
Fe (ppm)	19.82±1.00	21.83±0.94	13.69±1.11	20.55±1.65	21.18±2.26	4.881	0.004
Mn (ppm)	10.09±0.82	16.42±3.68	14.79±3.22	14.97±3.001	23.10±2.45	2.77	0.04
Cu (ppm)	0.93±0.04	1.34±0.42	2.12±0.39	1.01±0.04	1.07±0.06	3.441	0.02
N (Kg/h)	374.52±42.34	537.37±67.81	207.69±16.70	477.96±79.61	565.99±96.80	4.735	0.005
WHC (%)	51.66±5.32	57.7±8.54	57.18±9.41	61.09±2.20	60.42±6.59	0.2916	ns
CLAY (%)	3.185±0.29	3.25±0.26	2.95±0.39	2.61±0.16	3.055±0.47	0.572	ns
SILT (%)	21.46±1.30	11.27±1.14	13.68±2.60	18.55±0.99	18.08±1.88	5.796	0.001
SAND (%)	75.35±1.24	86.35±1.24	83.35±2.59	84.26±1.77	78.86±1.80	6.086	0.001

However, the overall analysis of variance across the sampling sites displayed that eight of soil parameters (B, Fe, Cu, N, WHC, Clay, Silt and Sand) were found significantly difference (Table 6.5). The *P*- value revealed that boron and clay were strongly significant ($P < 0.0001$) while iron, copper, silt and nitrogen were moderately significant ($P < 0.001$). Additionally, water holding capacity and sand also shows significant variation between invaded and uninvaded across all the sampling sites ($P < 0.05$).

Table 6.4: Results of the analysis of variance (ANOVA) test for soil properties in the uninvaded (UN) plots across all the sampling sites in Pakke Tiger Reserve. Significance levels: $p<0.05$= Significance; $p<0.001$=moderately significant; $p<0.0001$=strong significant and ns= not significant.

Soil parameters	**Forest Types**					***F- value***	***p-value***
	AAPSEF	**EHMCF**	**EHSTWH**	**NF**	**RF**		
PH	4.49±0.16	4.91±0.27	4.97±0.15	4.07±0.32	3.98±0.14	4.144	0.01
EC (dS/m)	0.65±0.05	0.32±0.02	0.82±0.09	0.47±0.09	0.39±0.04	7.279	0.0004
OC (%)	0.69±0.10	0.79±0.17	1.96±0.39	0.45±0.15	0.47±0.12	8.148	0.0002
P (Kg/h)	13.08±2.27	29.61±9.62	27.83±3.24	26.30±3.02	19.13±4.85	1.695	ns
K (Kg/h)	174.06±19.84	253.50±27.06	188.98±44.01	75±6.75	103.29±16.08	7.504	0.0004
S (ppm)	32.91±4.07	12.56±2.63	15.29±3.72	19.67±6.71	23.27±4.16	3.154	0.03
Zn (ppm)	1.10±0.24	0.75±0.22	1.81±0.39	0.83±0.31	0.88±0.29	2.062	ns
B (ppm)	1.02±0.13	1.65±0.26	2.11±0.19	1.32±0.05	1.88±0.15	6.017	0.001
Fe (ppm)	14.84±1.75	16.28±1.80	19.27±1.89	14.31±1.80	14.71±1.51	1.338	ns
Mn (ppm)	15.13±1.91	12.15±3.14	13.85±2.74	7.04±0.99	14.58±2.58	1.866	ns
Cu (ppm)	0.68±0.09	0.95±0.11	0.91±0.18	0.88±0.08	0.86±0.06	0.871	ns
N (Kg/h)	231.68±29.52	250.80±25.14	494.29±98.82	212.48±13.26	280.34±37.41	5.128	0.003
WHC (%)	66.63±3.57	48.48±9.39	70.58±2.87	69.38±2.18	70.5±2.43	3.741	0.01
CLAY (%)	2.56±0.47	1.80±0.55	1.71±0.33	1.40±0.47	1.82±0.48	0.8273	ns
SILT (%)	15.56±2.10	13.58±2.73	9.22±3.23	12.71±1.85	11.42±2.26	0.9078	ns
SAND (%)	81±2.21	84.65±3.01	89.02±3.13	85.91±2.05	86.74±2.66	0.9741	ns

6.2.4 Cluster Analysis Among the Forest Sites

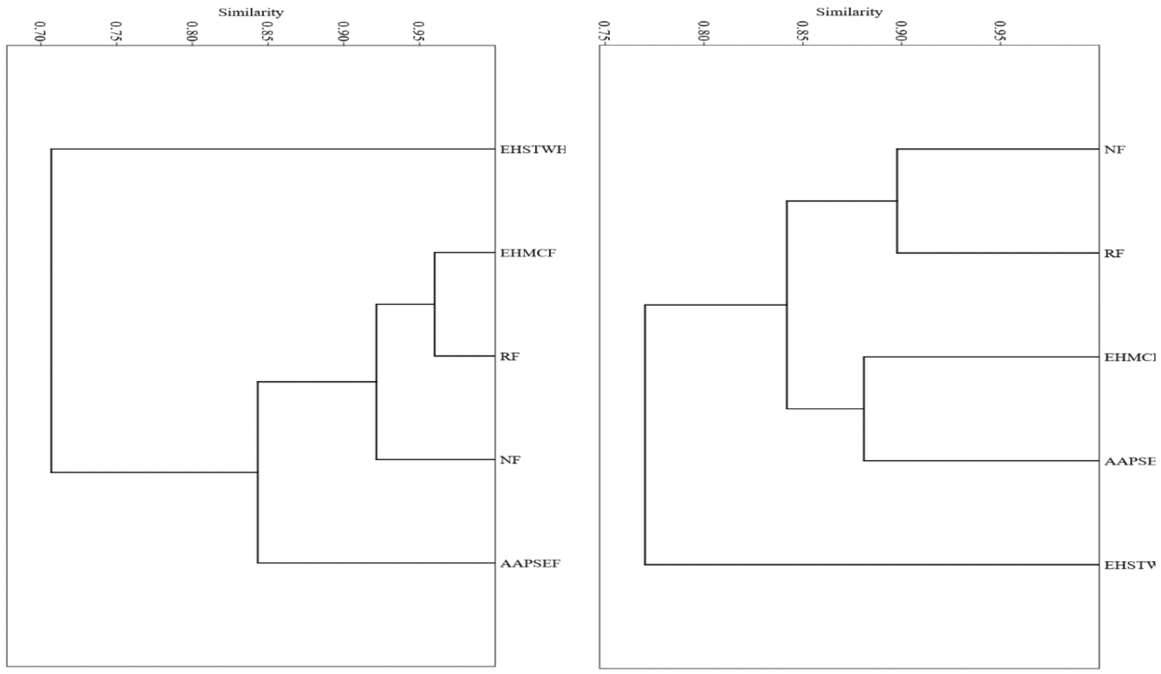
This analysis among the invaded and uninvaded sites clearly shows that EHSTWH is different in both sites indicating their variation in soil parameters (Fig. 6.2). The dendrogram of Bray- Curtis similarity index represents the similarity of soil parameters in each site ranging from 0 to 1 (1-highest similarity and near 0-greater dissimilarity). The cluster analysis at invaded sites (IN) with cophenetic correlation coefficient of 0.9753 using paired group (UPGMA- unweighted pair group method with arithmetic mean) reflects how the soil variables is between the forest types, more similar variables in forest types forming

Table 6.5: Results of the analysis of variance (ANOVA) test for soil properties in *C. odorata* invaded and uninvaded plots across all the forest sampling sites in Pakke Tiger Reserve. * EC-Electrical conductivity, OC- Organic carbon, P-Phosphorus, K-Potassium, S-Sulphur, Zn- Zinc, N-Nitrogen, B-Boron, Fe-Iron, Mn-Manganese, Cu-Copper, WHC-Water holding capacity Significance levels: $p<0.05$= Significance; $p<0.001$=moderately significant, $p<0.0001$=strong significant and Mean±StanD.

Soil variables	**Invaded (n=30)**	**Uninvaded (n=30)**	***p- value***
pH	4.16±0.76	4.17±0.71	0.9
EC (dS/m)	0.46±0.12	0.49±0.23	0.2
OC (%)	0.83±0.46	0.92±0.82	0.2
P (kg/h)	32.18±24.4	26.44±20.4	0.5
K (kg/h)	136±67.8	113.14±63.9	0.9
S (ppm)	25.44±29.6	18.43±14.7	0.7
Zn (ppm)	1.12±0.7	1.11±0.88	0.06
B (ppm)	1.15±0.3	1.54±0.46	**0.0001*** **
Fe (ppm)	20.13±4.6	16.85±4.4	**0.003** **
Mn (ppm)	14.85±7.7	11.78±6.8	0.07
Cu (ppm)	1.20±0.6	0.87±0.25	**0.003** **
N (kg/h)	416.64±209.9	309.56±163.6	**0.004** **
WHC (%)	56.7±16.6	57.2±18.8	**0.05***
Clay (%)	2.03±1.1	1.57±1.19	**0.00003*** **
Silt (%)	12.55±5.9	12.9±5.95	**0.007** **
Sand (%)	85.44±6.6	85.54±6.7	**0.01***

clusters with shorter branches indicating greater similarity between the joined forest types (less dissimilarity) while longer branches exhibit lower similarity. EHSTWH joined the entire cluster at the lowest similarity level (around 0.71) indicating its distinct soil variables compared to all the other forest types in the study area. The two forest types (RF and EHMCF) cluster together at the highest similarity value (above 0.95) reflecting their similar soil variables. In addition, NF is moderately similarly to RF and EHMCF but less than RF and EHMCF similar were to each other. AAPSEF is clustered together with RF, EHMCF and NF but had a lower degree of similarity between them.

However, in uninvaded sites (UN) of clustering displayed cophenetic correlation coefficient of 0.8393, which is little less than invaded sites (0.9753) from unweighted pair group method with arithmetic mean according to Bray-Curtis similarity indices. Correspondingly, EHSTWH remained the distinct forest types in uninvaded sites and branches off with low similarity index (0.75) from other forest types. In uninvaded sites, EHMCF and AAPSEF were relatively similar, clustering together at around 0.85 similarity index while NF and RF cluster ed together at a high similarity index level (0.90) indicating their share of very similar soil variables.



(A) Invaded site

(B) Uninvaded site

Figure 6.2: Cluster analysis among the invaded and uninvaded sites of Pakke Tiger Reserve

6.2.5 Correlation

Pearson's analysis of the soil variables indicated a significant correlation among some of the variables in invaded and uninvaded sites of the study area (Table 6.5 & 6.6). The soil pH was negatively correlated with OC, P, K and Fe (*r = -0.43, -0.37, -0.39* and *-0.51*), but positively correlated with clay (*r = 0.40*) in invaded site. In the uninvaded site, soil pH was negatively correlated with silt (*r = -0.42*) while sand (*r = 0.44*) was positively correlated. The correlation matrix showed the positive association between the invaded site (WHC; *r = 0.38*) and uninvaded site (OC, Zn and N; *r = 0.60, 0.52* and *0.54*) respectively. However, soil OC show both positive (WHC; *r = 0.45*) and negative (B*; r = -0.38*) correlation in invaded sites, while in uninvaded site, S (*r = -0.40*) showed negative correlation and K, Zn, B, Fe, Mn and N displayed positive correlation (*r = 0.50, 0.70, 0.52, 0.57, 0.43* and *0.79*) respectively. Besides, soil P have positive correlation with K, B and Fe (*r = 0.46, 0.48* and *0.42*) in invaded site only. The available K have both positive and negative correlation in invaded (B, Fe, silt and sand*; r = 0.44, 0.41, P<0.05; r =0.50* and *-0.48, P<0.001*) and uninvaded site (Mn, N and WHC; *r = 0.45, 0.40* and *-0.45*), respectively. In addition, available S have negative correlation with Zn, Fe and Cu (*r = -0.40, -0.43* and *-0.61*) in uninvaded sites only. The micronutrient Zn have shown positive correlation in both invaded (Fe and silt; *r = 0.45* and *0.37*) and uninvaded sites (B, Fe, Mn, Cu and N; *r = 0.37, 0.68, 0.57, 0.63* and *0.43*). Moreover, available B exhibit positive correlation with silt (*r= 0.38*) and negative correlation with sand (*r = -0.45*) in invaded site, while in uninvaded site, Mn, Cu and N have positive correlation (*r = 0.39, 0.39,* and *0.59*) only. Similarly, the micronutrient Mn have positive correlation with N in both invaded (*r= 0.53*) and uninvaded site (*r = 0.50*). However, Cu have negative correlation with N (*r = -0.39*) in invaded site, similarly clay also have negative correlation with sand (*r = -0.40*) in uninvaded site. Furthermore, the texture of silt had shown negative correlation with sand in invaded (*r = -0.83*) as well as uninvaded site (*r = -99*) respectively (Table 6.6 & 6.7).

Again, the Pearson's correlation between the invaded and uninvaded sites shows most of the variables were correlated with each other (Table 6.8). In between the invaded and uninvaded, the soil EC is strongly correlated with OC (*r =0.75*) but OC was found correlated with B (*r= 0.65*). Similarly, Zn also establish positive correlation with N (*r= 0.76*). Moreover, Fe was found positively significant with Mn (*r*= 0.85) and N (*r*= 0.81). Furthermore, S was found negatively correlated with Zn and B (*r= -0.65* and *-0.73*) while positively correlated with Cu (*r* =0.65). The micro nutrient, Cu was negatively related with silt (*r*= -0.71) and positively associate with sand (*r*= 0.67). Furthermore, the soil texture of clay was positively significant with silt (*r*= 0.69) and negatively with sand (*r*= -0.81) but silt was found highly negative correlate with sand (r= -0.98), respectively.

6.2.6 Principal Component Analysis

According to broken stick rule, principal components with eigenvalues greater than those predicted by the broken stick distribution were considered for the study. So, the principal components (PC1 and PC2) were retained for the study as both were significant for the present study. In addition, PC1 explained 95.88% of the total variance, which is unquestionably significant and PC2 account for 3.04% which provide much less influential information but potentially important for the study. Together, these two Principal components (PC1 and PC2) explained 98.92% of the variance in the data, making them highly representative of the dataset.

Table 6.6: Pearson's correlation matrix among the soil properties in invaded (IN) sites across the five forest types in the Pakke Tiger Reserve.
*EC-Electrical conductivity, OC- Organic carbon, P-Phosphorus, K-Potassium, S-Sulphur, Zn- Zinc, N-Nitrogen, B-Boron, Fe-Iron, Mn-Manganese, Cu-Copper, WHC-Water holding capacity; *Correlation is significant at p<0.05 (two- tailed).

	pH	EC	OC	P	K	S	Zn	B	Fe	Mn	Cu	N	WHC	Clay	Silt	Sand
pH	1															
EC	-0.03	1														
OC	-0.43*	0.20	1													
P	-0.37*	0.16	-0.04	1												
K	-0.39*	-0.21	-0.06	0.46*	1											
S	0.04	0.01	-0.18	-0.19	-0.12	1										
Zn	-0.27	-0.05	0.16	0.20	0.61	-0.08	1									
B	-0.11	-0.30	-0.38*	0.48*	0.44*	-0.22	0.26	1								
Fe	-0.51*	-0.19	0.16	0.42*	0.41*	-0.22	0.45*	0.01	1							
Mn	-0.02	0.36	-0.29	0.14	-0.11	0.27	-0.20	-0.09	0.07	1						
Cu	-0.07	0.29	0.20	0.20	-0.28	0.01	-0.35	-0.14	-0.23	-0.09	1					
N	-0.23	0.22	-0.07	0.24	0.34	0.20	0.32	-0.09	0.27	0.53*	-0.39*	1				
WHC	-0.36	0.38*	0.45*	0.00	0.09	-0.13	0.10	-0.25	-0.08	-0.05	0.28	0.23	1			
Clay	0.40*	-0.03	-0.26	0.19	0.07	0.13	-0.02	0.24	-0.23	0.06	-0.13	0.18	-0.23	1		
Silt	-0.27	-0.06	-0.22	0.03	0.50*	-0.06	0.37*	0.38*	0.04	-0.15	-0.33	0.02	0.16	-0.22	1	
Sand	0.19	-0.01	0.17	-0.07	-0.48*	0.11	-0.29	-0.45*	0.09	0.04	0.30	-0.05	-0.05	0.00	-0.83*	1

Table 6.7: Pearson's correlation matrix among the soil properties in uninvaded (UN) sites across the five forest types in the Pakke Tiger Reserve. *EC-Electrical conductivity, OC- Organic carbon, P-Phosphorus, K-Potassium, S-Sulphur, Zn- Zinc, N-Nitrogen, B-Boron, Fe-Iron, Mn-Manganese, Cu-Copper, WHC-Water holding capacity; *Correlation is significant at $p<0.05$ (two- tailed).

	pH	**EC**	**OC**	**P**	**K**	**S**	**Zn**	**B**	**Fe**	**Mn**	**Cu**	**N**	**WHC**	**Clay**	**Silt**	**Sand**
pH	1															
EC	0.21	1														
OC	0.24	0.60*	1													
P	0.18	0.07	0.25	1												
K	0.27	0.10	0.50*	0.05	1											
S	-0.04	-0.12	-0.40*	-0.19	-0.15	1										
Zn	-0.07	0.52*	0.70*	0.11	0.34	-0.40*	1									
B	0.09	0.19	0.52*	0.26	0.27	-0.32	0.37*	1								
Fe	0.09	0.19	0.57*	0.05	0.19	-0.43*	0.68*	0.20	1							
Mn	-0.23	0.19	0.43*	-0.31	0.45*	-0.11	0.57*	0.39*	0.41	1						
Cu	-0.09	-0.02	0.31	0.19	0.17	-0.61*	0.63*	0.39*	0.53	0.31	1					
N	0.04	0.54*	0.79*	-0.05	0.40*	-0.31	0.43*	0.59*	0.30	0.50*	0.03	1				
WHC	-0.25	0.22	0.02	0.02	-0.45*	0.22	-0.08	-0.07	-0.12	-0.04	-0.27	0.15	1			
Clay	-0.25	-0.01	-0.10	-0.34	0.18	0.06	0.12	0.01	0.05	0.33	0.17	-0.03	-0.23	1		
Silt	-0.42*	-0.03	-0.04	-0.19	0.10	-0.02	-0.08	-0.35	0.06	0.11	-0.13	0.05	0.05	0.27	1	
Sand	0.44*	0.03	0.05	0.18	-0.11	0.01	0.06	0.33	-0.07	-0.14	0.10	-0.03	-0.05	-0.40*	-0.99*	1

Table 6.8: Pearson's correlation matrix between invaded (IN) and uninvaded (UN) sites in the Pakke Tiger Reserve. *EC-Electrical conductivity, OC- Organic carbon, P-Phosphorus, K-Potassium, S-Sulphur, Zn- Zinc, N-Nitrogen, B-Boron, Fe-Iron, Mn-Manganese, Cu-Copper, WHC- Water holding capacity; *Correlation is significant at p<0.05 (two- tailed).

	pH	EC	OC	P	K	S	Zn	B	Fe	Mn	Cu	N	WHC	Clay	Silt	Sand
pH	1															
EC	0.00	1														
OC	0.25	0.75*	1													
P	-0.40	-0.17	-0.14	1												
K	0.08	0.00	-0.07	-0.15	1											
S	-0.19	-0.21	-0.31	0.01	0.16	1										
Zn	-0.12	0.15	0.34	0.35	0.37	-0.65*	1									
B	0.04	0.42	0.64*	-0.19	-0.41	-0.73*	0.42	1								
Fe	-0.24	0.16	0.19	0.10	0.44	0.18	0.34	-0.22	1							
Mn	-0.02	0.08	0.25	0.01	0.16	0.33	0.05	-0.17	0.85*	1						
Cu	0.24	0.07	0.21	-0.07	-0.10	0.65*	-0.54	-0.42	0.31	0.49	1					
N	-0.20	0.25	0.29	0.26	0.51	-0.22	0.76*	0.05	0.81*	0.57	-0.12	1				
WHC	-0.31	0.02	-0.46	-0.12	-0.07	0.39	-0.49	-0.35	0.07	0.21	-0.08	-0.14	1			
Clay	-0.47	-0.61	-0.60	0.36	0.49	0.12	0.37	-0.42	0.38	0.13	-0.35	0.37	0.10	1		
Silt	-0.29	-0.49	-0.56	0.55	0.17	-0.25	0.44	-0.14	-0.08	-0.18	-0.71*	0.24	0.20	0.69*	1	
Sand	0.35	0.54	0.60	-0.54	-0.26	0.18	-0.46	0.21	-0.03	0.12	0.67*	-0.29	-0.18	-0.81*	-0.98*	1

The PCA suggest that the forest types were clearly separated along the PC1 and PC2 axis, with INWH and UNNF clustering together, INAAP and UNMCF at the opposite ends and the other forest types falling in between. Moreover, INNF stand out dramatically due to exceptionally high concentration of P and N in invaded while the WHC and sand show variation that may subtly influence UNAAP and INWH. Most of the variables (EC, OC, pH, S, Fe, P, Mn, S, B, Cu, Zn, clay and silt) were similarly correlated and have comparable contribution to the principal components but N and K were distinctly away from the main cluster indicating strong and unique influence on the PC1 and PC2 respectively. The principal component (PC1) had positive loadings on N and sand but negative loadings on pH, EC, OC, B, Zn, Mn, Cu and clay while PC2 had positive loading on K only, but negative loadings on N, WHC and sand, respectively (Table 6.9).

6.3 Discussion

IAPs with the ability to alter the ecosystem traits, outcompete the native vegetation (Walker and Vitousek, 1991; Brown and Gurevitch, 2004.) leading to changes in the structure of community (García-Robledo and Murcia, 2005) and below ground soil pools and their processes by interfering with the quality of litter (Allison and Vitousek, 2004), mineralization and soil moisture (Blank and Young, 2002). The overall study showed that IAPs had a significant impact on the physicochemical properties of soil across the different forest types in Pakke Tiger Reserve. Similar findings have been reported form the previous study, which supports the present results on impact of IAPs on natural ecosystem properties (Ehrenfiled, 2003; Levine et al., 2003; Liao et al., 2008; Debnath et al., 2018; Ahmad et al., 2019).

6.3.1 Variation in the Soil Physical Properties

In the present study, the water holding capacity (WHC) have lower concentration in all four invaded sites except for EHMCF ($P>0.05$), as compared to uninvaded sites across the five forest types, clearly indicating that the *C. odorata* does not preferring high WHC in the soil properties. In contrast, clay and silt percentage in the texture class was higher in all invaded sites in comparison to uninvaded sites across the forest, except for the silt in EHMCF ($P>0.05$). However, texture class in sand percentage was lower in all invaded sites except for EHMCF ($P>0.05$) in the sampling sites. Moreover, the lower concentration of WHC in the invaded site (Naude, 2012; Tererai et al., 2015), may be of particular species-driven (Stanek et al., 2020) but decreased in water availability might be reverse once the IAPs removed from the sites (Eviner and Hawkes, 2012). However, these alterations of WHC in soil properties might remain for months to decades even after the active removal management in those particular areas (Van der Putten et al., 2007).

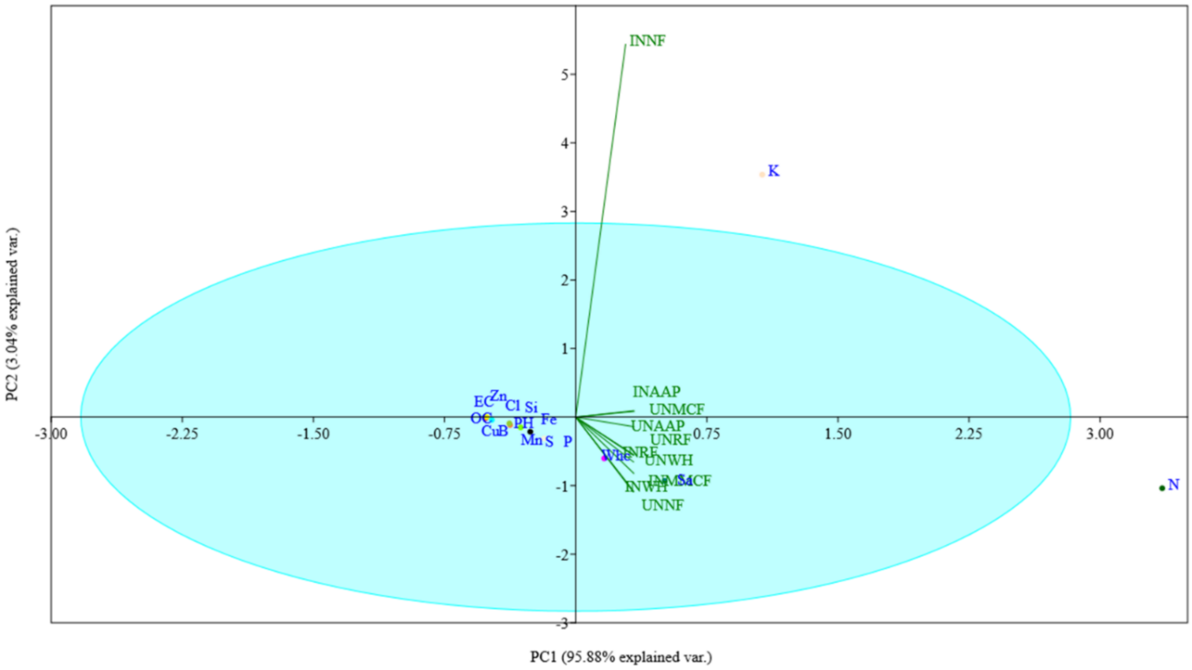


Figure 6.3: Principal Component Analysis (PCA) based on the physicochemical properties across the sampling sites of five forests in Pakke Tiger Reserve. * IN-Invasive, UN-Un-invasive; AAP- Assam Alluvial Plain Semi Evergreen Forest, MCF- East Himalayan Mixed Coniferous Forest, WH- the East Himalayan Subtropical Wet Hill, NF- Non-Forest and RF-Riverine Forest.

Soil particle fractions (clay, silt and sand) is an important factor which need to be understand to study the succession of invasion. The content of clay is known to influence the available nutrients in the soil properties (Tererai et al., 2015) because of the availability of higher nutrients content in the former soil, in which IAPs became dominant (Chacon et al., 2008). The outcome is corresponded with the previous finding had been reported by Hazlett et al. (2005) also in which all the invaded sites have high content of clay. Similarly, the study reported by Sharma and Gupta (1989) and Adella et al. (2020) have the same evidence of high silt in invaded sites.

Table 6.9. Physicochemical proportion of variation expressed in terms of vector loadings (varimax rotation) across the sampling sites.

	PC 1	PC 2	PC 3	PC 4	PC 5	PC 6	PC 7	PC 8	PC 9	PC 10
pH	-0.48[a]	-0.04	-0.43[a]	0.27	-0.32	0.30	0.22	0.27	-0.51[a]	2.15[a]
EC	-0.52[a]	0.00	-0.58[a]	0.25	-0.27	0.18	0.60[a]	0.74[a]	-0.38[a]	0.41[a]
OC	-0.52[a]	-0.01	-0.58[a]	0.29	-0.20	0.33[a]	0.30	0.84[a]	-1.20[a]	-1.62[a]
P	-0.18	-0.31	0.16	-3.55[a]	0.30	0.37[a]	-0.78[a]	0.66[a]	0.05	-0.05
K	1.07	3.54[a]	0.64[a]	-0.05	-0.05	0.03	-0.05	0.03	0.00	0.00
S	-0.26	-0.21	0.59[a]	0.11	3.31[a]	-0.45[a]	1.32[a]	-0.80[a]	-0.11	0.00
Zn	-0.52[a]	0.00	-0.59[a]	0.19	-0.37[a]	0.22	0.52[a]	0.29	-0.26	-0.43[a]
B	-0.51[a]	-0.01	-0.56[a]	0.26	-0.38[a]	0.22	0.55[a]	0.25	-0.57[a]	-1.53[a]
Fe	-0.31	-0.15	-0.12	0.62[a]	0.22	0.12	-0.84[a]	0.67[a]	3.10[a]	-0.05
Mn	-0.38[a]	-0.12	-0.27	0.95[a]	0.83[a]	-0.09	-3.22[a]	-0.59[a]	-1.02	-0.11
Cu	-0.52[a]	-0.01	-0.54[a]	0.30	-0.12	0.28	0.40	0.68[a]	-0.13	2.00[a]
N	3.35[a]	-1.04[a]	-1.30[a]	0.10	0.09	-0.04	0.12	-0.02	-0.04	0.01
WHC	0.16	-0.60[a]	1.66[a]	0.17	-0.80[a]	-3.05[a]	0.00	0.93[a]	-0.21	-0.02
Clay	-0.51[a]	0.00	-0.57[a]	0.13	-0.38[a]	-0.01	0.50[a]	-0.57[a]	1.41[a]	-0.98[a]
Silt	-0.38[a]	-0.09	-0.24	-0.57[a]	-1.24[a]	-0.58[a]	0.25	-3.15[a]	0.03	0.30
Sand	0.51[a]	-0.93[a]	2.74[a]	0.55[a]	-0.62[a]	2.15[a]	0.12	-0.25	-0.15	-0.09

[a] Soil property related to the principal components (PCs)

However, the content of sand in invaded sites were low as compared to uninvaded sites in the present study which is alternative to some of the findings in which IAPs have high sand content (Debnath et al., 2018; Comole et al., 2021). But Gebrekiros and Tessema (2018) revelation of higher sand percentage in soil texture concur with the present result. Also, Stefanowicz et al. (2017) reported that IAPs also grow in low sand content, even under harsh condition in high sand content (Widmann et al., 1990; Hannan-Jones and Playford, 2002). Furthermore, the variation of soil texture (clay, silt and sand) in the study area might be due to the increasing biological activities which may have enhanced the weathering process and provide optimal moisture underneath leading to alteration in physicochemical properties of soil (Roba et al., 2017). The overall findings of physical soil properties of the present study were in consistent with the outcomes of prior research investigations (Belnap and Phillips, 2001; Sharma and Raghubanshi, 2009; Zhang et al., 2009; Mandal and Joshi, 2014b; Debnath et al. 2018; Gebrekiros and Tessema, 2018; Chacon et al., 2020; Stanek et al., 2020).

6.3.2 Variation in Soil Chemical Properties

The soil pH in the present study revealed that the soils were characterized by the acidic nature of pH with an average of 4, due to excessive rainfall and leaching of bases. Moreover, pH was non-significant in the invaded sites ($P>0.05$) while in natural forest areas, it was found to significant ($P<0.05$). These might be due to relatively low decomposition rates of low fungal and bacterial activity in the soil (Tererai et al., 2015) and *C. odorata* favor more acidic because of acidic extrudes generated by this species (Ikhajiagbe, 2016). Similar results in accordance to the present study were also presented by Dobrylovska (2001), Gentili et al. (2019) because IAPs tend to acidify the soil (Agusto et al., 2002; Guckland et al., 2009). However, increase and decrease of soil pH have been reported in the plant invasion due to IAPs ability to uptake nitrate ions (Ehrenfeld et al., 2001; Ehrenfeld, 2003). But, impact on pH by IAPs may depend on soil types and physiology of the particular species (Stanek et al., 2020). In addition, Diekmann et al. (2016) also reported that IAPs have affected pH in riparian habitats which is in accordance with the result of the present study (NF; $P<0.05$). The soil electrical conductivity (EC) had significant difference between invaded and uninvaded in all sampling sites except for NF did not show significant. In addition, EC was lower in the invaded sites of AAPSEF and EHSTWH which is in concordance with the report of Osunkoya and Perrett (2011). In contrast, it also shows higher in EHMCF and RF in the invaded sites. Similar results of these higher EC were also reported by Shiferaw et al. (2021) and Akshayakumari et al. (2024).

In *C. odorata* invaded area, soil organic carbon (OC) provides a measurement of soil organic matter (Manjaiah et al., 2000), which is an important indicator of soil quality (Kibblewhite et al., 2008). In the present study, OC have high OC in four invaded sites except for EHSTWH which show non-significant. This revelation of higher values of OC in invaded sites were in agreements with the observation made by Koutika et al. (2005), Tondoh et al. (2013) and Kone et al. (2021) in the *C. odorata* dominated area. The chemical properties in soil were primarily influenced by OC combined with high level of magnesium (Tondoh et al., 2013). Moreover, *C. odorata* invaded area had high OC content, meaning the severity of invasion is significantly high in the surface as well as subsurface soil layers (Wei et al., 2017) and preclude native plant.

The amounts of nutrients in the soil can change due to the presence of IAPs (Dassonville et al., 2008). It is also evident that nutrient enrichment encourages plant invasion (Lake and Leishman, 2004) and values of the soil nutrients in IAPs invaded sites were found to high as compared to natural sites (Oludare and Muoghalu, 2014). Both macro and micro nutrients were critical parts of soil components which give the quality of soil indicators. Commeasuring, soil macronutrients (N, P, K and S) is crucial for preserving biodiversity since they make up a significant amount of soil quality indicators (Li et al., 2013). In the present study, soil macronutrients i.e., N and K was significantly difference in all invaded and uninvaded sites and the concentration was high in all four invaded forest sites except for EHSTWH. Moreover, N and K presence can have impact on the growth of IAPs (Nozzolillo, 1970) and the results of high contents of both macronutrients beneath the *C. odorata* infested soil were supported by earlier findings (Tondoh et al., 2013; Wei et al., 2017; Kone et al., 2021; Lini et al., 2022). The increase of nutrients in the *C. odorata* invaded sites might be due to accumulation of litter fall (Ehrenfeld, 2010; Lini et al., 2022). The greater resource disposal increases the vulnerability of more invasions in the natural forest (Burke and Grim, 1996; Wei et al., 2017). However, P and S also showed significant difference in four invaded sites except for EHMCF in P and NF in S. Additionally, P and S have high and low contents in invaded sites. Similar studies of *C. odorata* invaded sites containing high P content have been reported (Kone et al., 2021). High concentration of P in *C. odorata* invaded area is because of faster decomposition of leaf litter (Kone et al., 2021) and the input of plant residues is substantial and significant portion of soluble P comes from plant residues (Ha et al., 2007). Moreover, the content of S was found high because of decrease bacterial movement and accessibility of free nutrients for plant root uptake (Liptzin and Silver, 2009). The present results were in line with those of Osunkoya and Perret (2011) who reported high concentration of Sulphur in *Lantana* invaded site in Australia.

Furthermore, the soil micro nutrients, viz., Fe (Iron), Cu (Copper), Mn (Manganese), B (Boron) and Zn (Zinc) also play small part, though not major but enough to understand the chemical properties of soil dynamics. In the present study, the first four micronutrients i.e., Fe, Cu, and Mn have shown significant difference in all invaded and uninvaded sites in all sampling plots and found considerably high in most of the invaded site as compare to uninvaded sites. However, Zn was not significant in all invaded and uninvaded site of the study but found high in almost all invaded sites also. The result derived from this study is consistent with the result reported by Cuda et al. (2017) following *Impatiens glandulifera* invasion in Europe. However, the result contradicts with the findings of Osunkoya and Perrett (2011), Debnath et al. (2018) and Ahmad et al. (2019) who reported decrease of micronutrients in the soil following *Lantana camara* invasion in Australia, *Chromolaena odorata* and *Leucanthemum vulgare* invasion in India.

In contrast, the remaining micronutrient B had lower content in almost all invaded site and found significant in between invaded and uninvaded sites. This result contradicts the report of Tekiela and Barney (2015) and Cuda et al. (2017) who reported increase of B in the invaded site. However, the present study is in line with the report of Tererai et al. (2015) which show that B content were high in uninvaded site following the seasonal invasion study of *Eucalyptus camaldulensis* in South Africa. The variation of increase and decrease of soil micronutrients in the invaded soil properties in the present study might be an eye-opener for future researches. However, there had been limited study on soil micronutrients regarding plant invasion and the possible reason behind the increase could be the continuous supply of cation concentration in the soil (Nikolic and Pavlovic, 2018) by accumulation of litter quality and interaction of soil microbes (Castro-Diez et al., 2012). Decrease concentration in soil properties might be of constant rapid uptake of nutrients, leading to their depletion in invaded areas (Osunkoya and Perrett, 2011). Moreover, studies were more confined to nutrients (N, P, K, Mg, Ca and S) in plant invasion but limited target in micro nutrients. So, in order to fully understand the mechanism of IAPs beneath the soil, more broad study of micronutrients is also necessary. Therefore, the present study was done to understand the dynamics of soil chemical properties including both macro and micronutrients for the first time from eastern Himalayan region.

6.3.3 Analysis of Variance Among and Between the Sites

The analysis of variance across all the sampling sites shows that eight of the soil parameters exhibit significant difference between the invaded and uninvaded (Table 6.5). Across all forest sites, boron, water holding capacity, silt and sand were found high in uninvaded sites in comparison to invaded sites and significantly different. The results of the present study concurred with the findings of Fink and Wilson (2011), Perret et al. (2012), Cuda et al. (2017), Osunkoya et al. (2017), Stefanowicz et al. (2018) and Stanek et al. (2020). The soil texture i.e., silt and sand having high content in uninvaded sites might be attributed by increasing biological activity resulting weathering process and provision of moisture

(Abdella et al., 2020), under forest canopy. The level of boron concentration found in the study is considered safe as it does not cross the toxicity level (Nable et al., 1997). Moreover, less boron in invaded sites lead to increase biomass production, good production of seed and healthy vigor of plant (Gupta et al., 2014; Mapaura et al., 2024). However, lower water holding capacity in invaded sites may be driven by particular species (Stanek et al., 2020).

In addition, iron, nitrogen, copper and clay in all forest sites were higher in invaded sites which were lines with the study of Fink and Wilson (2011), Osunkoya and Perret (2011), Perret et al. (2012), Cuda et al. (2017), Ahmad et al. (2019) and Mapaura et al. (2024). Moreover, high level of Fe and Cu in plant causes oxidative stress and halted microbial function, thereby reducing the growth rate (Liptzin and Silver, 2009) though invasive species with its capability, utilized in plant photosynthetic capacity through its interaction with carbohydrates and proteins (Perret et al., 2012). In addition, they uptake the nutrients readily than the native species making them more proliferation in invaded region by cycling these elements in useful way (Penuelas et al., 2010). Additionally, mean concentration of copper was lower than those known to be toxic in the present study (Marschner, 2011). Similarly, nitrogen is high in the invaded sites due to accumulation of litter decomposition of above ground biomass in comparison to co-occurring plants (Ruwanza and Shackleton, 2016). Furthermore, *C. odorata* is considered dangerous to livestock and not recommended for feeding (Shackleton et al., 2017; Aigbedion-Atalor et al., 2019) leading to litter accumulation which later accelerate invasion in the area. The physical properties of clay were also high in invaded sites which might be due to presence of more cation that binds soil water than silt and sand resulting in rich nutrients (Simba et al., 2013). The reason behind the variation in soil texture between invaded and uninvaded sites might be attributed by increasing biological activity resulting weathering process and provision of moisture (Abdella et al., 2020). The variation observed in the present study suggests that invasive alien plant species play a role in alteration of physicochemical properties of soil (Comole et al., 2021).

6.3.4 Correlation of Soil Parameters

In the present study, correlation analysis between the invaded and uninvaded sites across the forest types had shown significant correlation with each other in some of the soil variables. The findings presented in this present study is unique as compared to previous studies in which IAPs strongly correlating with these soil properties like pH, K, P and N between the invaded and uninvaded sites (Simba et al, 2013; Osunkoya and Perrett, 2011; Soti and Jayachandran, 2016; Ahmad et al., 2019). More specifically, the study reveal that the soil EC was strongly positive correlated with OC, which is contradictory to some of the studies who reported negative correlation with OC (Terra et al., 2004; Werban et al., 2009; Ahmad et al., 2019; Iranmanesh and Sadeghi, 2019). However, the results were in line with the findings of Behera and Shukla (2015), Johnson and Smith (2017) and Gudla et al. (2023) and possible reason might be increasing uptake of minerals and higher biomass production, lower soil organic carbon mineralization and higher sediment deposition (Terra et al., 2004). Moreover, increase or decrease of nutrient had comparable or distinct effect on other. In addition, Terra et al. (2004) also reported that EC and OC have both negative and positive correlation with each other depending on the site. Similarly, soil organic carbon (OC) showed strong positive correlation with boron and is concordance with the results of numerous other studies (Dey et al., 2014; Kumari et al., 2017; Lamare et al., 2023). The reason behind the close relation between OC and B can be attributed to the fact that OC acts as a source of B limiting the transition of B from residual pool to other pools, leading to higher concentration of residual B in soils with higher organic matter content (Lamare et al., 2023). The strong association between these components and OC indicate that soil organic matter plays a crucial role in determines the availability of B in soils (Mathur and Sudan, 2011).

Also, the present study discloses the significantly positive correlation of micronutrient Zn with macronutrients N. This finding is consistent with numerous other studies (Wijebandara et al., 2011; Zhang et al., 2014; Baghel and Bachkaiya, 2023). The tenable evidence for this may be OC instantaneously engaged in the accessibility of almost all components of Zn in soil system which immediately raised the quantity of Zn (Zhang et al., 2014) and interfere with the transition into a nonavailable form there by raising the supply of Zn to an optimal level (Mandal and Mandal, 1986; Verma et al., 2012). The study also showed the positive correlation of Fe with Mn and N which is in agreement with the finding of Kader et al. (2013) and Hamer (2020). The possible reason might be the microbial activities toward iron regarding nitrogen fixation and nutrient uptake involving the synergistic response to balanced nutrition (Kerkeb and Connolly, 2006; Sheoran et al., 2018). In addition, Cu was negatively correlated with silt but sand was found positively with Cu and the result is concurring with the report of Wei and Jing (1990) and Karami et al. (2009). The plausible explanation for this might be the responsibility of adsorption by the copper with regards to clay, pH and organic matter (Al-Qunaibit et al., 2004). Furthermore, the available Sulphur was found correlated negatively with zinc and boron while positively with Cu. However, the present result contradicts the finding of Shukla et al. (2021) which show positive correlation of Zn, B and Cu. This unique finding may be due to the complex interactions involving nutrient availability and microbial activity leading more uptakes of essential nutrients in the invaded areas. In the context of soil texture, the present study revealed that clay was negatively strong correlated with sand but positively associated with silt. Moreover, silt was significantly negatively correlated with sand. These results concur with the finding reported by Banin and Amiel (1970), Mousavifard et al. (2013), Zhao et al. (2015) and Raheem and Omar (2021). The probable reason for this could be the silt content which does not change widely and if discovered connected with clay content, it will inversely correlate with sand content and vice versa. Overall, increasing clay content results in increased surface area and consequently, the cation exchange capacity and moisture retention at various phade increases concurrently (Banin and Amiel, 1970). The negative correlation is due to the compositional nature of soil texture between clay and sand while the positive correlation often occurs because silt and clay were fine soil particles to accumulate together in depositional environment (Brady, 1984; Hillel, 2003). Moreover, the degree of the correlation is related to the difference in the underlying soil properties between the invaded and uninvaded.

Principal component analysis (PCA) is a multivariate statistical method that turns a collection of correlated variables into a set of aligned, uncorrelated axes known as principal components (PCs) via orthogonal transformation (James and McCulloch, 1990; Robertson et a., 2001; Gotelli and Ellison, 2004) and deals with multivariate datasets (Janzekovic and Novak, 2012). However, retaining the important PCs is a confusing task (Peres-Neto et al., 2005) but PCs in the present study were selected following through broken stick rule method (MacArthur, 1957). Moreover, Harper (1999) state that the PCA identifies hypothetical variables that explains for as much of the difference in a multivariate data as feasible. PCA result of the present study showed that INNF physicochemical properties of the soil were different from the rest of soil sample sites as denoted by PC1. This might be due to INNF being non forest areas with grass and shrub domination in the low elevation with high anthropogenic activities in the areas. The soil properties that governed the differentiation were N, pH, EC, OC, B, Zn, Mn, Cu and clay. Moreover, K, N, WHC and sand were the main soil variables influencer that defined the separation of invaded and uninvaded sites. The variables that were influential to component scores and loadings i.e., PC1 and PC2 were often used to describe the finding of PCA according to Feoli and Orloci, (1992). Furthermore, the present finding of the continuity of variations in the general features of the soil at each site supported that the invasion of *C. odorata* changes in the soil, which led to the division of invaded and uninvaded sites.

6.3.5 Implications of Controlling Invaders

An additional managerial restriction on IAPs is that *C. odorata* proliferate very fast with wind dispersal seeds (Kone et al., 2021), inhibiting co-occurring native by secreting allelochemical compound (Mandal and Joshi, 2014a), increasing microbial activities and nutrient availability (Liao et al., 2008; Wei et al., 2017), improving soil fertility and nutrients conditions (Koutika et al., 2004; Aigbedion-Atalor, 2020) and ability to thrive in heavy contaminated sites (Ayesa et al., 2018). With these said ability, *C. odorata* can easily changes the dynamics of soil properties in any invaded sites (Xu et al., 2020). Moreover, this species is considered plague around the world (Kone et al., 2021) because of its ability to damaged biodiversity (Maroun, 2017) and giving serious impact on socio-economic sector (Cuthberth et al., 2021). While there have been several attempts to combat and restored the invaded areas either by physically, chemical and biologically (Muniappan et al., 2005; Mugwedi, 2020; Aigbedion-Atalor, 2020) and efforts to address this issue rarely achieved success though the consequences of IAPs usually remain in the spoil even after their removal (Aigbedion-Atalor et al., 2019; Rai and Singh, 2024). Furthermore, Corbin and D'Antonio (2004) state that long lasting effects of IAPs on soil processes and their influences on restoration alternatives were yet unknown. The nutrient dynamics of post clearing may hinder the establishment of native vegetation also (Lindsay and French, 2005) and need to be monitored closely. Removal of *C. odorata* from the previous invaded areas does not imply avoidance of penetration because of seed bank presence (Shen et al., 2006) and inversional meltdown phenomenon (Simberloff and Von Holle, 1996; Chen et al., 2022). To achieve complete eradication of *C. odorata* from the invaded areas, plant- soil feedbacks are the utmost importance in controlling and management of IAPs invasion (Sun et al., 2017; Raheem et al., 2024). Consequently, in order to properly managed and re-establish invaded regions, simply elimination of IAPs from the ground surface is not adequate. To decrease the influence of invaders on essential soil qualities and dynamics, it is vital to obtain knowledge into the feedback loops that exist between the IAPs and soil below ground.

6.4 Conclusion

For the first time the present study had revealed a hidden gem from the eastern Himalayan region related to IAPs effects on physicochemical properties of soil. The study had provided enough evidences of *C. odorata* impact on the physicochemical properties of soil between the invaded and uninvaded sites across the different forest types in Pakke Tiger Reserve, Arunachal Pradesh. The soil underneath *C. odorata* had high concentration of soil organic carbon and available nitrogen defines the increasing invasion in the region. The results clearly show that the invasion of *C. odorata* causes alteration of soil nutrients concentration by utilizing its maximum uptake. Moreover, increase in some of the soil parameters and significant differences were observed from the study. This might have paved the way of successful invasion of *C. odorata* across the different forest types and inhibit the native population. It also shed light on the decrease of water holding capacity and available boron in *C. odorata* invaded areas. The present study had made a noteworthy sustenance in understanding the impact of *C odorata* on the underground soil system dynamics from the eastern Himalayan state of Arunachal Pradesh. However, these findings cannot be deemed flawless without further comprehensive experimentation on physicochemical properties of soil. Therefore, more future researches regarding IAPs involvement in physicochemical properties of soil and plant diversity relationship are to be encouraged in coming years. So, that the invasion ecology i.e., below ground as well as above ground should be thoroughly understood in order to create more scientific approaches to control the IAPs in these fragile ecosystems of eastern Himalaya.

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Chapter 7

7.0 Conclusion

The present study was conducted to determine the distribution and invasion pattern, climatic influence of phenology and ecological impact of dominant Invasive Alien Plant species (IAPs) across different forest types of Pakke Tiger Reserve. The study revealed a total of diversity 151 species in which IAPs was 28% in the reserve. The species richness of IAPs displays an increasing trend with increasing elevation but decreasing trend with the level of invasion. *Chromolaena odorata* was the most dominant IAPs which invades all the forest types in the study area followed by *Ageratum conyzoides, Mikania micrantha* and *U. lobata.* The entire selected invasive species exhibited contagious in all forest, except for *U. lobata* displayed random distribution in non-forest. AAPSEF, NF and RF were more invaded in comparison to EHMCF and EHSTWH. *Bidens Pilosa* had the longest flowering phenophases in EHSTWH. Similarly, *C. odorata* had the maximum duration in EHSTWH. For leaf initiation and fall phenophases, *Urena lobata* displayed the longest duration in RF. Coefficient of variation showed that most of species display asynchronous phenophases across the forest types. Precipitation, minimum and maximum temperature directly influenced the phenophases. Relative humidity was influencing the phenophases when in cohesion with other climatic parameters. The soil pH was found acidic in all forest types but had a role in invasion process. Clay and copper had role in invasion as their contents were high in all invaded sites of the study area. EHSTWH had the fluctuating soil parameters in comparison to other forest types in Pakke Tiger Reserve. The content of uninvaded sites in EHSTWH were almost higher than invaded sites. Soil physical property i.e., water holding capacity, clay, silt and sand were impacted by the *Chromolaena odorata* invasion. Micronutrients in the invaded sites were also altered by the *Chromolaena odorata* invasion. Considering their increasing proliferation in the eastern Himalayan region, following recommendations and future scope need to be taken into consideration for managing the invaded region. Suggestive management strategies to control their proliferation of the present study sites are given in Figure 7.1.

7.1 Recommendations

- Local and staff awareness programmed of IAPs and their effects on surrounding environment and inside the Pakke Tiger Reserve.
- During the visit of tourists and students, certain IAPs awareness and its harmful effects should be given to them.
- Restriction of personnel automobiles or proper cleansing of automobiles if used. As they were used for transportation, road clearance and hazard removal which might serve as carrier of IAPs to random places inside the reserve.
- Identification of heavy disturbance sites/habitats of IAPs and their removals before flowering and fruiting to restrict dispersal of seeds.
- Identification of native species co-occurring with IAPs and restoring the sites with native flora after eradication or removal.

7.2 Future scope of the study

- For chapter 4: Future research must establish Long-Term Ecological Research (LTER) plots to monito IAPs population dynamics and spread rates over a multi-year timeframe (5-10 years). Controlled experiments are required to quantify the specific allelopathic effects of dominant IAPs on native flora germination and test impact on ecosystem functions (e.g., nutrient cycling). Integrating data with species distribution modelling (SDM) is essential to predict habitat suitability and colonization corridors under projected climate change and land use scenarios. Applied ecology efforts must pivot to rigorously testing the efficacy, cost effectiveness and non-target impacts of various control methodologies (manual, chemical). Ultimately, research should develop and validate native species restoration protocols specifically for areas post-IAPs removal to ensure long term ecosystem stability.
- For chapter 5: Future research should focus on delving deeper into mechanistic drivers of the observed phenological variation by moving beyond correlation with micro-climate and experimentally investigating plot-level factors such as soil properties and light intensity. given the limitation of a one-year study, a crucial next step is to initiate long-term, inter-annual monitoring (at least 3-5 years) to capture natural climatic variability and validate predictive models of invasive spread. The analysis should also be refined by treating elevation as a continuous variable to model the precise rate of

phenological shift per unit of altitude, which is highly relevant for climate change projections. Finally, research should quantify the ecological impact of these phenological shifts, specifically determining the competitive advantage they confer upon invasive species over native flora.

- For chapter 6: Future studies should prioritize understanding the long-term ecological legacy of *C. odorata* by monitoring soil recovery post eradication to inform effective restoration strategies. A deeper investigation into the mechanisms of nutrient cycling, such as decomposition rates, is required to explain the observed buildup of organic matter and nutrients. Research should also explore the interplay between the invasive plant and soil microbial communities to uncover specific soil-plant feedbacks. Finally, extending this comparative work to other prominent invasive alien plant species (IAPS) in the eastern Himalayas could reveal if these alterations are unique to *C. odorata* or a general invasive syndrome.

7.3 Management Programs of Invasive Alien Plant Species in India

Although some of the Indian states i.e., Himachal Pradesh, Jammu & Kashmir, Tamil Nadu and Uttarakhand have priorities on the management of invasive alien plant species. However remaining states of India don’t have any proper management strategies to counter the imbalance causes by these species in their surrounding areas or forest region. India being parts of The Aichi Biodiversity Targets at the 10th meeting of the conference of the parties (COP 10) held in Nagoya, Japan in October, 2010 clearly state to reduce the burden on biodiversity in five strategic goals, which IAPs comes under Strategic Goal B (Aichi Target 9). The goal was to reduce the direct pressures on biodiversity and improves sustainable uses among the participating countries by 2020. This particular target was renamed as National Biodiversity Target 4 and adopted to counter the proliferation of invasive alien species in India. However, lack of inventories and information on IAPs was the main reason of failure for the specified target. Therefore, the Ministry of Environment, Forest and Climate Changes (MoEFCC) launched the National Action Plan on Invasive Alien Species (NAPINVAS) in 2019 with a plan to focus on the prevention, early detection, control and managements of IAPs. As many countries failed to achieved the targeted goals including India, a new framework i.e., Kunming-Montreal Global Biodiversity Framework was again adopted in December, 2022 under the international agreement with UN-CBD required the member states to reduce the impact causes by IAPs on biodiversity and ecosystem services by 2030.

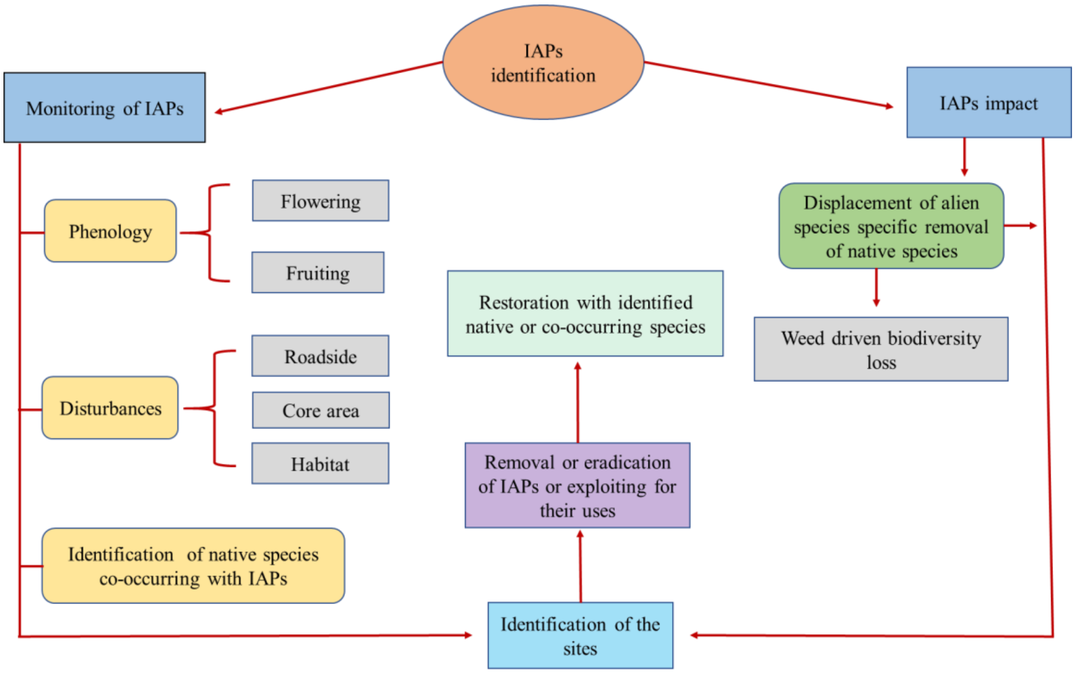

Figure 7.1: Schematic management strategies of Invasive Alien Plant species in Pakke Tiger Reserve

With only five years remaining on the targeted year, still there were limited studies and management strategies on IAPs in India. Moreover, the entire northeast region of India presents a more alarming situation as states clearly lacking information and management protocols including the study region. Northeast India, being a biodiversity hotspot characterized from tropical rainforest to alpine scrubs, with different kinds of flora and fauna supporting 50% of the country flora especially the endemic plant species also. IAPs having the capabilities of altering biodiversity, if left unchecked in these regions will lead to dangerous outcomes in future. Therefore, urgent needs for management's strategies are required within the country to counter these invisible yet destructive species.

References

Aalders L.E., Hall I.V. and Forsyth F.R. 1969. Effects of partial defoliation and light intensity on fruit-set and berry development in the lowbush blueberry. *Horticultural Research,* 9: 124–129.

Abbad, A. and Benchaabane, A. 2004. The phenological study of Atriplex halimus L. *African Journal of Ecology,* 42: 69–73.

Abdella, M., Nigatu, L. and Akuma, A. 2020. Impact of parkland trees (Faidherbia albida Delile and Cordia africana Lam) on selected soil properties and sorghum yield in Eastern Oromia, Ethiopia. *Agriculture, Forestry and Fisheries*, 9(3): 54.

Acevedo-Rodríguez, P. and Strong, M.T. 2012. Catalogue of seed plants of the West Indies. Washington, DC USA: Smithsonian institution, +1192p.

Adhikari, B.S. and Babu, M.M. 2008. Floral diversity of Baanganga Wetland, Uttarakhand, India. *Check list*, 4(3): 279-290.

Adjibode, A.G., Tougan, U.P., Youssao, A.K.I., Mensah, G.A., Hanzen, C. and Koutinhouin, G.B. 2015. *Synedrella nodiflora* (L.) Gaertn: a review on its phytochemical screening and uses in animal husbandry and medicine. *International Journal of Advanced Scientific and Technical Research,* 3.

Afreen, T., Srivastava, P., Singh, H. and Singh, J.S. 2018. Effect of invasion by *Hyptis suaveolens* on plant diversity and selected soil properties of a constructed tropical grassland. *Journal of Plant Ecology,* 11(5): 751-760.

Agrawal, A.A., Kotanen, P.M., Mitchell, C.E., Power, A.G., Godsoe, W. and Klironomos, J. 2005. Enemy release? An experiment with congeneric plant pairs and diverse above- and belowground enemies. *Ecology,* 86(11): 2979-2989.

Ahmad, M., Uniyal, S.K., Batish, D.R., Singh, H.P., Jaryan, V., Rathee, S., Sharma, P. and Kohli, R.K. 2020. Patterns of plant communities along vertical gradient in Dhauladhar Mountains in Lesser Himalayas in North-Western India. *Science of The Total Environment*, 716: 136919.

Ahmad, R., Khuroo, A.A., Hamid, M. and Rashid, I. 2019. Plant invasion alters the physico-chemical dynamics of soil system: insights from invasive *Leucanthemum vulgare* in the Indian Himalaya. *Environmental monitoring and assessment,* 191(Suppl 3): 792.

Aigbedion-Atalor, P.O. 2020. Weed or not a weed? Density, perceptions and management of *Chromolaena odorata* (Asteraceae) in West Africa: voices from Ghana. *Weed Research*, 60(6): 406-414.

Aigbedion-Atalor, P.O., Adom, M., Day, M.D., Uyi, O., Egbon, I.N., Idemudia, I., Igbinosa, I.B., Paterson, I.D., Braimah, H., Wilson, D.D. and Zachariades, C. 2019. Eight decades of invasion by *Chromolaena odorata* (Asteraceae) and its biological control in West Africa: the story so far. *Biocontrol Science and Technology*, 29(12): 1215-1233.

Akomolafe, G.F. and Rosazlina, R. 2023. Evidence of invasion: land use influences the invasion of *Urena lobata* in the Guinea Savanna ecosystems of Nigeria. *Biological Invasions,* 25(7): 2175-2184.

Akshayakumari, A., Hareesh, T.S., Sathish, B.N., Nayak, B.G., Devagiri, G.M., Rohan, P.L. and Rashmitha, H.R. 2024. Impact of Alien Invasive Species (*Cassia spectabilis* DC.) on Soil Properties in Nagarahole Tiger Reserve. *Indian Journal of Ecology*, 51(1): 77-83.

Aldrich, P.R. and Hamrick, J.L. 1998. Reproductive dominance of pasture trees in a fragmented tropical forest mosaic. *Science*, 281(5373): 103-105.

Alexander, J.M., Naylor, B., Poll, M., Edwards, P.J. and Dietz, H. 2009. Plant invasions along mountain roads: the altitudinal amplitude of alien Asteraceae forbs in their native and introduced ranges. *Ecography*, 32(2): 334-344.

Allen, E.B., Steers, R.J. and Dickens, S.J. 2011. Impacts of fire and invasive species on desert soil ecology. *Rangeland Ecology & Management*, 64(5): 450-462.

Allison, S.D. and Vitousek, P.M. 2004. Rapid nutrient cycling in leaf litter from invasive plants in Hawai'i. *Oecologia*, 141: 612-619.

Alpert, P., Bone, E. and Holzapfel, C. 2000. Invasiveness, invasibility and the role of environmental stress in the spread of non-native plants. *Perspectives in plant ecology, evolution and systematics*, 3(1): 52-66.

Al-Qunaibit, M.H., Mekhemer, W.K. and Zaghloul, A.A. 2005. The adsorption of Cu (II) ions on bentonite - a kinetic study. *Journal of Colloid and Interface Science,* 283(2): 316-321.

Alvarez, M.E. and Cushman, J.H. 2002. Community-level consequences of a plant invasion: effects on three habitats in coastal California. *Ecological Applications*, 12(5): 1434-1444.

Alyokhin, A. and Sewell, G. 2004. Changes in a lady beetle community following the establishment of three alien species. *Biological Invasions*, 6: 463-471.

An, R., Chen, W., Hu, H., and Li, P. 2020. Effects of storage temperature on senescence and active ingredients of postharvest shanghai green leaves. *Packaging Eng*.,41(9), pp. 7–16.

Andreu, J. and Vilà, M. 2010. Risk analysis of potential invasive plants in Spain. *Journal for Nature Conservation*, 18(1): 34-44.

An-Ming, L. and D'Arcy, W.G. 1986. Solanaceae in China, 79-85. In Solanaceae: biology and systematics, An-Ming, L. and D'Arcy, W.G. (eds.), Columbia University Press, New York.

Anonymous. 1883-1885 and 1886-1970. Index Kewensis Plantarum Phanerogamarum. Vol. I-II and 15 suppl. Oxford: Clarendon Press.

Antonelli, A., Kissling, W.D., Flantua, S.G., Bermúdez, M.A., Mulch, A., Muellner-Riehl, A.N., Kreft, H., Linder, H.P., Badgley, C., Fjeldså, J. and Fritz, S.A. 2018. Geological and climatic influences on mountain biodiversity. *Nature Geoscience*, 11(10): 718-725.

Areces-Berazain, F. 2016. *Solanum torvum* (turkey berry). Forest. Downloaded from https://cabidigitallibrary.org by 2409:408b:1e:1a77:6d9b:c273:f4a1:f210, (Accessed on 03.09.2024).

Arévalo, J.R., Delgado, J.D., Otto, R., Naranjo, A., Salas, M. and Fernández-Palacios, J.M. 2005. Distribution of alien vs. native plant species in roadside communities along an altitudinal gradient in Tenerife and Gran Canaria (Canary Islands). *Perspectives in Plant Ecology, Evolution and Systematics*, 7(3): 185-202.

Arroyo, M.T.K., Armesto, J.J. and Villagran, C. 1981. Plant phenological patterns in the high Andean Cordillera of central Chile. *The Journal of Ecology*, 69: 205-223.

Arthur, G.D., Naidoo, K.K. and Coopoosamy, R.M. 2012. *Bidens pilosa* L.: Agricultural and pharmaceutical importance. *Journal of Medicinal Plants Research*, 6(17): 3282-3287.

Arunachalam, A. and Arunachalam, K. 2002. Evaluation of bamboos in eco-restoration of ‘jhum’fallows in Arunachal Pradesh: ground vegetation, soil and microbial biomass. *Forest Ecology and Management*, 159(3): 231-239.

Ashutosh, S. and Roy, P.S. 2021. Three decades of nationwide forest cover mapping using Indian remote sensing satellite data: a success story of monitoring forests for conservation in India. *Journal of the Indian Society of Remote Sensing*, 49(1): 61-70.

Aterrado, E.D. and Bachiller, N.S.J. 2002. Biological control of *Chromolaena odorata*: pre liminary studies on the use of the gallforming fly *Cecidochares connexa* in the Philippines. In Proceedings of the fifth international workshop on biological control and management of *Chromolaena odorata*, Durban, South Africa, Zachariades, C., Muniappan, R., Strathie, L.W, (eds.). Agricultural Research Council, Plant Protection Research Institute. pp. 137-139.

Augspurger, C.K. 1980. Mass-flowering of a tropical shrub (*Hybanthus prunifolius*): influence on pollinator attraction and movement. Evolution, 475-488.

Augspurger, C.K. 1982. A cue for synchronous flowering. The ecology of a tropical forest-Seasonal rhythms and long-term changes, 87: 133-150.

Augusto, L., Ranger, J., Binkley, D. and Rothe, A. 2002. Impact of several common tree species of European temperate forests on soil fertility. *Annals of Forest Science*, 59(3): 233-253.

Auld, B.A. and Johnson, S.B. 2014. Invasive alien plant management. *CABI Reviews*, 9(37): 1-12.

Austin, D.F. 1999. Caesar's weed (*Urena lobata*): An invasive exotic or a Florida native. *Wildland Weeds*, 3(1): 13-16.

Averett, J.P., McCune, B., Parks, C.G., Naylor, B.J., DelCurto, T. and Mata-Gonzalez, R., 2016. Non-native plant invasion along elevation and canopy closure gradients in a middle rocky mountain ecosystem. *PloS one*, 11(1): e0147826.

Awan, T.H., Chauhan, B.S. and Cruz, P.C.S., 2014. Influence of environmental factors on the germination of Urena lobata L. and its response to herbicides. *PLoS O*ne, 9(3): e90305.

Ayesa, S.A., Chukwuka, K.S. and Odeyemi, O.O. 2018. Tolerance of *Tithonia diversifolia* and *Chromolaena odorata* in heavy metal simulated-polluted soils and three selected dumpsites. *Toxicology Reports*, 5: 1134-1139.

Azmi, M. 2002. Status and biological control of *Chromolaena odorata* in Malaysia, 27-28. In: Proceedings 2002 Fifth International Workshop on Biological Control and Management of *Chromolaena odorata*, Durban, South Africa.

Babu, S.S., Madhuri, D.B. and Ali, S.L. 2016. A pharmacological review of Urena lobata plant. *Asian Journal of Pharmaceutical and Clinical Research*, 9(2): 20-22.

Bacher, S., Blackburn, T.M., Essl, F., Genovesi, P., Heikkilä, J., Jeschke, J.M., Jones, G., Keller, R., Kenis, M., Kueffer, C. and Martinou, A.F. 2018. Socio-economic impact classification of alien taxa (SEICAT). *Methods in Ecology and Evolution*, 9(1): 159-168.

Baghel, C., Tiwari, A. and Bachkaiya, V. 2023. Study of Correlation between Soil Chemical Properties, Zinc Fractions and Yield of Rice under Long Term Fertilization in Chromustert Soil. *International Journal of Plant & Soil Science,* 35(22): 351-358.

Baillie, J.E.M., Hilton-Taylor, C. and Stuart, S.N. 2004. 2004 IUCN Red List of Threatened Species. A Global Species Assessment. IUCN, Gland, Switzerland and Cambridge, UK. xxiv + 191 pp.

Balangcod, K.D. and Balangcod, A.K.D. 2021. *Chromolaena odorata* (L.) RM King & H. Rob Asteraceae, 1-7. In Ethnobotany of the Mountain Regions of Southeast Asia, Franco, F.M. (eds.). Cham: Springer International Publishing.

Balkrishna, A., Joshi, B., Srivastava, A., Shankar, R., Vashistha, R.K., Kumar, A. and Mishra, R.K. 2021. Medicinal plants of seijosa circle, pakke-kessang district, Arunachal Pradesh, India. *Indian Journal of Natural Products and Resources* (IJNPR), 12(1): 101-115.

Ballard, R. 1986. Bidens pilosa complex (Asteraceae) in North and Central America. *American Journal of Botany*, 73(10): 1452-1465.

Banerjee, A.K. and Sankaran, K.V. 2023. An Action Plan to Prevent and Manage Alien Plant Invasions in India, 397-418. In Plant Invasions and Global Climate Change, Tripathi, S, Bhadouria, R, Srivastava, P., Singh, R and Batish, D.R (eds.). Singapore: Springer Nature Singapore.

Banin, A. and Amiel, A. 1970. A correlative study of the chemical and physical properties of a group of natural soils of Israel. *Geoderma*, 3(3): 185-198.

Bapu, T.D. and Nimasow, G. 2017. Land cover change assessment of Pakke Tiger Reserve (PTR), East Kameng district of Arunachal Pradesh. Methodology, 136: 41.

Baquero, R.A., Oficialdegui, F.J., Ayllón, D. and Nicola, G.G. 2023. The challenge of managing threatened invasive species at a continental scale. Conservation Biology, 37(5): e14165.

Baranová, B., Fazekašová, D. and Manko, P. 2017. Variations of selected soil properties in the grass fields invaded and uninvaded by invasive goldenrod (*Solidago canadensis* L.). Ekológia (Bratislava), 36(2): 101-111.

Barik, S.K. and Adhikari, D. 2012. Predicting the geographical distribution of an invasive species (*Chromolaena odorata* L. (King) & HE Robins) in the Indian subcontinent under climate change scenarios, 77-88. In Invasive alien plants: an ecological appraisal for the Indian subcontinent, Bhatt, J.R., Singh, S, Singh, S.P., Tripathi, R.S. and Kohli, R.K.(eds.). Wallingford UK: CABI.

Barney, J.N. 2006. North American history of two invasive plant species: phytogeographic distribution, dispersal vectors, and multiple introductions. B*iological Invasions,* 8: 703-717.

Barney, J.N., Tekiela, D.R., Barrios-Garcia, M.N., Dimarco, R.D., Hufbauer, R.A., Leipzig-Scott, P., Nunez, M.A., Pauchard, A., Pyšek, P., Vítková, M. and Maxwell, B.D. 2015. Global Invader Impact Network (GIIN): toward standardized evaluation of the ecological impacts of invasive plants. *Ecology and Evolution,* 5(14): 2878-2889.

Barney, J.N., Tekiela, D.R., Dollete, E.S. and Tomasek, B.J. 2013. What is the “real” impact of invasive plant species?. Frontiers in Ecology and the Environment, 11(6): 322-329.

Barrett, A. and Brown, L. 2021. Effects of rainfall, temperature and photoperiod on the phenology of ephemeral resources for selected bushveld woody plant species in southern Africa. *Plos one*, 16(5): e0251421.

Bartolome, A.P., Villaseñor, I.M. and Yang, W.C. 2013. *Bidens pilosa* L. (Asteraceae): botanical properties, traditional uses, phytochemistry, and pharmacology. *Evidence-based complementary and alternative medicine,* 2013(1): 340215.

Bartz, R. and Kowarik, I. 2019. Assessing the environmental impacts of invasive alien plants: a review of assessment approaches. *NeoBiota,* 43: 69-99.

Barve, V.V., Brenskelle, L., Li, D., Stucky, B.J., Barve, N.V., Hantak, M.M., McLean, B.S., Paluh, D.J., Oswald, J.A., Belitz, M.W. and Folk, R.A. 2020. Methods for broad-scale plant phenology assessments using citizen scientists' photographs. *Applications in Plant Sciences*, 8(1): e11315.

Basistha, B.C., Sharma, N.P., Lepcha, L., Arrawatia, M.L. and Sen, A. 2010. Ecology of *Hippophae salicifolia* D. Don of temperate and sub-alpine forests of North Sikkim Himalayas - a case study. *Symbiosis*, 50: 87-95.

Batish, D.R., Kaur, S., Singh, H.P. and Kohli, R.K. 2009. Nature of interference potential of leaf debris of Ageratum conyzoides. *Plant Growth Regulation*, 57: 137-144.

Bayer, C, and Kubitzky, K. 2003. Malvaceae, 225-311. In *The families and genera of vascular plants,* Kubitzky K and Bayer C (eds.). Berlin, Heidelberg and New York, Germany and USA: Springer.

Beaubien, E.G. and Freeland, H.J. 2000. Spring phenology trends in Alberta, Canada: links to ocean temperature. *International Journal of Biometeorology*, 44: 53-59.

Behera, M.D. and Kushwaha, S.P.S., 2007. An analysis of altitudinal behavior of tree species in Subansiri district, Eastern Himalaya, 277-291. In Plant conservation and biodiversity, Hawksworth, D.L., Bull, A.T. (eds), Springer, Dordrecht.

Behera, S.K. and Shukla, A.K. 2015. Spatial distribution of surface soil acidity, electrical conductivity, soil organic carbon content and exchangeable potassium, calcium and magnesium in some cropped acid soils of India. *Land Degradation & Development,* 26(1): 71-79.

Bell, C.E., Wilen, C.A. and Stanton, A.E. 2003. Invasive plants of horticultural origin. *Hortscience*, 38(1): 14-16.

Bellard, C., Cassey, P. and Blackburn, T.M. 2016. Alien species as a driver of recent extinctions. *Biology letters*, 12(2): 20150623.

Bellini, G., Schrieber, K., Kirleis, W. and Erfmeier, A. 2024. Exploring the complex pre-adaptations of invasive plants to anthropogenic disturbance: a call for integration of archaeobotanical approaches. *Frontiers in Plant Science*, 15: 1307364.

Belnap, J. and Phillips, S.L. 2001. Soil biota in an ungrazed grassland: response to annual grass (*Bromus tectorum*) invasion. *Ecological applications*, 11(5): 1261-1275.

Beniwal B.S. 1987. Phenological study of trees in Arunachal Pradesh. Indian Forester. 113 (12): 779- 791.

Bernal, R., Gradstein, S.R. and Celis, M. 2015. Catálogo de plantas y líquenes de Colombia. Instituto de Ciencias Naturales, Universidad Nacional de Colombia, Bogotá

Bhatta, S., Hejda, M. and Pyšek, P. 2024. Impact of invasive plants on vegetation in protected areas of Nepal. *Biological Invasions,* 26(11): 3745-3759.

Bhattarai, K.R., Måren, I.E. and Subedi, S.C. 2014. Biodiversity and invasibility: Distribution patterns of invasive plant species in the Himalayas, Nepal. *Journal of Mountain Science,* 11: 688-696.

Binggeli P. 1999. *Chromolaena odorata* (L.) King & Robinson (Asteraceae). p.4.http://www.fs.fed.us/global/iitf/pdf/shrubs/Chromolaena%20odoratum.pdf.

Binggeli, P. 1996. A taxonomic, biogeographical and ecological overview of invasive woody plants. *Journal of Vegetation Science,* 7(1): 121-124.

Birand, A. and Pawar, S. 2004. An ornithological survey in north-east India. *Forktail,* 20: 15-24.

Bisht, V.K., Kuniyal, C.P., Bhandari, A.K., Nautiyal, B.P. and Prasad, P. 2014. Phenology of plants in relation to ambient environment in a subalpine forest of Uttarakhand, western Himalaya. *Physiology and Molecular Biology of Plants,* 20: 399-403.

Blackburn, T.M., Bellard, C. and Ricciardi, A. 2019. Alien versus native species as drivers of recent extinctions. *Frontiers in Ecology and the Environment,* 17(4): 203-207.

Blackburn, T.M., Essl, F., Evans, T., Hulme, P.E., Jeschke, J.M., Kühn, I., Kumschick, S., Marková, Z., Mrugała, A., Nentwig, W. and Pergl, J. 2014. A unified classification of alien species based on the magnitude of their environmental impacts. *Plos Biology,* 12(5): e1001850.

Blank, R.R. and Young, J.A. 2002. Influence of the exotic invasive crucifer, *Lepidium latifolium,* on soil properties and elemental cycling. *Soil Science,* 167(12): 821-829.

Boadie-Ampong, M. and Nishi, M., 2024. Exploring the benefits of invasive alien plant species for human well-being: a systematic review of the state-of-the-art and directions for prospective research. *Discover Sustainability,* 5(1): 329.

Bor, N.L. 1940. Flora of Assam, Vol.5. Assam Govt. Press, Shillong.

Borah, D., Taram, M., Tasser, M. and Hughes, M. 2024. A new species *Begonia ziroensis* and a new record of Begonia siamensis from Arunachal Pradesh, Northeast India. *New Zealand Journal of Botany*, 63(1): 1-8.

Borchert, R. 1983. Phenology and control of flowering in tropical trees. *Biotropica*, 15(2): 81-89.

Bosch, C.H. 2012. *Triumfetta rhomboidei*, 446-449. In *Plant Resources of Tropical Africa, No. 16: Fibres*, Brink M, and Achigan-Dako E.G (eds.), Wageningen, Netherlands: PROTA Foundation/CTA.

Bovini, M., Oliveira, R., Faria, M. and Kurtz, B. 2014. Floristic diversity of the Cagarras Islands Natural Monument, Rio de Janeiro, Brazil. *Check list*, 10(2): 366-373.

Bradley, B.A., Oppenheimer, M. and Wilcove, D.S. 2009. Climate change and plant invasions: restoration opportunities ahead?. *Global Change Biology*, 15(6): 1511-1521.

Brady, N.C. 1984. The nature and properties of soils, +750pp.

Brooks, M.L., D'antonio, C.M., Richardson, D.M., Grace, J.B., Keeley, J.E., DiTomaso, J.M., Hobbs, R.J., Pellant, M. and Pyke, D. 2004. Effects of invasive alien plants on fire regimes. *BioScience*, 54(7): 677-688.

Brooks, T.M., Mittermeier, R.A., Da Fonseca, G.A., Gerlach, J., Hoffmann, M., Lamoreux, J.F., Mittermeier, C.G., Pilgrim, J.D. and Rodrigues, A.S. 2006. Global biodiversity conservation priorities. *Science*, 313(5783): 58-61.

Brown, K.A. and Gurevitch, J., 2004. Long-term impacts of logging on forest diversity in Madagascar. *Proceedings of the National Academy of Sciences*, 101(16): 6045-6049.

Brown, K.A., Scatena, F.N. and Gurevitch, J. 2006. Effects of an invasive tree on community structure and diversity in a tropical forest in Puerto Rico. *Forest Ecology and Management*, 226(1-3): 145-152.

BSI 2020. Invasive Alien species. **http://www.bsienvis.nic.in/database/invasive_alien_species_15896.aspx**. (Accessed on 29.05.2024).

Bucher, S.F. and Römermann, C. 2020. Flowering patterns change along elevational gradients and relate to life-history strategies in 29 herbaceous species. *Alpine Botany*, 130(1): 41-58.

Buragohain, J. 2011. Ethnomedicinal plants used by the ethnic communities of Tinsukia district of Assam, India. *Recent research in Science and Technology*, 3(9): 31-42.

Burke, M.J. and Grime, J.P. 1996. An experimental study of plant community invasibility. *Ecology*, 77(3): 776-790.

Burkill, H.M. 1985. The Useful Plants of West Tropical Africa. Royal Botanical Gardens, Kew, +981pp.

Burkill, I.H. 1935. A dictionary of the economic products of the Malay Peninsula. Volumes 1 and 2. London, UK: Governments of the Straits Settlements and Federal Malay States, Crown Agents for the Colonies.

Burns, J.H., Pardini, E.A., Schutzenhofer, M.R., Chung, Y.A., Seidler, K.J. and Knight, T.M. 2013. Greater sexual reproduction contributes to differences in demography of invasive plants and their noninvasive relatives. *Ecology,* 94(5): 995-1004.

Bushi, D., Bam, K., Mahato, R., Nimasow, G., Dai Nimasow, O. and Tag, H. 2021. Ethnomedicinal plants used by the indigenous tribal communities of Arunachal Pradesh, India: a review. *Ethnobotany Research and Applications*, 22: 1-40.

Butchart, S.H., Walpole, M., Collen, B., Van Strien, A., Scharlemann, J.P., Almond, R.E., Baillie, J.E., Bomhard, B., Brown, C., Bruno, J. and Carpenter, K.E. 2010. Global biodiversity: indicators of recent declines. *Science,* 328(5982): 1164-1168.

Byrne, M. and Warren, R. 2024. Intra and interspecific competition via allelopathy among native and non-native plants. *Biology Theses,* 53.

Byun, C., de Blois, S. and Brisson, J. 2018. Management of invasive plants through ecological resistance. *Biological Invasions,* 20: 13-27.

Caicedo, A.L., Stinchcombe, J.R., Olsen, K.M., Schmitt, J. and Purugganan, M.D., 2004. Epistatic interaction between Arabidopsis FRI and FLC flowering time genes generates a latitudinal cline in a life history trait. *Proceedings of the National Academy of Sciences*, 101(44), pp.15670-15675.

CABI 2024. CABI Compendium. Wallingford, UK: CAB International. https://www.cabi.org/. (Accessed on 29.05.2024).

Cadotte, M. and Lovett-Doust, J. 2002. Ecological and taxonomic differences between rare and common plants of southwestern Ontario. *Ecoscience*, 9(3): 397-406.

Calder, I.R. and Dye, P. 2001. Hydrological impacts of invasive alien plants. *Land use and water resources research*, 1: 1-12.

Calizza, E., Rossi, L., Careddu, G., Sporta Caputi, S. and Costantini, M.L. 2021. A novel approach to quantifying trophic interaction strengths and impact of invasive species in food webs. *Biological Invasions*, 23(7): 2093-2107.

Call, N.M. 1998. Biology and physiology of the noxious weed, tropical soda apple (*Solanum viarum* Dunal), Ph.D. thesis, North Carolina State University.

Callaway, R., Newingham, B., Zabinski, C.A. and Mahall, B.E. 2001. Compensatory growth and competitive ability of an invasive weed are enhanced by soil fungi and native neighbours. *Ecology Letters*, 4(5): 429-433.

Callaway, R.M. and Aschehoug, E.T., 2000. Invasive plants versus their new and old neighbors: a mechanism for exotic invasion. Science, 290(5491), pp.521-523.

Callaway, R.M., Cipollini, D., Barto, K., Thelen, G.C., Hallett, S.G., Prati, D., Stinson, K. and Klironomos, J. 2008. Novel weapons: invasive plant suppresses fungal mutualists in America but not in its native Europe. *Ecology*, 89(4): 1043-1055.

Callaway, R.M., Thelen, G.C., Rodriguez, A. and Holben, W.E. 2004. Soil biota and exotic plant invasion. *Nature*, 427(6976): 731-733.

Camarero, J.J. and Rubio-Cuadrado, Á. 2024. Leaf phenology and tree growth are uncoupled in three deciduous tree species. *Trees, Forests and People*, p.100761.

Campagnaro, T., Brundu, G. and Sitzia, T. 2018. Five major invasive alien tree species in European Union forest habitat types of the Alpine and Continental biogeographical regions. *Journal for Nature Conservation*, 43: 227-238.

Cao, H.L., Feng, H.L., Deng, X., Zhang, L.Y. and Ye, W.H. 2003. Study on invasion, harm, prevention and control of Mikania micrantha, 239-248. In Biological Invasion: Theory and Practice, Xu, R.M and Ye, W. Y (eds.), Science Press, Beijing, (inChinese).

Capinha, C., Essl, F., Seebens, H., Moser, D. and Pereira, H.M. 2015. The dispersal of alien species redefines biogeography in the Anthropocene. *Science*, 348(6240): 1248-1251.

Caplat, P., Anand, M. and Bauch, C. 2010. Modelling invasibility in endogenously oscillating tree populations: timing of invasion matters. *Biological Invasions*, 12: 219-231.

Captain, A., Deepak, V., Pandit, R., Bhatt, B. and Athreya, R. 2019. A new species of pitviper (Serpentes: Viperidae: *Trimeresurus Lacepède*, 1804) from west Kameng District, Arunachal Pradesh, India. *Russian Journal of Herpetology,* 26(2): 111-122.

Cara, P.A.D.A. 2006. Efeito de borda sobre a fenologia, as síndromes de polinização e a dispersão de sementes de uma comunidade arbórea na Floresta Atlântica ao norte do Rio São Francisco., Brazil. Ph.D. thesis, Universidade Federal de Pernambuco.

Cara-Donna, P.J., Iler, A.M. and Inouye, D.W. 2014. Shifts in flowering phenology reshape a subalpine plant community. *Proceedings of the National Academy of Sciences,* 111(13): 4916-4921.

Carlton, J.T. 2009. Deep invasion ecology and the assembly of communities in historical time, 13-56. In Biological invasions in marine ecosystems: ecological, management, and geographic perspectives, Rilov. G and Crooks, J.A. (eds.), Berlin, Heidelberg: Springer Berlin Heidelberg.

Castro-Diez, P., Fierro-Brunnenmeister, N., González-Muñoz, N. and Gallardo, A. 2012. Effects of exotic and native tree leaf litter on soil properties of two contrasting sites in the Iberian Peninsula. *Plant and Soil,* 350: 179-191.

Castro-Díez, P., Godoy, O., Saldaña, A. and Richardson, D.M. 2011. Predicting invasiveness of Australian acacias on the basis of their native climatic affinities, life history traits and human use. *Diversity and distributions,* 17(5): 934-945.

Castro-Díez, P., Vaz, A.S., Silva, J.S., Van Loo, M., Alonso, Á., Aponte, C., Bayón, Á., Bellingham, P.J.,Chiuffo, M.C., DiManno, N. and Julian, K. 2019. Global effects of non-native tree species on multiple ecosystem services. *Biological Reviews,* 94(4): 1477-1501.

CBD. 2002. Sixth Conference of the Parties, the Hague, the Netherlands. 7–19 April 2002: Decision VI/23: Alien Species that Threaten Ecosystems, Habitats or Species to Which is Annexed Guiding Principles for the Prevention, Introduction and Mitigation of Impacts of Alien Species that Threaten Ecosystems, Habitats or Species.

CBD. 2009. Convention on Biological Diversity. (Accessed on 21.05.2024).

Celesti-Grapow, L., Di Marzio, P. and Blasi, C. 2003. Temporal niche separation of the alien flora of Rome Italy, 101-111. In Plant invasions: Ecological threats and management solutions, L.E. Child, J.H. Brock, G. Brundu, K. Prach, P. Pyšek, P.M. Wade and M. Williamson (eds.), Backhuys Publishers, Leiden, The Netherlands.

Census of India. 2011. Office of registrar general and census commissioner of India, ministry of home affairs, Government of India, https://www.census2011.co.in/census/state/arunachal+pradesh.html.

Chacon, N., Herrera, I., Flores, S., González, J.A. and Nassar, J.M. 2009. Chemical, physical, and biochemical soil properties and plant roots as affected by native and exotic plants in Neotropical arid zones. *Biology and Fertility of Soils,* 45: 321-328.

Champion, H.G. and Seth, S.K. 1968. A Revised Survey of the Forest Types of India. Manager of publications.

Chandra, V. and Srivastava, S.N. 1978. *Solanum viarum* Dunal syn. *Solanum khadianum* Clarke, a crop for production of Solasadine. *Indian Drugs* 16: 53–60.

Chao, A., Kubota, Y., Zelený, D., Chiu, C.H., Li, C.F., Kusumoto, B., Yasuhara, M., Thorn, S., Wei, C.L., Costello, M.J. and Colwell, R.K. 2020. Quantifying sample completeness and comparing diversities among assemblages. *Ecological Research*, 35(2): 292-314.

Chapin, F.S., Autumn, K. and Pugnaire, F., 1993. Evolution of suites of traits in response to environmental stress. *The American Naturalist,* 142, pp.S78-S92.

Chapin, F.S., Zavaleta, E.S., Eviner, V.T., Naylor, R.L., Vitousek, P.M., Reynolds, H.L., Hooper, D.U., Lavorel, S., Sala, O.E., Hobbie, S.E. and Mack, M.C. 2000. Consequences of changing biodiversity. *Nature*, 405(6783): 234-242.

Chapin, F.S., Matson, P.A., Mooney, H.A. and Vitousek, P.M. 2002. Principles of terrestrial ecosystem ecology. Springer Verlag, New York, + 436pp.

Chapman, C.A., Wrangham, R.W., Chapman, L.J., Kennard, D.K. and Zanne, A.E. 1999. Fruit and flower phenology at two sites in Kibale National Park, Uganda. *Journal of Tropical ecology*, 15(2): 189-211.

Chaudhary, A., Adhikari, B.S., Joshi, N.C. and Rawat, G.S. 2019. Patterns of invasion by crofton weed (*Ageratina adenophora*) in Kailash sacred landscape region of western Himalaya (India). *Environment Conservation Journal*, 20(3): 9-17.

Chauhan, B.S. and Johnson, D.E. 2009. Seed germination and seedling emergence of synedrella (*Synedrella nodiflora*) in a tropical environment. *Weed Science*, 57(1): 36-42.

Chauhan, B.S., Ali, H.H. and Florentine, S. 2019. Seed germination ecology of Bidens pilosa and its implications for weed management. *Scientific Reports*, 9(1): 16004.

Chazdon, R.L. 2008. Beyond deforestation: restoring forests and ecosystem services on degraded lands. *Science*, 320(5882): 1458-1460.

Chen, D. and van Kleunen, M. 2022. Invasional meltdown mediated by plant–soil feedbacks may depend on community diversity. *New Phytologist*, 235(4): 1589-1598.

Chen, X., Liu, Y., Liu, H., Wang, H., Yang, D. and Huangfu, C. 2015. Impacts of four invasive Asteraceae on soil physico-chemical properties and AM fungi community. *American Journal of Plant Sciences*, 6(17): 2734-2743.

Chengzhen, W., Wei, H., Jilin, W. and Qunrui, Z. 2000. Studies on Kernel density estimation of species abundance distribution in two communities of rare and endangered plants. *Journal of Tropical and Subtropical Botany*, 8(4): 301-307.

Chevalier, A. 1947. Les Jujubiers ou Ziziphus de l'Ancien monde et l'utilisation de leurs fruits. *Journal d'agriculture traditionnelle et de botanique appliquée*, 27(301): 470-483.

Chou, C.H., Hsu, Y.M., Huang, T.J., Liu, F.C. and Weng, J.R. 2012. Sterodial sapogenins from *Solanum torvum*. *Biochemical Systematics and Ecology*, 45: 108-110.

Chowdhery, H.J. 1998. Orchid Flora of Arunachal Pradesh: HJ Chowdhery. Bishen Sing Mahendra Pal Singh.

Chowdhery, H.J., G.S.Giri, G.D. Pal, A. Pramanik & S.K. Das. 1996. Materials for the Flora of Arunachal Pradesh. Vol.I. Botanical Survey of India, Calcutta. +. 693pp.

Christian, J.M. and Wilson, S.D. 1999. Long-term ecosystem impacts of an introduced grass in the northern Great Plains. *Ecology*, 80(7): 2397-2407.

Chuine, I. and Beaubien, E.G. 2001. Phenology is a major determinant of tree species range. *Ecology Letters*, 4(5): 500-510.

Chuine, I. 2010. Why does phenology drive species distribution?. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 365(1555): 3149-3160.

Chuine, I., Cambon, G. and Comtois, P. 2000. Scaling phenology from the local to the regional level: advances from species-specific phenological models. *Global Change Biology*, 6(8): 943-952.

Chytrý, M., Pyšek, P., Tichý, L., Knollová, I. and Danihelka, J. 2005. Invasions by alien plants in the Czech Republic: a quantitative assessment across habitats. Preslia, 77(4): 339-354.

Chytrý, M., Pyšek, P., Wild, J., Pino, J., Maskell, L.C. and Vilà, M. 2009. European map of alien plant invasions based on the quantitative assessment across habitats. Diversity and Distributions, 15(1): 98-107.

Clarke, C. B. 1876. Compositae Indicae: Descriptae et secus Genera Benthami Ordinatae. (Calcutta: Thacker Spink and Co.)

Cleland, E.E., Chuine, I., Menzel, A., Mooney, H.A. and Schwartz, M.D. 2007. Shifting plant phenology in response to global change. *Trends in ecology & evolution*, 22(7): 357-365.

Cleland, E.E., Chuine, I., Menzel, A., Mooney, H.A. and Schwartz, M.D., 2007. Shifting plant phenology in response to global change. *Trends in ecology & evolution*, 22(7), pp.357-365.

Cleland, E.E., Esch, E. and McKinney, J. 2015. Priority effects vary with species identity and origin in an experiment varying the timing of seed arrival. *Oikos*, 124(1): 33-40.

Coile, N.C. 1993. Tropical soda apple, *Solanum viarum* Dunal: The plant from hell. *Botany circular*, 27.

Colautti, R.I., Ågren, J. and Anderson, J.T. 2017. Phenological shifts of native and invasive species under climate change: insights from the Boechera–Lythrum model. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 372(1712): 20160032.

Collins, R.J., Copenheaver, C.A., Barney, J.N. and Radtke, P.J. 2020. Using invasional meltdown theory to understand patterns of invasive richness and abundance in forests of the northeastern USA. *Natural Areas Journal*, 40(4): 336-344.

Comole, A.A., Malan, P.W. and Tiawoun, M.A.P. 2021. Effects of *Prosopis velutina* Invasion on Soil Characteristics along the Riverine System of the Molopo River in North-West Province, South Africa. *International Journal of Ecology*,1: 6681577.

Corbin, J.D. and D'Antonio, C.M. 2004. Effects of exotic species on soil nitrogen cycling: Implications for restoration1. *Weed Technology*, 18(sp1): 1464-1467.

Corlett, R.T. 1988. The naturalized flora of Singapore. *Journal of Biogeography*, 15: 657-663.

Cornelius, C., Leingärtner, A., Hoiss, B., Krauss, J., Steffan-Dewenter, I. and Menzel, A. 2013. Phenological response of grassland species to manipulative snowmelt and drought along an altitudinal gradient. *Journal of experimental botany,* 64(1): 241-251.

Cortés-Flores, J., Andresen, E., Cornejo-Tenorio, G., and Ibarra-Manríquez, G. 2013. Fruiting phenology of seed dispersal syndromes in a Mexican Neotropical temperate forest, Forest Ecology and Management, 289, pp. 445–454. https://doi.org/10.1016/j.foreco.2012.10.025

Crawley, M.J., Harvey, P.H. and Purvis, A.N.D.A. 1996. Comparative ecology of the native and alien floras of the British Isles. *Philosophical Transactions of the Royal Society of London. Series B: Biological Sciences,* 351(1345): 1251-1259.

Crimmins, T.M., Crimmins, M.A. and Bertelsen, C.D. 2013. Spring and summer patterns in flowering onset, duration, and constancy across a water-limited gradient. *American Journal of Botany,* 100(6): 1137-1147.

Cristina-Maria, A.V.R.A.M., Proorocu, M., Mălinaș, A. and Mălinaș, C. 2023a. A Review of European Union Strategies for Biodiversity Conservation: Actions Against Invasive Alien Plant Species. *ProEnvironment Promediu*, 16(55).

Cristina-Maria, A.V.R.A.M., Proorocu, M., Mălinaș, A. and Mălinaș, C. 2023b. Unraveling Plant Invasion: Insights into Major Pathways and Mechanisms of Invasive Species Spread. *ProEnvironment Promediu,* 16(55).

Cronk, Q.C. and Fuller, J.L. 1995. Plant invaders: the threat to natural ecosystems. London: Chapman & Hall.

Cruttwell, R. 1972. The insects of *Eupatorium odoratum* L. in Trinidad and their potential as agents for biological control, Ph.D. thesis, University of the West Indies, Trinidad.

Cruz, R.V., Harasawa, H., Lal, M., Wu, S., Anokhin, Y., Punsalmaa, B., Honda, Y., Jafari, M., Li, C. and Huu Ninh, N. 2007. Climate Change 2007: Impacts, Adaptation and Vulnerability, 46-506. In Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Parry, M.L (eds.), Cambridge University Press, Cambridge.

Cuda, J., Vitkova, M., Albrechtova, M., Guo, W.Y., Barney, J.N. and Pyšek, P. 2017. Invasive herb *Impatiens glandulifera* had minimal impact on multiple components of temperate forest ecosystem function. *Biological Invasions*, 19; 3051-3066.

Curtis, J.T and Cotton, G. 1956. Plant ecology workbook. A laboratory, field and reference manual.

Curtis, J.T. and McIntosh, R.P. 1950. The interrelations of certain analytic and synthetic phytosociological characters. *Ecology*, 31(3): 434-455.

Curtis, J.T. 1959. The vegetation of Wisconsin: an ordination of plant communities. University of Wisconsin Press.

da Silva, W.B., Périco, E., Dalzochio, M.S., Santos, M. and Cajaiba, R.L. 2018. Are litterfall and litter decomposition processes indicators of forest regeneration in the neotropics? Insights from a case study in the Brazilian Amazon. Forest *Ecology and Management*, 429: 189-197.

Daehler, C.C. 2003. Performance comparisons of co-occurring native and alien invasive plants: implications for conservation and restoration. *Annual Review of Ecology, Evolution, and Systematics*, 34(1): 183-211.

Dainese, M., Aikio, S., Hulme, P.E., Bertolli, A., Prosser, F. and Marini, L. 2017. Human disturbance and upward expansion of plants in a warming climate. *Nature Climate Change*, 7(8): 577-580.

D'antonio, C.A.R.L.A. and Meyerson, L.A. 2002. Exotic plant species as problems and solutions in ecological restoration: a synthesis. *Restoration ecology*, 10(4); 703-713.

D'Antonio, C.M. and Vitousek, P.M. 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. *Annual Review of Ecology and Systematics*, 23(1): 63-87.

Dar, P.A., Reshi, Z.A. and Shah, A.B. 2018. Altitudinal distribution of native and alien plant species along roadsides in Kashmir Himalaya, *Tropical Ecology*, 59(1): 45-55.

Darwin, C. 1859. On the Origin of Species by Means of Natural Selection, or the Preservation of Favoured Races in the Struggle for Life, John Murray, London,+502pp..

Das, A.P. and Lahiri, A.K. 1997. Phytosociological studies of the ground flora in different types of vegetation on Tiger Hill, Darjeeling District, West Bengal. *Indian Forester*, 123(12): 1176-1187

Das, J. and Agarwala, B.K. 2011. Changes in leaf chemicals in different phenological stages of *Chromolaena odorata* L.(King and Robinson) from Tripura. *Vegetos*, 24: 38-40.

Dasgupta, S., Bhattacharya, T. and Kaul, R. 2022. Role of vegetation characteristics on the distribution of three hornbill species in and around Pakke Tiger Reserve, Arunachal Pradesh, India. *Current Science*,120: 617-625.

Dash, S.S., Panday, S., Rawat, D.S., Kumar, V., Lahiri, S., Sinha, B.K. and Singh, P. 2021. Quantitative assessment of vegetation layers in tropical evergreen forests of Arunachal Pradesh, Eastern Himalaya, India. *Current Science*,120(5): 850-858.

da Silva, W.B., Périco, E., Dalzochio, M.S., Santos, M. and Cajaiba, R.L., 2018. Are litterfall and litter decomposition processes indicators of forest regeneration in the neotropics? Insights from a case study in the Brazilian Amazon. *Forest Ecology and Management*, 429, pp.189-197.

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

Datta A. 2000. An ecological study of sympatric hornbills and fruiting patterns in a tropical forest in Arunachal Pradesh, Ph.D. thesis, Saurashtra University, Gujarat.

Datta, A. and Goyal, S.P. 1997. Response of Arboreal Mammals to Selective Logging in Arunanchal Pradesh. Final Report, WII, Dehradun.

Datta, A. 1998. Hornbill abundance in unlogged forest, selectively logged forest and a forest plantation in Arunachal Pradesh, India. *Oryx*, 32(4): 285-294.

Datta, A., Singh, P., Athreya, R.M. and Karthikeyan, S. 1998. Birds of Pakhui Wildlife Sanctuary in western Arunachal Pradesh, North East India. *Newsletter for Birdwatchers*, 38(6): 91-96.

Datta, A., 2001. An ecological study of sympatric hornbills and fruiting patterns in a tropical forest in Arunachal Pradesh. PhD Thesis, Saurashtra University, Gujarat.

Davidse, G., Sousa, M.S., Knapp, S. and Chiang Cabrera, F. 2018. Flora Mesoamericana vol. 5, part 2. Asteraceae. Universidad Nacional Autónoma de México, Missouri Botanical Garden Press, and the Natural History Museum (London).

Davidson, A.M., Jennions, M. and Nicotra, A.B. 2011. Do invasive species show higher phenotypic plasticity than native species and, if so, is it adaptive? A meta-analysis. *Ecology letters*, 14(4): 419-431.

Davis, M.A., 2009. Invasion biology. Oxford University Press.

Davis, M.A., Grime, J.P. and Thompson, K., 2000. Fluctuating resources in plant communities: a general theory of invasibility. *Journal of ecology*, 88(3): 528-534.

Davis, M.A., Thompson, K. and Grime, J.P. 2001. Charles S. Elton and the dissociation of invasion ecology from the rest of ecology. *Diversity and Distributions*, 7(1-2): 97-102.

Day, M.D., Clements, D.R., Gile, C., Senaratne, W.K., Shen, S., Weston, L.A. and Zhang, F. 2016. Biology and impacts of Pacific Islands invasive species. 13. *Mikania micrantha* Kunth (Asteraceae) 1. *Pacific Science*, 70(3): 257-285.

Day, M.D., Kawi, A., Kurika, K., Dewhurst, C.F., Waisale, S., Saul-Maora, J., Fidelis, J., Bokosou, J., Moxon, J., Orapa, W. and Senaratne, K.A.D. 2012. *Mikania micrantha* Kunth (Asteraceae)(Mile-a-Minute): Its Distribution and Physical and Socioeconomic Impacts in Papua New Guinea1. *Pacific Science*, 66(2): 213-223**.**

de Beurs, K.M. and Henebry, G.M. 2008. Northern annular mode effects on the land surface phenologies of northern Eurasia. *Journal of Climate*, 21(17): 4257-4279.

De Candolle, A. P. 1855. Géographie Botanique Raisonné, vol. 2. V. Masson, Paris.

De la Haba, P., De la Mata, L., Molina, E. and Agüera, E. 2014. High temperature promotes early senescence in primary leaves of sunflower (*Helianthus annuus* L.) plants. *Canadian Journal of Plant Science*, 94(4): 659-669.

De, J., Mandal, A.K. and Ghosh, M. 2006. Zool. Surv. India, Fauna of Arunachal Pradesh, State Fauna Series, *Fauna of Arunachal Pradesh*, (13): 21-68.

Deb, S., Barbhuiya, A.R., Arunachalam, A. and Arunachalam, K. 2008. Ecological analysis of traditional agroforest and tropical forest in the foothills of Indian eastern Himalaya: vegetation, soil and microbial biomass. *Tropical Ecology*, 49(1): 73.

Debnath, A., Rajib, P. and Debnath, B. 2018. Effects of the Invasive shrub, *Chromolaena odorata* on Soil Properties in the Atharamura forest ecosystem: Indian Himalayan state of Tripura, North East India. *Vegetos*-An International Journal of Plant Research & Biotechnology, 31(2): 77-90.

Dech, J.P. and Nosko, P. 2004. Rapid growth and early flowering in an invasive plant, purple loosestrife (*Lythrum salicaria* L.) during an El Nino spring. *International Journal of Biometeorolog*y, 49: 26-31.

Deka, B.C., Thirugnanavel, A., Patel, R.K., Nath, A. and Deshmukh, N. 2012. Horticultural diversity in North-East India and its improvement for value addition. *Indian Journal of Genetics and Plant Breeding*, 72(02): 157-167.

Denny, E.G., Gerst, K.L., Miller-Rushing, A.J., Tierney, G.L., Crimmins, T.M., Enquist, C.A., Guertin, P., Rosemartin, A.H., Schwartz, M.D., Thomas, K.A. and Weltzin, J.F. 2014. Standardized phenology monitoring methods to track plant and animal activity for science and resource management applications. *International Journal of Biometeorology,* 58(4): 591-601.

Desai, P.B. and Patel, N.K. 2010. 1. Phenological Study of Trees Species Of Satlasana Range Forest (North Gujarat). *Life sciences Leaflets*, 3: 41-46.

Dey, A., Dwivedi, B.S., Meena, M.C., Datta, S.P. and Kumar, A. 2014. Soil boron fractions and their contribution towards boron availability and uptake by wheat on a Typic haplustept under long-term fertilization. *Journal of the Indian Society of Soil Science*, 62(4): 384-390.

Dhakal, S., Shrestha, B.B., Sharma, K.P., Paudel, S. and Siwakoti, M. 2024. Grasslands are more vulnerable to plant invasions than forests in south-central Nepal. *Environmental Challenge*s, 15: 100929.

Diagne, C., Leroy, B., Gozlan, R.E., Vaissière, A.C., Assailly, C., Nuninger, L., Roiz, D., Jourdain, F., Jarić, I. and Courchamp, F. 2020. InvaCost, a public database of the economic costs of biological invasions worldwide. *Scientific data*, 7(1): 277.

Diaz, H.F., Grosjean, M. and Graumlich, L. 2003. Climate variability and change in high elevation regions: past, present and future. *Climatic Change*, 59(1): 1-4.

Dickson, T.L., Hopwood, J.L. and Wilsey, B.J., 2012. Do priority effects benefit invasive plants more than native plants? An experiment with six grassland species. *Biological Invasions*, 14: 2617-2624.

Diekmann, M. and Lawesson, J.E. 1999. Shifts in ecological behaviour of herbaceous forest species along a transect from northern Central to North Europe. *Folia Geobotanica,* 34: 127-141.

Diekmann, M. 1996. Relationship between flowering phenology of perennial herbs and meteorological data in deciduous forests of Sweden. *Canadian Journal of Botany*, 74(4): 528-537.

Diekmann, M., Effertz, H., Baranowski, M. and Dupré, C. 2016. Weak effects on plant diversity of two invasive *Impatiens* species. *Plant Ecology*, 217: 1503-1514.

Dlugosch, K.M. and Parker, I.M., 2008. Invading populations of an ornamental shrub show rapid life history evolution despite genetic bottlenecks. *Ecology letters*, 11(7): 701-709.

Dobrylovska, D. 2001. Litter decomposition of red oak, larch and lime tree and its effect on selected soil characteristics. *Journal of Forest Science-UZPI (Czech Republic*), 47(11): 477-485.

Dogra, K.S., Kohli, R.K., Sood, S.K. and Dobhal, P.K. 2009. Impact of Ageratum conyzoides L. on the diversity and composition of vegetation in the Shivalik hills of Himachal Pradesh (Northwestern Himalaya), India. *International Journal of Biodiversity and Conservation,* 1(5): 135-145.

Downey, P.O. and Richardson, D.M. 2016. Alien plant invasions and native plant extinctions: a six-threshold framework. *AoB plants*, 8: plw047.

Drake, J.A., Mooney, H.A., Di Castri, F., Groves, R.H., Kruger, F.J., Rejmanek, M., Williamson, M. and Invasions, B. 1989. A global perspective. Chichester, UK: Wiley & Sons.

Drake, J.M., Drury, K.L., Lodge, D.M., Blukacz, A., Yan, N.D. and Dwyer, G. 2006. Demographic stochadticity, environmental variability, and windows of invasion risk for *Bythotrephes longimanus* in North America. *Biological Invasions*, 8: 843-861.

Dreiss, L.M., 2016. The role of phenology in invasive plant species success in temperate forest understories. Ph.D.thesis, University of Connecticut Graduate School.

Duda, J.J., Freeman, D.C., Emlen, J.M., Belnap, J., Kitchen, S.G., Zak, J.C., Sobek, E., Tracy, M. and Montante, J. 2003. Differences in native soil ecology associated with invasion of the exotic annual chenopod, *Halogeton glomeratus*. *Biology and Fertility of Soils,* 38: 72-77.

Duenas, M.A., Hemming, D.J., Roberts, A. and Diaz-Soltero, H. 2021. The threat of invasive species to IUCN-listed critically endangered species: A systematic review. *Global Ecology and Conservation*, 26: e01476.

Dunham, A.E., Razafindratsima, O.H., Rakotonirina, P. and Wright, P.C. 2018. Fruiting phenology is linked to rainfall variability in a tropical rain forest. *Biotropica,* 50(3): 396-404.

Durham, R.A., Mummey, D.L., Shreading, L. and Ramsey, P.W. 2017. Phenological patterns differ between exotic and native plants: Field observations from the Sapphire Mountains, Montana. *Natural Areas Journal,* 37(3): 361-381.

Dutta, G. 2013. Plant species diversity, community structure and regeneration of tree species in tropical forests of Assam, Ph.D. thesis, Tezpur University, Department of Environmental science.

Dutta, P.K., Dutta, B.K., Sundriyal, R.C. and Das, A.K. 2013. Diversity, representativeness and biotic pressure on plant species along alpine timberline of Western Arunachal Pradesh in the Eastern Himalaya, India. *Current Science,* 105(5): 701-708.

Dyderski, M.K. and Jagodziński, A.M. 2019. Context-dependence of urban forest vegetation invasion level and alien species' ecological success. *Forests*, 10(1): 26.

Early, R., Bradley, B.A., Dukes, J.S., Lawler, J.J., Olden, J.D., Blumenthal, D.M., Gonzalez, P., Grosholz, E.D., Ibañez, I., Miller, L.P. and Sorte, C.J. 2016. Global threats from invasive alien species in the twenty-first century and national response capacities. *Nature communications*, 7(1): 12485.

EEA. 2002. Europe's biodiversity - Biogeographical regions and seas. Biogeographical regions in Europe: The Alpine region - Mountains of Europe. European Environment Agency, Copenhagen.

Ehrenfeld, J.G. 2003. Effects of exotic plant invasions on soil nutrient cycling processes. *Ecosystems,* 6: 503-523..

Ehrenfeld, J.G. 2010. Ecosystem consequences of biological invasions. *Annual review of ecology, evolution, and systematics*, 41: 59-80.

Ehrenfeld, J.G., Kourtev, P. and Huang, W. 2001. Changes in soil functions following invasions of exotic understory plants in deciduous forests. *Ecological Applications*, 11(5): 1287-1300.

Ekeleme, F., Forcella, F., Archer, D.W., Akobundu, I.O. and Chikoye, D. 2005. Seedling emergence model for tropic *Ageratum* (*Ageratum conyzoides*). *Weed Science*, 53(1): 55-61.

El-Barougy, R.F., Dakhil, M.A., Halmy, M.W., Gray, S.M., Abdelaal, M., Khedr, A.H.A. and Bersier, L.F. 2021. Invasion risk assessment using trait-environment and species distribution modelling techniques in an arid protected area: Towards conservation prioritization. *Ecological Indicators*, 129: 107951.

Elgersma, K.J. and Ehrenfeld, J.G. 2011. Linear and non-linear impacts of a non-native plant invasion on soil microbial community structure and function. *Biological Invasions*, 13: 757-768.

Elliott, M. 2003. Biological pollutants and biological pollution—an increasing cause for concern. *Marine Pollution Bulletin*, 46(3): 275-280.

Elsen, P.R., Monahan, W.B. and Merenlender, A.M. 2018. Global patterns of protection of elevational gradients in mountain ranges. *Proceedings of the National Academy of Sciences*, 115(23): 6004-6009..

Elton, C.S. 1958. The ecology of invasions by animals and plants. Chicago, University of Chicago Press.

Enomoto, T. 1999. Naturalized weeds from foreign countries into Japan,1-14. In Biological invasions of ecosystem by pests and beneficial organisms, Yano. E, Matsuo, M, Shiyomi,M and andow, D.A. (eds.) National Institute of Agro-Enviromental Science, Tsukuba.

Enserink, M. 1999. Biological invaders sweep in. *Science*, 285: 1834–1836.

Essl, F., Dullinger, S., Rabitsch, W., Hulme, P.E., Hülber, K., Jarošík, V., Kleinbauer, I., Krausmann, F., Kühn, I., Nentwig, W. and Vilà, M. 2011. Socioeconomic legacy yields an invasion debt. *Proceedings of the National Academy of Sciences*, 108(1): 203-207.

Eviner, V.T. and Hawkes, C.V. 2012. The effects of plant-soil feedbacks on invasive plants: mechanisms and potential management options, 122-141. In Invasive plant ecology and management: Linking processes to practice, Monaco, T.A. and Sheley, R.L. (eds.), Wallingford UK: CABI, pp.122-141.

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

FAO. 2006. Global Forest Resources Assessment 2005. FAO Forestry Paper 147, Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO. 2021. Standard operating procedure for soil pH determination. Rome.

Fargione, J. and Tilman, D. 2005. Niche differences in phenology and rooting depth promote coexistence with a dominant C 4 bunchgrass. *Oecologia*, 143: 598-606.

Fei, S., Kong, N., Stringer, J. and Bowker, D. 2008. Invasion pattern of exotic plants in forest ecosystems, 59-70. In Invasive plants and forest ecosystems, Kohli. R.K., Jose, S., Singh, H.P and Batish, D.R (eds.), CRC Press, Boca Raton.

Fei, S., Phillips, J. and Shouse, M. 2014. Biogeomorphic impacts of invasive species. Annual Review of Ecology, *Evolution, and Systematics*, 45: 69-87.

Feng, J. and Zhu, Y. 2010. Alien invasive plants in China: risk assessment and spatial patterns. *Biodiversity and Conservation*, 19: 3489-3497.

Feoli, E. and Orlóci, L. 1991. The properties and interpretation of observations in vegetation study, 3-13. In Computer Assisted Vegetation Analysis, Feoli, E. and Orlóci, L. (eds.), Kluwer, Dordrecht, Netherland.

Fernandes, P., Antunes, C., Correia, O. and Máguas, C. 2015. Do climatic and habitat conditions affect the reproductive success of an invasive tree species? An assessment of the phenology of *Acacia longifolia* in Portugal. *Plant Ecology*, 216: 343-355.

Fields, C.B., Mortsch, L.D., Brklacich, M., Forbes, D.L., Kovacs, P., Patz, J.A., Running, S.W., Scott, M.J., Andrey, J., Cayan, D. and Demuth, M., 2007. North America. Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change.Cambridge, UK: Cambridge University Press.

Fink, K.A. and Wilson, S.D. 2011. *Bromus inermis* invasion of a native grassland: diversity and resource reduction. *Botany*, 89(3): 157-164.

Fitchett, J.M., Grab, S.W. and Thompson, D.I. 2015. Plant phenology and climate change: Progress in methodological approaches and application. *Progress in Physical Geography*, 39(4): 460-482.

Fitter, A.H. and Fitter, R.S.R. 2002. Rapid changes in flowering time in British plants. *Science,* 296(5573): 1689-1691.

Fletcher, R.A., Brooks, R.K., Lakoba, V.T., Sharma, G., Heminger, A.R., Dickinson, C.C. and Barney, J.N. 2019. Invasive plants negatively impact native, but not exotic, animals. *Global Change Biology*, 25(11): 3694-3705.

Florida Exotic Pest Plant Council. 2011. Florida EPPC's 2011 Invasive Plant Species List. http://www.fleppc.org/list/11list.html

Fogelström, E., Zacchello, G., Guasconi, D., Dahlgren, J.P. and Ehrlén, J. 2022. Spring and autumn phenology in an understory herb are uncorrelated and driven by different factors. *American Journal of Botany*, 109(2): 226-236.

Forrest, J. and Miller-Rushing, A.J., 2010. Toward a synthetic understanding of the role of phenology in ecology and evolution. *Philosophical Transactions of the Royal Society B: Biological Sciences,* 365(1555): 3101-3112.

Foster, R.B. 1985. Plant seasonality in the forests of Panama. Monographs in systematic botany from the Missouri Botanical Garden (USA).

Fox, G.A. 1990. Components of flowering time variation in a desert annual. *Evolution,* 44(6); 1404-1423.

Foy, C.D. 1984. Physiological effects of hydrogen, aluminum, and manganese toxicities in acid soil. *Soil Acidity and Liming,* 12: 57-97.

Francis J.K. 2000. Wildland Shrubs of the United States and its Territories: Thamnic Descriptions. General Technical Report - International Institute of Tropical Forestry, IITF-WB-1. http://www.fs.fed.us/global/iitf/wildland_shrubs.htm

Francis, J.K. 2004. Wildland Shrubs of the United States and its Territories: Thamnic Descriptions, Volume I. USDA Forest Service, International Institute of Tropocal Forestry, San Juan PR; Rocky Mountain Research Station, Fort Collins CO; General Technical Report.

Frankie, G.W., Baker, H.G. and Opler, P.A. 1974. Comparative phenological studies of trees in tropical wet and dry forests in the lowlands of Costa Rica. *The Journal of Ecology*, 62(3): 881-919.

Fridley, J.D. 2012. Extended leaf phenology and the autumn niche in deciduous forest invasions. *Nature*, 485(7398): 359-362.

FSI. 2021. Atlas Forest Types of India. Published by Forest Survey of India.

FSI. 2009. State of Forest Report. Published by Forest Survey of India. https://fsi.nic.in/forest-report-2009. (Accessed on 29.05.2024).

Fu, Y.H., Piao, S., Zhao, H., Jeong, S.J., Wang, X., Vitasse, Y., Ciais, P. and Janssens, I.A. 2014. Unexpected role of winter precipitation in determining heat requirement for spring vegetation green-up at northern middle and high latitudes. *Global Change Biology*, 20(12); 3743-3755.

Fui, L.H. 1992. Knowledge and use of forest product as traditional medicine: The case of the forest-dwelling communities,385-400. In Proceedings of the Conference on Medicinal Products from Tropical Rain Forest. K. Shaari, A. A. Adir, A. R. M. Ali (eds), Forest Research Institute of Malaysia, Kuala Lumpur.

Funk, J.L. and Vitousek, P.M. 2007. Resource-use efficiency and plant invasion in low-resource systems. *Nature*, 446(7139): 1079-1081.

Fürer-Haimendorf, C.V. 1962. The Apa Tanis and their Neighbours: a primitive civilization of the eastern Himalayas. RKP. +166pp.

Fusco, E.J., Balch, J.K., Mahood, A.L., Nagy, R.C., Syphard, A.D. and Bradley, B.A. 2022. The human–grass–fire cycle: how people and invasives co-occur to drive fire regimes. *Frontiers in Ecology and the Environment*, 20(2): 117-126.

Gaertner, J. 1791. De Fructibus et Seminibus Plantarum, 2(3): 456-457.

Gaertner, M., Biggs, R., Te Beest, M., Hui, C., Molofsky, J. and Richardson, D.M. 2014. Invasive plants as drivers of regime shifts: identifying high-priority invaders that alter feedback relationships. *Diversity and Distributions*, 20(7): 733-744.

Gaertner, M., Den Breeyen, A., Hui, C. and Richardson, D.M. 2009. Impacts of alien plant invasions on species richness in Mediterranean-type ecosystems: a meta-analysis. *Progress in Physical Geography*, 33(3): 319-338.

Gaertner, M., Holmes, P.M. and Richardson, D.M. 2012. Biological invasions, resilience and restoration, 265-280. In Restoration Ecology: The New Frontier, Andel, J,A. and Aronson, J. (eds.), Blackwell publishing Ltd.

Gallardo, B., Bacher, S., Bradley, B., Comín, F.A., Gallien, L., Jeschke, J.M., Sorte, C.J. and Vilà, M. 2019. InvasiBES: Understanding and managing the impacts of Invasive alien species on Biodiversity and Ecosystem Services. *NeoBiota*, 50: 109-122.

Gallien, L. and Carboni, M. 2017. The community ecology of invasive species: where are we and what's next?. *Ecography*, 40(2): 335-352.

García-Robledo, C.A. and Murcia, C. 2005. Comparative habitat susceptibility to invasion by Chinese ash (*Fraxinus chinensis*: Oleaceae) in a tropical Andean landscape. *Biological Invasio*ns, 7: 405-415.

Gautier, L. 1992. Taxonomy and distribution of a tropical weed: Chromolaena odorata (L.) R. King & H. Robinson. *Candollea*, 47(2); 645-662.

Gebrekiros, M.G. and Tessema, Z.K. 2018. Effect of *Senna obtusifolia* (L.) invasion on herbaceous vegetation and soil properties of rangelands in the western Tigray, northern Ethiopia. *Ecological Processes*, **7**: 1-12.

Gentili, R., Ferrè, C., Cardarelli, E., Montagnani, C., Bogliani, G., Citterio, S. and Comolli, R. 2019. Comparing negative impacts of Prunus serotina, Quercus rubra and Robinia pseudoacacia on native forest ecosystems. *Forests*, 10(10): 842.

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

Ghayal, N.G., Padhye, A. and Dhumal, K. 2010. Larvicidal activity of invasive weeds Cassia uniflora and Synedrella nodiflora. *International Journal of Pharma and Bio Science*s, 1: 1 10.

Ghermandi, A., Galil, B., Gowdy, J. and Nunes, P.A. 2015. Jellyfish outbreak impacts on recreation in the Mediterranean Sea: welfare estimates from a socioeconomic pilot survey in Israel. *Ecosystem Services*, 11: 140-147.

Ghosh, C. 2006. Biology of tea garden weeds in Darjeeling district of West Bengal (India), Ph.D. thesis, University of North Bengal.

Ghosh, C., Das, P., Poddar, A. and Das, A.P. 2022. Phenology for the weed flora of the tea gardens in terai and hills of Darjeeling district of West Bengal, India. *Indian Forester*, 148(8): 845-853.

Gibbons, S.M., Lekberg, Y., Mummey, D.L., Sangwan, N., Ramsey, P.W. and Gilbert, J.A. 2017. Invasive plants rapidly reshape soil properties in a grassland ecosystem. *MSystem*s, 2(2): 10-1128.

Gilbert, B. and Levine, J.M. 2013. Plant invasions and extinction debts. *Proceedings of the National Academy of Science*s, 110(5): 1744-1749.

Giorgis, M.A., Cingolani, A.M., Tecco, P.A., Cabido, M., Poca, M. and Von Wehrden, H. 2016. Testing alien plant distribution and habitat invasibility in mountain ecosystems: growth form matters. *Biological Invasions,* 18(7): 2017-2028.

Gioria, M., Pyšek, P. and Osborne, B.A. 2018. Timing is everything: does early and late germination favor invasions by herbaceous alien plants?. *Journal of Plant Ecology,* 11(1): 4-16.

Girija, T., John, C.L. and Abraham, C.T. 2015. Seasonality of emergence of selected annual weeds in coconut garden. *Indian Journal of Weed Science*, 47(1): 55-58.

GISD. 2010. Global Invasive Species Database. Available at http//www.issg.org.database (Accesed on 02.09.2024).

GISD. 2024. Global Invasive Species Database Species profile: Bidens pilosa. http://www.iucngisd.org/gisd/species.php?sc=1431. (Accessed on 02.09.2024).

GISIN. 2008. Global Invasive Species Information Network. http://www.gisinetwork.org/. (Accessed on 29.05.2024).

GISP. 2000. Global strategy on invasive alien species. In: Global Invasive Species Programme Workshop, September, Cape Town, South Africa.

Godoy, O., Richardson, D.M., Valladares, F. and Castro-Díez, P. 2009. Flowering phenology of invasive alien plant species compared with native species in three Mediterranean-type ecosystems. *Annals of botany*, 103(3): 485-494.

Goedickemeier, I., Wildi, O. and Kienast, F. 1997. Sampling for vegetation survey: some properties of a GIS-based stratification compared to other statistical sampling methods. *Coenoses,*12(1): 43-50.

Goodall, J.M. and Erasmus, D.J. 1996. Review of the status and integrated control of the invasive alien weed, *Chromolaena odorata*, in South Africa. *Agriculture, Ecosystems & Environment*, 56(3): 151-164.

Gonzalez-Moreno, P., Pino, J., Gassó, N. and Vila, M. 2013. Landscape context modulates alien plant invasion in Mediterranean forest edges. *Biological invasions*, 15(3): 547-557.

Gosper, C.R. 2004. Fruit characteristics of invasive bitou bush, *Chrysanthemoides monilifera* (Asteraceae), and a comparison with co-occurring native plant species. *Australian Journal of Botany*, 52(2): 223-230.

Gotelli, N.J. and Ellison, A.M. 2004. A primer of ecological statistics Sunderland: Sinauer Associates, 1:+640pp.

Groves, R. 1991. A short history of biological invasions of Australia, 59-63. In Biography of Mediterranean Invasions. Cambridge University Press, New York, pp.59-63.

Guckland, A., Jacob, M., Flessa, H., Thomas, F.M. and Leuschner, C. 2009. Acidity, nutrient stocks, and organic-matter content in soils of a temperate deciduous forest with different abundance of European beech (*Fagus sylvatica* L.). *Journal of Plant Nutrition and Soil Scienc*e, 172(4): 500-511.

Gudla, S.L., Devarakonda, N., Ray, S. and Varikuppala, M. 2023. Evaluating the Primary Macronutrients and their Correlations with pH, Electrical conductivity, Organic Carbon and Soil Nutrient index in the Arid and Semi-Arid Climatic Zones of Anantapur District, Andhra Pradesh, India. *International Journal of Plant & Soil Science*, 35(20): 490-497.

Gugger, S., Kesselring, H., Stöcklin, J. and Hamann, E. 2015. Lower plasticity exhibited by high-versus mid-elevation species in their phenological responses to manipulated temperature and drought. *Annals of Botany*, 116(6): 953-962.

Guirado, M., Pino, J. and Rodà, F. 2006. Understorey plant species richness and composition in metropolitan forest archipelagos: effects of forest size, adjacent land use and distance to the edge. *Global Ecology and Biogeography*, 15(1): 50-62.

Gupta, U., Verma, P. and Solanki, H. 2014. Impact of high boron concentration on plants. *Biolife*, 2: 610-614.

Gurevitch, J. and Padilla, D.K. 2004. Are invasive species a major cause of extinctions?. *Trends In Ecology & Evolution*, 19(9); 470-474.

Ha, K.V., Marschner, P., Bünemann, E.K. and Smernik, R.J. 2007. Chemical changes and phosphorus release during decomposition of pea residues in soil. *Soil Biology and Biochemistry*, 39(10): 2696-2699.

Hajra, P.K., Verma, D.M. and Giri., G.S. 1996. Materials for the Flora of Arunachal Pradesh, Vol. 1. Botanical Survey of India, Kolkata, +693pp.

Hajra, P.K., Verma, D.M. and Giri., G.S. 2008. Materials for the Flora of Arunachal Pradesh, Vol. 2. Botanical Survey of India, Kolkata, +486pp.

Hajra, P.K., Verma, D.M. and Giri., G.S. 2009. Materials for the Flora of Arunachal Pradesh, Vol. 3. Botanical Survey of India, Kolkata, +313pp.

Hamann, A. 2004. Flowering and fruiting phenology of a Philippine submontane rain forest: climatic factors as proximate and ultimate causes. *Journal of Ecology*, 92(1): 24-31.

Hamer, K., Gudenschwager, I. and Pichler, T. 2020. Manganese (Mn) concentrations and the Mn-Fe relationship in shallow groundwater: implications for groundwater monitoring. *Soil Systems*, 4(3): 49.

Hammer, Ø., Harper, D.A.T and Ryan, P.D. 2001. PAST: Paleontological Statistics Software Package for Education and Data Analysis, *Palaeontologia Electronica*,4(1): 9.

Hannan-Jones, M.A. and Playford, J. 2002. The biology of Australian weeds 40. Bryophyllum Salisb. species. *Plant Protection Quarterly*, 17(2): 42-57.

Hansen, M.H.; Hurwitz, W.N. and Madow, W.G. 1953.Sample Survey Methods and Theory.Volumes I and II, John Wiley & Sons.New York

Hanway, J.J. and Heidel, H. 1952. Soil analysis methods as used in the Iowa state college soil testing laboratory.

Haque, A., Zahan, R., Nahar, L., Mosaddik, A. and Haque, E. 2012. Anti-inflammatory and insecticidal activities of *Synedrella nodiflora*. *Mol Clin Pharmacol*, 2012: 60-67.

Hatfield, J.L. and Prueger, J.H., 2015. Temperature extremes: Effect on plant growth and development. *Weather and climate extremes*, 10, pp.4-10.

Harper, D.A.T. 1999. Numerical Palaeobiology. Computer-Based Modelling and Analysis of Fossils and their Distributions, John Wiley & Sons, +468pp.

Harrington, R.A., Brown, B.J. and Reich, P.B. 1989. Ecophysiology of exotic and native shrubs in southern Wisconsin: I. Relationship of leaf characteristics, resource availability, and phenology to seasonal patterns of carbon gain. *Oecologia*, 80: 356-367.

Harris, P.J.C. 1986. Dormancy of *Urena lobata* L. seeds. I. Development of sulphuric acid scarification techniques. *Ghana Journal of Agriculatural Science*, 14-19: 79-84.

Hartmann, D.L., Tank, A.M.K., Rusticucci, M., Alexander, L.V., Brönnimann, S., Charabi, Y.A.R., Dentener, F.J., Dlugokencky, E.J., Easterling, D.R., Kaplan, A. and Soden, B.J. 2013. Observations: atmosphere and surface, 159-254. In Climate change 2013 the physical science basis: Working group I contribution to the fifth assessment report of the intergovernmental panel on climate change, Cambridge University Press..

Hayes, S.J. and Holzmueller, E.J. 2012. Relationship between invasive plant species and forest fauna in eastern North America. *Forests*, 3(3): 840-852.

Hazlett, P.W., Gordon, A.M., Sibley, P.K. and Buttle, J.M. 2005. Stand carbon stocks and soil carbon and nitrogen storage for riparian and upland forests of boreal lakes in northeastern Ontario. *Forest Ecology and Management*, 219(1): 56-68.

Hegarty, E.E., 1990. Leaf life-span and leafing phenology of lianes and associated trees during a rainforest succession. *The Journal of Ecology*, 78: 300-312.

Hegazy, A.K., Alatar, A.A., Lovett-Doust, J. and El-Adawy, H.A. 2012. Spatial and temporal plant phenological niche differentiation in the Wadi Degla desert ecosystem (Egypt). *Acta Botanica Croatica*, 71(2): 261-277.

Heideman, P.D. 1989. Temporal and spatial variation in the phenology of flowering and fruiting in a tropical rainforest. The Journal of Ecology, 77: 1059-1079.

Hejda, M. and Pyšek, P. 2006. What is the impact of Impatiens glandulifera on species diversity of invaded riparian vegetation? *Biological Conservation*, 132(2): 143-152.

Hejda, M., Pyšek, P. and Jarošík, V. 2009. Impact of invasive plants on the species richness, diversity and composition of invaded communities. *Journal of Ecology*, 97(3): 393-403.

Henderson, L. 2001. Alien weeds and invasive plants. A complete guide to declared weeds and invaders in South Africa. Handbook No. 12. Pretoria, South Africa: ARC-PPRI.

Henderson, S., Dawson, T.P. and Whittaker, R.J. 2006. Progress in invasive plants research. *Progress in Physical Geography*, 30(1); .25-46.

Hernández, L., Martínez-Fernández, J., Cañellas, I. and de la Cueva, A.V. 2014. Assessing spatio-temporal rates, patterns and determinants of biological invasions in forest ecosystems. The case of Acacia species in NW Spain. *Forest Ecology and Management*, 329: 206-213.

Hierro, J.L., Maron, J.L. and Callaway, R.M. 2005. A biogeographical approach to plant invasions: the importance of studying exotics in their introduced and native range. *Journal of Ecology*, 93(1): 5-15.

Hillel, D. 2003. Introduction to environmental soil physics. *Elsevier*,+485pp.

Hollister, R.D., Webber, P.J. and Bay, C. 2005. Plant response to temperature in northern Alaska: implications for predicting vegetation change. *Ecology*, 86(6); 1562-1570.

Holm, L., Doll, J., Holm, E., Pancho, J.V. and Herberger, J.P. 1997. World weeds: natural histories and distribution. John Wiley & Sons.

Holm, L., Pancho, J.V., Herberger, J.P. and Plucknett, D.L. 1991. A geographical atlas of world weeds, Malabar, Florida, USA: Krieger Publishing Company, +391pp.

Holm, L.G., Plucknett, D.L., Pancho, J.V. and Herberger, J.P. 1977. The world's worst weeds. Distribution and biology. Honolulu, HI: University Press of Hawaii.

Holmes, P.M., Esler, K.J., Richardson, D.M. and Witkowski, E.T.F. 2008. Guidelines for improved management of riparian zones invaded by alien plants in South Africa. *South African Journal of Botany*, 74(3): 538-552.

Holmes, W.C. 1975. A Revision of Mikania Scandens and Relatives (compositae), Ph.D. thesis, Mississippi State University.

Honu, Y.A. and Dang, Q.L. 2000. Responses of tree seedlings to the removal of Chromolaena odorata Linn. in a degraded forest in Ghana. *Forest Ecology and Management*, 137(1-3); 5-82.

Hooker, J.D. 1882. The Flora of British India, vol. III. Reeve and Co., Ashford, UK.

Horbach, S., Rauschkolb, R. and Römermann, C. 2023. Flowering and leaf phenology are more variable and stronger associated to functional traits in herbaceous compared to tree species. *Flora*, 300: 152218.

Hornoy, B., Tarayre, M., Hervé, M., Gigord, L. and Atlan, A. 2011. Invasive plants and enemy release: evolution of trait means and trait correlations in Ulex europaeus. *PLoS One*, 6(10): e26275.

Houlahan, J.E. and Findlay, C.S. 2004. Effect of invasive plant species on temperate wetland plant diversity. *Conservation Biology*, 18(4): 1132-1138.

Hovick, S.M., Peterson, C.J. and Carson, W.P. 2012. Predicting invasiveness and range size in wetland plants using biological traits: a multivariate experimental approach. *Journal of Ecology*, 100(6): 1373-1382.

Howard R.A. 1989. Flora of the Lesser Antilles, Leeward and Windward Islands. Jamaica Plain, MA: Arnold Arboretume, Harvard University Press,+658pp.

Hua, Z. 2012. Biogeographical Divergence of the Flora of Yunnan, Southwestern China Initiated by the Uplift of Himalaya and Extrusion of Indochina Block. *Plos One* 7(9): e45601.

Huang, Z.L., Cao, H.L., Liang, X.D., Ye, W.H., Feng, H.L. and Cai, C.X. 2000. The growth and damaging effect of M. *micranth*a H.B.K in different habitats. *Journal of Tropical and Subtropical Botany*, 8(2): 131-138.

Huebner, C.D. 2021. Patterns of invasive plant abundance in disturbed versus undisturbed forests within three land types over 16 years. *Diversity and Distributions*, 27(1): 130-143.

Hughes, C.E. and Atchison, G.W. 2015. The ubiquity of alpine plant radiations: from the Andes to the Hengduan Mountains. *New Phytologist*, 207(2): 275-282.

Hulme, P.E. 2007. Biological invasions in Europe: drivers, pressures, states, impacts and responses. *Biodiversity Under Threat*, 25: 56-80.

Hulme, P.E. 2011. Contrasting impacts of climate-driven flowering phenology on changes in alien and native plant species distributions. *New Phytologist*, 189(1): 272-281.

Hulme, P.E. 2012. Weed risk assessment: a way forward or a waste of time?. *Journal of Applied Ecology,* 49(1): 10-19.

Hulme, P.E. 2014. Invasive species challenge the global response to emerging diseases. *Trends in parasitology*, 30(6): 267-270.

Hulme, P.E. 2015. Invasion pathways at a crossroad: policy and research challenges for managing alien species introductions. *Journal of Applied Ecology,* 52: 1418-1424.

Hulme, P.E. 2017. Climate change and biological invasions: evidence, expectations, and response options. *Biological Reviews*, 92(3): 1297-1313.

Idu, M. and Onyibe, H.I. 2007. Medicinal plants of Edo State, Nigeria. *Research Journal of Medicinal Plant,* 2: 32–41.

Ikhajiagbe, B. 2016. Possible adaptive growth responses of *Chromolaena odorata* during heavy metal remediation. *Ife Journal of Science,* 18(2): 403-411.

Imaizumi, T. and Kay, S.A. 2006. Photoperiodic control of flowering: not only by coincidence. *Trends In Plant Science,* 11(11): 550-558.

Inouye, D.W., Saavedra, F. and Lee-Yang, W. 2003. Environmental influences on the phenology and abundance of flowering by *Androsace septentrionalis* (Primulaceae). *American Journal of Botany,* 90(6): 905-910.

Inuthai, J. 2023. Flowering and fruiting phenology of herbs, climbers, shrubs, and trees in the deciduous dipterocarp forest of Northern Thailand. *Journal of Ecology and Environment,* 47(3): 134-145.

IOP. 2019. Inception of administration. Government of Arunachal Pradesh. https://web.archive.org/web/20090410004725/http://arunachalpradesh.nic.in/rural/html/glance.htm. (Accessed on 19 February 2024).

IPCC. 2014. Climate change 2014: impacts, adaptation, and vulnerability. Working Group II contribution to the IPCC Fifth June 2019 Assessment Report. Cambridge, United Kingdom: Cambridge University Press.

IPCC. 2007. The Physical Science basis, 113-119. Contribution of working group i to the fourth assessment report of the intergovernmental panel on climate change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, +996pp.

Iqbal, M., Mahmooduzzafar, Nighat, F. and Khan, P.R. 2010. Photosynthetic, metabolic and growth responses of *Triumfetta rhomboidea* to coal-smoke pollution at different stages of plant ontogeny. Journal of Plant Interactions, 5(1): 11-19.

Iranmanesh, M. and Sadeghi, H. 2019. The effect of soil organic matter, electrical conductivity and acidity on the soil's carbon sequestration ability via two species of Tamarisk (*Tamarix* Spp.). *Environmental Progress & Sustainable Energy*, 38(6): 13230.

Islam, M.T. and Uddin, M.A. 2017. A revision on *Urena lobata* L. *International Journal of Medicine*, 5(1): 126-131.

Jain, S.K., Borthakur, S.K. and D'Arcy, W.G. 1986. Solanaceae in Indian tradition, folklore, and medicine, 577-583. In Solanaceae Biology and Systematics, D'Arcy, W.G. (eds.), Columbia University Press, New York.

James, F.C. and McCulloch, C.E. 1990. Multivariate analysis in ecology and systematics: panacea or Pandora's box? *Annual Review of Ecology and Systematics*, 21: 129-166.

Jandová, K., Klinerová, T., Müllerová, J., Pyšek, P., Pergl, J., Cajthaml, T. and Dostál, P. 2014. Long-term impact of *Heracleum mantegazzianum* invasion on soil chemical and biological characteristics. *Soil Biology and Biochemistry*, 68: 270-278.

Jansky, L., Ives, J.D., Furuyashiki, K. and Watanabe, T. 2002. Global mountain research for sustainable development. *Global Environmental Change*, 12(3): 231-239.

Janzekovic, F. and Novak, T. 2012. PCA–a powerful method for analyze ecological niches, 127-142. In Principal Component Analysis–Multidisciplinary Applications, Parinya, S. (eds.), InTech Publishing.

Jaroenkit, T., Buamasung, B., Ussahatanonta, S. and Manochai, P. 2012, December. Effects of temperature, relative humidity and rainfall on leaf flushing, flowering and yield of longan grown in Chiang Mai Province, Thailand, 169-175. In IV International Symposium on Lychee, Longan and Other Sapindaceae Fruits 1029.

Jeddi, K., Cortina, J. and Chaieb, M. 2009. *Acacia salicina, Pinus halepensis* and *Eucalyptus occidentalis* improve soil surface conditions in arid southern Tunisia. *Journal of Arid Environments*, 73(11): 1005-1013.

Jelbert, K.J. 2018. The Comparative Demography of Invasive Plants. University of Exeter United Kingdom.

Jeschke, J.M., Bacher, S., Blackburn, T.M., Dick, J.T., Essl, F., Evans, T., Gaertner, M., Hulme, P.E., Kühn, I., Mrugała, A. and Pergl, J. 2014. Defining the impact of non-native species. *Conservation Biology*, 28(5): 1188-1194.

John, M.K., Chuah, H.H. and Neufeld, J.H. 1975. Application of improved azomethine-H method to the determination of boron in soils and plants. *Analytical Letters*, 8(8): 559-568.

Johnson, A.B. and Smith, C.D. 2017. Organic Carbon and Soil Electrical Conductivity: Relationships and Implications for Soil Health. *Soil and Environmental Science*, 24(2): 189-205.

Johnston, F.M. and Pickering, C.M. 2001. Alien plants in the Australian Alps. *Mountain Research and Development*, 21(3): 284-291.

Jolly, W.M. and Running, S.W. 2004. Effects of precipitation and soil water potential on drought deciduous phenology in the Kalahari. *Global Change Biology*, 10(3): 303-308.

Jones, B.A. 2019. Tree shade, temperature, and human health: evidence from invasive species-induced deforestation. *Ecological Economics*, 156: 12-23.

Jose, S., Kohli, R.K., Singh, H.P., Batish, D.R. and Pieterson, E.C. 2009. Invasive plants: a threat to the integrity and sustainability of forest ecosystems, 3-10. In Invasive Plants and Forest Ecosystems, Kohli, R.K., Jose, S., Singh, H.P and batish, D.R. (eds.), CRC Press.

Joshi, C.M. 2001. Invasive banmara, *Chromolaena odorata*: spatial detection and prediction. MSc Thesis, International Institute for Aerospace Survey and Earth Sciences, The Netherlands, 53 pp.

Jourgholami, M., Feghhi, J., Picchio, R., Tavankar, F. and Venanzi, R. 2021. Efficiency of leaf litter mulch in the restoration of soil physiochemical properties and enzyme activities in temporary skid roads in mixed high forests. *Catena*, 198: 105012.

Kader, M.A., Sleutel, S., Begum, S.A., Moslehuddin, A.Z.M. and De Neve, S. 2013. Nitrogen mineralization in sub-tropical paddy soils in relation to soil mineralogy, management, pH, carbon, nitrogen and iron contents. *European Journal of Soil Science,* 64(1): 47-57.

Kala, C.P. 2005. Ethnomedicinal botany of the Apatani in the Eastern Himalayan region of India. *Journal of Ethnobiology and Ethnomedicine*, 1: 1-8.

Kalwij, J.M., Robertson, M.P. and van Rensburg, B.J. 2008. Human activity facilitates altitudinal expansion of exotic plants along a road in montane grassland, South Africa. *Applied Vegetation Science,* 11(4): 491-498.

Kanjilal, U.N., Kanjilal, P.C. and Das, A. 1934. Flora of Assam, Vol.1 part 1. Assam Govt. Press, Shillong.

Kanjilal, U.N., Kanjilal, P.C. and Das, A. 1936. Flora of Assam, Vol.1 part 2. Assam Govt. Press, Shillong.

Kanjilal, U.N., Kanjilal, P.C. and Das, A. 1938. Flora of Assam, Vol. 2. Assam Govt. Press, Shillong.

Kanjilal, U.N., Kanjilal, P.C. and Das, A. 1939. Flora of Assam, Vol. 3. Assam Govt. Press, Shillong.

Kanjilal, U.N., Kanjilal, P.C. and Das, A. 1940. Flora of Assam, Vol. 4. Assam Govt. Press, Shillong.

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

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

Kanwal, K.S. and Lodhi, M.S. 2018. Climate change impact on plant biodiversity of Arunachal Himalaya: a review. *Bulletin of Arunachal Forest Research*, 33(2): 15-26.

Karami, M., Afyuni, M., Khoshgoftarmanesh, A.H., Papritz, A. and Schulin, R. 2009. Grain zinc, iron, and copper concentrations of wheat grown in central Iran and their relationships with soil and climate variables. *Journal of Agricultural and Food Chemistry*, 57(22): 10876-10882.

Kato-Noguchi, H. and Kurniadie, D. 2024. The invasive mechanisms of the noxious alien plant species *Bidens pilosa*. *Plants*, 13(3): 356.

Kaufman, S.R. and Kaufman, W. 2023. Invasive plants: guide to identification and the impacts and control of common North American species. Rowman & Littlefield, +535pp.

Kaur, A., Kaur, S., Singh, H.P., Datta, A., Chauhan, B.S., Ullah, H., Kohli, R.K. and Batish, D.R. 2023. Ecology, biology, environmental impacts, and management of an agro-environmental weed *Ageratum conyzoides*. *Plants*, 12(12): 2329.

Kaur, S., Batish, D.R., Kohli, R.K. and Singh, H.P. 2012. *Ageratum conyzoides*: an alien invasive weed in India, 57-76. In Invasive alien plants: an ecological appraisal for the Indian subcontinent, J.R., Singh, J.S., Singh, S.P., Tripathi, R.S. and Kohli, R.K. (eds.), Wallingford UK: CABI.

Keller, R.P., Geist, J., Jeschke, J.M. and Kühn, I. 2011. Invasive species in Europe: ecology, status, and policy. *Environmental Sciences Europe*, 23: 1-17.

Kelly, A.E. and Goulden, M.L. 2008. Rapid shifts in plant distribution with recent climate change. *Proceedings of the national academy of sciences*, 105(33): 11823-11826.

Kemp, D.J. 1998. Oviposition behaviour of post-diapause *Hypolimnas bolina* (L.) (Lepidoptera: Nymphalidae) in tropical Australia. *Australian Journal of Zoology*, 46(5): 451-459.

Kendre, N. and Wakte, P. 2023. Comparative extraction and quantification of Scutellarein from leaves of *Triumfetta rhomboidea* using RP-HPLC. *Indian Journal of Pharmaceutical Education and Research,* 57: 101-106.

Kenis, M., Auger-Rozenberg, M.A., Roques, A., Timms, L., Péré, C., Cock, M.J.W., Settele, J., Augustin, S. and Lopez-Vaamonde, C. 2009. Ecological effects of invasive alien insects. *Biological Invasions*,11: 21-45.

Kennedy, T.A., Naeem, S., Howe, K.M., Knops, J.M., Tilman, D. and Reich, P. 2002. Biodiversity as a barrier to ecological invasion. *Nature*, 417(6889): 636-638.

Kerkeb, L. and Connolly, E.L. 2006. Iron transport and metabolism in plants, 119-140. In Genetic Engineering: Principles and Methods, Setlow, J.K. (eds.), Springer New York, NY.

Kesari, G.K. 2010. Geology and mineral resources of Arunachal Pradesh. Geological Survey of India, Government of India.

Khan, A.M., Qureshi, R., Arshad, M. and Mirza, S.N. 2018. Climatic and flowering phenological relationships of western Himalayan flora of Muzaffarabad district, Azad Jammu and Kashmir, Pakistan. *Pakistan Journal of Botany*, 50(3): 1093-1112.

Khan, A.M., Qureshi, R., Qaseem, M.F., Munir, M., Ilyas, M. and Saqib, Z. 2015. Floristic checklist of district kotli, Azad Jammu & Kashmir. *Pakistan Journal of Botany*, 47(5): 1957-1968.

Khan, S.M., Page, S., Ahmad,H, Shaheen, H and Harper, D.M. 2012. Vegetation dynamics in the Western Himalayas, diversity indices and climate change. *Science, Technology and Development*, 31(3): 232-243.

Khatri, K., Negi, B., Bargali, K. and Bargali, S.S. 2022. Trait variability in co-occurring invasive and native plant species in road side population of Kumaun Himalaya. *Brazilian Journal of Botany*, 45(3): 1099-1110.

Khodorova, N.V. and Boitel-Conti, M., 2013. The role of temperature in the growth and flowering of geophytes. *Plants*, 2(4), pp.699-711.

Khound, A.K. and Barua, P.C., 2016. Influence of weather parameter on flowering and fruiting spike of medicinal plant Piper longum L. in Assam, India. *Adv. Res. J. Crop Improv.*, 7(2), pp. 8–12.

Khuroo, A.A., Ahmad, R., Hamid, M., Ruquia, G., Malik, A.H. and Rashid, I. 2021. Alien flora of Indian Himalayan Region: taxonomic, biogeographical and ecological assessment. *New Vistas in Indian Flora*, 2: 497-518.

Khuroo, A.A., Reshi, Z.A., Rashid, I. and Dar, G.H. 2011. Towards an integrated research framework and policy agenda on biological invasions in the developing world: a case-study of India. *Environmental Research*, 111(7): 999-1006.

Kibblewhite, M.G., Ritz, K. and Swift, M.J. 2008. Soil health in agricultural systems. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 363(1492): 685-701.

Kier, G., Mutke, J., Dinerstein, E., Ricketts, T.H., Küper, W., Kreft, H. and Barthlott, W. 2005. Global patterns of plant diversity and floristic knowledge. *Journal of Biogeography*, 32(7): 1107-1116.

Kikim, A. and Yadava, P.S. 2001. Phenology of tree species in subtropical forests of Manipur in north eastern India. *Tropical Ecology*, 42(2): 269-276.

King, R.M. and Robinson, H. 1970. Studies in the Eupatorieae (Compositae). XXIV. the genus *Chromolaena*. *Phytologia*, 20: 196-209.

Kirkham, M.B. 2005. Principles of soil and plant water relations. Elsevier. (1st Edn.) DOI: https://doi.org/10.1016/B978-0-12-409753.X5000-2

Kissmann, G. and Groth, D. 1993. Plantas Infestantes e Nocivas. Basf Brasileira, Sao Paulo, Brazil.

Klironomos, J.N. 2002. Feedback with soil biota contributes to plant rarity and invasiveness in communities. *Nature*, 417(6884): 67-70.

Knapp, S. and Kühn, I. 2012. Origin matters: widely distributed native and non-native species benefit from different functional traits. *Ecology Letters*, 15(7): 696-703.

Knapp, S. 2024a. *Solanum torvum* Sw. In Solanacea Source. https://solanaceaesource.myspecies.info/taxonomy/term/110004/descriptions. (Accessed on 29.05. 2024).

Knapp, S. 2024b. *Solanum viarum* Dunal. In Solanacea Source. https://solanaceaesource.myspecies.info/taxonomy/term/110343/descriptions. (Accessed on 29.05. 2024).

Koch F. 2004. Kowarik, i. 2003. Biologische Invasionen: Neophyten und Neozoen in Mitteleuropa.-Stuttgart; Hohenheim.

Kohli, R.K., Batish, D.R., Singh, H.P. and Dogra, K.S. 2006. Status, invasiveness and environmental threats of three tropical American invasive weeds (*Parthenium* *hysterophorus* L., *Ageratum conyzoides* L., *Lantana camara* L.) in India. *Biological Invasions*, 8: 1501-1510.

Kohli, R.K., Dogra, K.S., Batish, D.R. and Singh, H.P. 2004. Impact of invasive plants on the structure and composition of natural vegetation of north western Indian Himalayas. *Weed Technology*, 18(sp1): 1296-1300.

Koné, A.W., Edoukou, E.F., Orendo-Smith, R. and Tondoh, J.E. 2012. Earthworms in *Chromolaena odorata* (L.) King and Robinson (Asteraceae) fallows along a chronosequence: changes in community structure and identification of persistent and indicator species. *Pedobiologia*, 55(4): 193-201.

Koné, A.W., Kassi, S.P.A., Koffi, B.Y., Masse, D., Maïga, A.A., Tondoh, J.E., Kisaka, O.M. and Touré, G.P.T. 2021. *Chromolaena odorata* (L.) K&R (Asteraceae) invasion effects on soil microbial biomass and activities in a forest-savanna mosaic. *Catena*, 207: 105619.

Kopp, C.W., Neto-Bradley, B.M., Lipsen, L.P., Sandhar, J. and Smith, S. 2020. Herbarium records indicate variation in bloom-time sensitivity to temperature across a geographically diverse region. *International Journal of Biometeorology*, 64: 873-880.

Koptur, S., Haber, W.A., Frankie, G.W. and Baker, H.G. 1988. Phenological studies of shrub and treelet species in tropical cloud forests of Costa Rica. *Journal of Tropical Ecology*, 4(4): 323-346.

Körner, C. and Spehn, E.M. 2019. Mountain Biodiversity: A Global Assessment (Vol. 7). Routledge.

Körner, C. 2000. The Alpine life zone under global change. Gayana. *Botánica*, 57(1): 1-17.

Korner, C. 2003. Alpine Plant Life: Functional Plant Ecology of High Mountain Ecosystems. Springer, Berlin.

Körner, C. 2004. Mountain biodiversity, its causes and function. *AMBIO: A Journal of the Human Environment*, 33(sp13): 11-17.

Körner, C. 2006. Plant CO2 responses: an issue of definition, time and resource supply. *New phytologist*, 172(3): 393-411.

Körner, C. 2007. The use of ‘altitude’in ecological research. *Trends In Ecology & Evolutio*n, 22(11): 569-574.

Kosaka, Y., Saikia, B., Mingki, T., Tag, H., Riba, T. and Ando, K. 2010. Roadside distribution patterns of invasive alien plants along an altitudinal gradient in Arunachal Himalaya, India. *Mountain Research and Development*, 30(3): 252-258.

Kothari, C.R. 2004. Research methodology: Methods and techniques. New Delhi: New Age International al (P) Limited Publishers.

Kourtev, P.S., Ehrenfeld, J.G. and Häggblom, M. 2002. Exotic plant species alter the microbial community structure and function in the soil. *Ecology*, 83(11): 3152-3166.

Kourtev, P.S., Ehrenfeld, J.G. and Häggblom, M. 2003. Experimental analysis of the effect of exotic and native plant species on the structure and function of soil microbial communities. *Soil Biology and Biochemistry*, 35(7): 895-905.

Koutika, L.S. and Rainey, H.J. 2010. *Chromolaena odorata* in different ecosystems: weed or fallow plant?. *Applied Ecology and Environmental Research*, 8(2): 131-142.

Koutika, L.S., Hauser, S., Meuteum Kamga, J.G. and Yerima, B. 2005. Comparative study of soil properties under *Chromolaena odorata*, *Pueraria phadeoloides* and *Calliandra calothyrsus*. *Plant and Soil*, 266: 315-323.

Koutika, L.S., Ndango, R. and Hauser, S. 2004. Nutrient concentrations and NH4+-N mineralization under different soil types and fallow forms in southern Cameroon. *Journal of Plant Nutrition and Soil Science*, 167(5): 591-595.

Kriticos, D.J., Yonow, T. and McFadyen, R.E. 2005. The potential distribution of *Chromolaena odorata* (Siam weed) in relation to climate. *Weed Research*, 45(4): 246-254.

Kueffer, C. and Kull, C.A. 2017. Non-native species and the aesthetics of nature, 311-324. In Impact of biological invasions on ecosystem services, Vila, M. and Hulme, P.E. (eds.), Springer Cham.

Kueffer, C., McDougall, K., Alexander, J., Daehler, C., Edwards, P., Haider, S., Milbau, A., Parks, C., Pauchard, A., Reshi, Z.A. and Rew, L.J. 2013. Plant invasions into mountain protected areas: assessment, prevention and control at multiple spatial scales, 89-113. In Plant invasions in protected areas: patterns, problems and challenges, Foxcroft, L. C., Pyšek, P., Richardson, D.M. and Genovesi, P. (eds.), Springer Dordrecht.

Kuete, V. 2017. Medicinal spices and vegetables from Africa: therapeutic potential against metabolic, inflammatory, infectious and systemic diseases. Academic Press.

Kumar, A. and Solanki, G.S. 2008. Population status and conservation of capped langurs (*Trachypithecus pileatus*) in and around Pakke Wildlife Sanctuary, Arunachal Pradesh, India. *Primate Conservation*, 23(1): 97-105.

Kumar, A., Sachan, S., Ghoshal, P. and Bharati, A. 2023. Identification key and check list of taxa of family Asteraceae of Jharkhand, India. *E-Planet*, 21(2): 150-171.

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

Kumar, S. and Singh, P. 1999. A study on pheasants and distribution in Arunachal Pradesh, Eastern Himalayas, India. Wildlife Institute of India, Dehradun.

Kumari, K., Nazir, G., Singh, A. and Kumar, P. 2017. Studies on boron fractions with different physico-chemical properties of cultivated soils of Himachal Pradesh, India. *International Journal of Current Microbiology and Applied Sciences*, 6(6): 1547-55.

Kumbhani, N.R. 2021. Comparative study of changes in phenology and phytochemistry of some selected fruit crops with aspect to different region of Gujarat, Ph.D. thesis, Gujarat University.

Kumschick, S., Bacher, S., Evans, T., Marková, Z., Pergl, J., Pyšek, P., Vaes-Petignat, S., van der Veer, G., Vilà, M. and Nentwig, W.,2015. Comparing impacts of alien plants and animals in E urope using a standard scoring system. *Journal of Applied Ecology*, 52(3): 552-561.

Kushwaha, C.P., Tripathi, S.K., Singh, G.S. and Singh, K.P., 2010. Diversity of deciduousness and phenological traits of key Indian dry tropical forest trees. *Annals of Forest Science*, 67(3), p.310.

Lacey, E.P. 1982. Timing of seed dispersal in *Daucus carota*. *Oikos*, 39: 83-91.

Lake, J.C. and Leishman, M.R. 2004. Invasion success of exotic plants in natural ecosystems: the role of disturbance, plant attributes and freedom from herbivores. *Biological Conservation*, 117(2): 215-226.

Lal, B., Parkash, O.M., Sharma, V., Singh, R.D. and Uniyal, S.K. 2009. *Synedrella Vialis* (Less.) A. Gray--A New Record to The Flora of Himachal Pradesh. *Indian Forester*, 135(1): 89-91.

Lalmuanpuii, R., Zothanpuia, Zodinpuii, B. and Lalbiaknunga, J. 2020. Phenological observations of selected wild edible vegetables from tropical and subtropical forest of Mizoram, Northeast India. *Vegetos*, 33: 409-419.

Lamare, R., Sarangthem, I., Singh, A.H., Luikham, E., Singh, G., Devi, N.S., Sharma, L.D. and Oinam, N. 2023. Correlation between the different fractions of boron with soil physico-chemical properties. *The Pharma Innovation Journal*, 12(1): 1593-1596.

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

Lang, W., Chen, X., Qian, S. and Schwartz, M.D. 2024. Temperature variations impacting leaf senescence initiation pathways alter leaf fall timing patterns in northern deciduous forests. *Science of The Total Environment*, 934: 173280.

Langeland, K.A., Cherry, H. M., McCormick, C. M. and Craddock-Burks, K. A. 2008. Identification and Biology of Non-native Plants in Florida's Natural Areas. Gainesville, Florida, USA: University of Florida IFAS Extension.

Lankau, R. 2010. Soil microbial communities alter allelopathic competition between *Alliaria petiolata* and a native species. *Biological Invasions*, 12(7): 2059-2068.

Latombe, G., Pyšek, P., Jeschke, J.M., Blackburn, T.M., Bacher, S., Capinha, C., Costello, M.J., Fernández, M., Gregory, R.D., Hobern, D. and Hui, C. 2017. A vision for global monitoring of biological invasions. *Biological Conservation*, 213: 295-308.

Laube, J. 2015. Performance of native and invasive plant species under climate change–phenology, competitive ability and stress tolerance, Ph.D. thesis, Technische Universität München.

Lawrence, G.B., Fernandez, I.J., Hazlett, P.W., Bailey, S.W., Ross, D.S., Villars, T.R., Quintana, A., Ouimet, R., McHale, M.R., Johnson, C.E. and Briggs, R.D. 2016. Methods of soil resampling to monitor changes in the chemical concentrations of forest soils. *JoVE (Journal of Visualized Experiments),* 117: e54815.

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

Lazzaro, L., Essl, F., Lugliè, A., Padedda, B.M., Pyšek, P. and Brundu, G., 2018. Invasive Alien Plant Impacts on Human Health and Well-being, 16-33. In Invasive Species and Human Health Pages, Mazza, G. and Tricario, E. (eds.), CAB International Publishing, Wallingford.

Leary, J.K., Hue, N.V., Singleton, P.W. and Borthakur, D. 2006. The major features of an infestation by the invasive weed legume gorse (*Ulex europaeus*) on volcanic soils in Hawaii. *Biology and Fertility of Soils*, 42: 215-223.

Leclère, D., Obersteiner, M., Barrett, M., Butchart, S.H., Chaudhary, A., De Palma, A., DeClerck, F.A., Di Marco, M., Doelman, J.C., Dürauer, M. and Freeman, R. 2020. Bending the curve of terrestrial biodiversity needs an integrated strategy. *Nature*, 585(7826): 551-556.

Lediuk, K.D., Damascos, M.A., Puntieri, J.G. and Svriz, M. 2014. Differences in phenology and fruit characteristic between invasive and native woody species favor exotic species invasiveness. *Plant Ecology*, 215: 1455-1467.

Lee, S., Jeong, S., Park, C.E. and Kim, J., 2022. A Simple Method of Predicting Autumn Leaf Coloring Date Using Machine Learning with Spring Leaf Unfolding Date: A Simple Method of Predicting Autumn Leaf Coloring Date Using Machine Learning with Spring Leaf Unfolding Date. *Asia-Pacific Journal of Atmospheric Sciences*, 58(2), pp.219-226.

Lee, S., Klinger, R., Brooks, M.L. and Ferrenberg, S. 2024. Homogenization of soil seed bank communities by fire and invasive species in the Mojave Desert. *Frontiers in Ecology and Evolution*, 12: 1271824.

Lesica, P. and Kittelson, P.M. 2010. Precipitation and temperature are associated with advanced flowering phenology in a semi-arid grassland. *Journal of Arid Environments,* 74(9): 1013-1017.

Levine, J.M., Adler, P.B. and Yelenik, S.G. 2004. A meta-analysis of biotic resistance to exotic plant invasions. *Ecology letters*, 7(10): 975-989.

Levine, J.M., Vilà, M., Antonio, C.M.D., Dukes, J.S., Grigulis, K. and Lavorel, S. 2003. Mechanisms underlying the impacts of exotic plant invasions. *Proceedings of the Royal Society of London. Series B: Biological Sciences*, 270(1517): 775-781.

Leyva, C., Espejel, I., Escofet, A. and Bullock, S.H. 2006. Coastal landscape fragmentation by tourism development: impacts and conservation alternatives. *Natural Areas Journal*, 26(2): 117-125.

Li, F.F., Hao, Q., Cui, X., Lin, R.Z., Luo, B.S. and Ma, J.S. 2024. Global invasive alien plant management lists: Assessing current practices and adapting to new demands. *Plant Diversity*. 47: 666-680.

Li, P., Zhang, T., Wang, X. and Yu, D. 2013. Development of biological soil quality indicator system for subtropical China. *Soil and Tillage Research*, 126: 112-118.

Li, W.H., Zhang, C.B., Jiang, H.B., Xin, G.R. and Yang, Z.Y. 2006. Changes in soil microbial community associated with invasion of the exotic weed, *Mikania micrantha* HBK. *Plant and Soil*, 281: 309-324.

Li, X., Shen, Y., Huang, Q., Fan, Z. and Huang, D. 2013. Regeneration capacity of small clonal fragments of the invasive *Mikania micrantha* HBK: effects of burial depth and stolon internode length. *PLoS One*, 8(12): e84657.

Liao, C., Peng, R., Luo, Y., Zhou, X., Wu, X., Fang, C., Chen, J. and Li, B. 2008. Altered ecosystem carbon and nitrogen cycles by plant invasion: a meta-analysis. *New phytologist*, 177(3): 706-714.

Liebhold, A.M., Brockerhoff, E.G., Kalisz, S., Nuñez, M.A., Wardle, D.A. and Wingfield, M.J. 2017. Biological invasions in forest ecosystems. *Biological Invasions*, 19(11): 3437-3458.

Lim, T.K. and Lim, T.K. 2013. *Solanum torvum*, 429-441. In Edible Medicinal and Non-Medicinal Plants: Volume 6, Fruits, Lim, T.K. (eds.), Springer Dordrecht.

Lin, H., Cao, M., Stoy, P.C. and Zhang, Y. 2009. Assessing self-organization of plant communities - A thermodynamic approach. *Ecological Modelling*, 220(6): 784-790.

Lin, W., Zhou, G., Cheng, X. and Xu, R. 2007. Fast economic development accelerates biological invasions in China. *PLoS One*, 2(11): e1208.

Lindenmayer, D.B. and McCarthy, M.A. 2001. The spatial distribution of non-native plant invaders in a pine–eucalypt landscape mosaic in south-eastern Australia. *Biological Conservation*, 102(1): 77-87.

Linders, T.E.W., Schaffner, U., Eschen, R., Abebe, A., Choge, S.K., Nigatu, L., Mbaabu, P.R., Shiferaw, H. and Allan, E. 2019. Direct and indirect effects of invasive species: Biodiversity loss is a major mechanism by which an invasive tree affects ecosystem functioning. *Journal of Ecology*, 107(6): 2660-2672.

Lindsay, E.A. and French, K. 2005. Litterfall and nitrogen cycling following invasion by *Chrysanthemoides monilifera* ssp. *rotundata* in coastal Australia. *Journal of Applied Ecology*, 42(3): 556-566.

Lini, N., Reghu, R.J., Narayanan, M., Santhosh, R.S., Sugathan, S., Chithra, A.A.K. and Sophy, A.J.K. 2022. Plant Invasion by *Chromolaena Odorata* Alters Soil Microbiome and Provides Insight into the Role of the Fcb Group of Bacteria. *Ecological Genetics and Genomics*, 26: 100157.

Linnaeus, C. 1751. Philosophia botanica, in qua explicantur fundamenta botanica cum definitionibus partium, exemplis terminorum, observationibus rariorum, adjectis figuris aeneis. Kiesewetter, Stockholm.

Liogier, A.H. 1995. Descriptive flora of Puerto Rico and adjacent islands (Vol. 4). La Editorial, UPR.

Liptzin, D. and Silver, W.L. 2009. Effects of carbon additions on iron reduction and phosphorus availability in a humid tropical forest soil. *Soil Biology and Biochemistry*, 41(8): 1696-1702.

Liu, D., Essl, F., Lenzner, B., Moser, D., Semenchuk, P., Blackburn, T.M., Cassey, P., Biancolini, D., Capinha, C., Dawson, W. and Dyer, E.E. 2024. Regional invasion history and land use shape the prevalence of non-native species in local assemblages. *Global Change Biology*, 30(7): e17426.

Liu, F., Liu,.J. and Ming, D. 2016. Ecological consequences of clonal integration in plants. *Frontiers in Plant Science*, 7(770).

Liu, J., Liang, S.C., Liu, F.H., Wang, R.Q. and Dong, M. 2005. Invasive alien plant species in China: regional distribution patterns. *Diversity and Distributions*, 11(4): 341-347.

Lloret, F., Medail, F., Brundu, G., Camarda, I., Moragues, E.V.A., Rita, J., Lambdon, P. and Hulme, P.E. 2005. Species attributes and invasion success by alien plants on Mediterranean islands. *Journal of Ecology,* 93: 512-520.

Lodge, D.M. 1993. Biological invasions: lessons for ecology. *Trends In Ecology & Evolution*, 8(4): 133-137.

Lonsdale, W.M. 1999. Global patterns of plant invasions and the concept of invasibility. *Ecology*, 80(5): 1522-1536.

Lovejoy, T.E.,, Bierregaard, R.O., Rylands, A.B., Malcolm, J.R., Quintela, C.E., Harper, L.H., Brown, K.S., Powell, A.W., Powell, G.V.N., Schubart, H.O.R., Hays, M.B. 1986. Edge and other tropical lowland rain forest trees. II. Pollination systems. *American Journal of Botany,* 72: 345-356.

Lowe, S., Browne, M., Boudjelas, S. and De Poorter, M. 2000. 100 of the world's worst invasive alien species: a selection from the global invasive species database (Vol. 12). Auckland: Invasive Species Specialist Group.

Luo, Z., Sun, O.J., Ge, Q., Xu, W. and Zheng, J. 2007. Phenological responses of plants to climate change in an urban environment. *Ecological Research*, 22: 507-514.

MacArthur, R.H. 1957. On the relative abundance of bird species. *Proceedings of the National Academy of Sciences,* 43(3): 293-295.

Macdonald, I.A., Thébaud, C., Strahm, W.A. and Strasberg, D. 1991. Effects of alien plant invasions on native vegetation remnants on La Réunion (Mascarene Islands, Indian Ocean). *Environmental Conservation*, 18(1): 51-61.

Macdonald, I.A.W., Reaser, J.K., Bright, C., Neville, L.E., Howard, G.W., Murphy, S.J. and Preston, G. 2003. Invasive alien species in Southern Africa: national reports and directory of resources. Global Invasive Species Programme, Cape Town.

Machado, I.C., Barros, L.M. and Sampaio, E.V. 1997. Phenology of caatinga species at Serra Talhada, PE, northeastern Brazil. *Biotropica,* 29(1): 57-68.

Mack, M.C. and D'Antonio, C.M. 1998. Impacts of biological invasions on disturbance regimes. *Trends in Ecology & Evolution*, 13(5): 195-198.

Mack, R.N., Simberloff, D., Mark Lonsdale, W., Evans, H., Clout, M. and Bazzaz, F.A. 2000. Biotic invasions: causes, epidemiology, global consequences, and control. *Ecological Applications,* 10(3): 689-710.

Mahmoodi, S., Aghaee, M. and Makhdumi, M.A. 2025. Invasive Plant Species and Their Consequences. *Iran Nature,* 9(6): 23-31.

Mainka, S.A. and Howard, G.W. 2010. Climate change and invasive species: double jeopardy. *Integrative Zoology*, 5(2): 102-111.

Malik, Z.A., Pandey, R. and Bhatt, A.B. 2016. Anthropogenic disturbances and their impact on vegetation in Western Himalaya, India. *Journal of Mountain Science*, 13(1): 69-82.

Manchester, S.J. and Bullock, J.M. 2000. The impacts of non-native species on UK biodiversity and the effectiveness of control. *Journal of Applied Ecology,* 37(5): 845-864.

Mandal, G. and Joshi, S.P. 2014a. Invasion establishment and habitat suitability of *Chromolaena odorata* (L.) King and Robinson over time and space in the western Himalayan forests of India. *Journal of Asia-Pacific Biodiversity*, 7(4): 391-400.

Mandal, G. and Joshi, S.P. 2014b. The role of habitat types and soil physicochemical properties in the spread of a non-native shrub *Lantana camara* in the Doon valley, Western Himalaya, India. *Journal of Environmental Geography*, **7**(3-4): 31-42.

Mandal, L.N. and Mandal, B. 1986. Zinc fractions in soils in relation to zinc nutrition of lowland rice. *Soil Science*, 142(3): 141-148.

Mangla, S.M., Inderjit, I. and Callaway, R.M. 2008. Exotic invasive plant accumulates native soil pathogens which inhibit native plants. *Journal of Ecology (Oxford*), 96(1): 58-67.

Manjaiah, K.M., Voroney, R.P. and Sen, U. 2000. Soil organic carbon stocks, storage profile and microbial biomass under different crop management systems in a tropical agricultural ecosystem. *Biology and Fertility of Soils*, 32: 273-278.

Mao, G., Liu, R. and Zhang, N. 2023. Similarity analysis of chemical elements based on compounds network. *Journal of Mathematical Chemistry*, 61(7): 1522-1531.

Mapaura, A., Canavan, K., Richardson, D.M., Clark, V.R., Sutton, G.F. and Steenhuisen, S.L. 2024. The impact of *Nassella trichotoma* (Nees) Hack. ex Arechav. on plant diversity, richness and soil properties in South Africa. *South African Journal of Botany*, 173: 175-183.

Maron, J.L., Vilà, M., Bommarco, R., Elmendorf, S. and Beardsley, P. 2004. Rapid evolution of an invasive plant. *Ecological Monographs*, 74(2): 261-280.

Marques, M.C., Roper, J.J. and Baggio Salvalaggio, A.P. 2004. Phenological patterns among plant life-forms in a subtropical forest in southern Brazil. *Plant Ecology*, 173: 203-213.

Marschner, H. 2011. Marschner's mineral nutrition of higher plants. Academic press.

Martin, R. and Chanthy, P. 2009. Weeds of upland crops in Cambodia. Australian Government, Australian Centre for International Agricultural Research.

Martina, M., Jumari, J. and Murningsih, M. 2021. Phenetic analysis of turkey berry (*Solanum torvum* Sw.) based on morphological character in Semarang region. In Journal of Physics: Conference Series, 1943(1), p. 012076). IOP Publishing.

Maryanto, A.E., Salamah, A., Windarti, C.K. and Syadewi, M. 2021. Local Adaptation of Invasive Plant, *Synedrella nodiflora*, in Urban Tropical Lowland Landscape, Universitas Indonesia. *Journal of Tropical Biodiversity and Biotechnology*, 6(3): 64622.

Mathakutha, R., Steyn, C., le Roux, P.C., Blom, I.J., Chown, S.L., Daru, B.H., Ripley, B.S., Louw, A. and Greve, M. 2019. Invasive species differ in key functional traits from native and non-invasive alien plant species. *Journal of Vegetation Science*, 30(5): 994-1006.

Mathur, R. and Sudan, P. 2011. Relationship and distribution of various forms of boron with different physico chemical properties of soil in bikanerdistrict. *Journal of Chemical and Pharmaceutical Research*, 3(3): 290-294.

Maurel, N., Salmon, S., Ponge, J.F., Machon, N., Moret, J. and Muratet, A. 2010. Does the invasive species *Reynoutria japonica* have an impact on soil and flora in urban wastelands? *Biological invasions*, 12(6): 1709-1719.

Mbatudde, M., Mucunguzi, P. and Lye, K.A. 2007. Phenology of Asteraceae in selected districts of central Uganda. *African Journal of Ecology*, 45: 67-72.

Mcalpine, K.G., Jesson, L.K. and Kubien, D.S. 2008. Photosynthesis and water-use efficiency: A comparison between invasive (exotic) and non-invasive (native) species. *Austral Ecology*, 33(1): 10-19.

McCormack, G. 2007. Cook Islands biodiversity database, version 2007.2. Cook Islands Natural Heritage Trust, Rarotonga.

McDougall, K.L., Morgan, J.W., Walsh, N.G. and Williams, R.J. 2005. Plant invasions in treeless vegetation of the Australian Alps. *Perspectives in Plant Ecology, Evolution and Systematics*, 7(3): 159-171.

McFadyen, R.C. and Skarratt, B. 1996. Potential distribution of *Chromolaena odorata* (siam weed) in Australia, Africa and Oceania. *Agriculture, Ecosystems & Environment*, 59(1-2): 89-96.

McFadyen, R.C. 1988, February. Ecology of *Chromolaena odorata* in the neotropics, 13-20. In Proceedings of the first international workshop on biological control of *Chromolaena odorata*). Mangilao, Guam: Agricultural Experiment Station, University of Guam.

McFadyen, R.C. 2003. Chromolaena in East Timor: history, extent and control. Chromolaena in the Asia-Pacific region, 8-10. In Proc. of the 6th International Workshop on Biological Control and Management of Chromolaena, Cairns, Australia.

McKinney, M.L. 2006. Urbanization as a major cause of biotic homogenization. *Biological Conservation*, 127(3): 247-260.

MEA. 2005. Millennium Ecosystem Assessment: Ecosystems and human well-being: wetlands and water. World Resources Institute, Washington, D.C., United States.

Medal, J., Overholt, W., Charudattan, R., Mullahey, J., Gaskala, R., Diaz, R. and Cuda, J. 2012. Tropical soda apple management plan. UF/IFAS and FDACS/DPI, Gainesville FL.

Medvecka, J., Jarolímek, I., Hegedüšová, K., Škodová, I., Bazalová, D., Botková, K. and Šibíková, M. 2018. Forest habitat invasions–Who with whom, where and why. *Forest Ecology and Management*, 409: 468-478.

Meffin, R., Miller, A.L., Hulme, P.E. and Duncan, R.P. 2010. Experimental introduction of the alien. *Diversity and Distributions*, 16: 804-815.

Mehra, A., Tewari, L.M. and Rawal, R.S. 2023. Structure and composition of vegetation and status of invasion in different forest types of Western Himalaya. *Advances in Zoology and Botany*, 11: 12-29.

Meitei, L.R., De, A. and Mao, A.A. 2022. An ethnobotanical study on the wild edible plants used by forest dwellers in Yangoupokpi Lokchao Wildlife Sanctuary, Manipur, India. *Ethnobotany Research and Applications*, 23: 15.

Menzel, A. 2002. Phenology: its importance to the global change community. *Climatic Change*, 54(4): 379.

Merceron, N.R., Leroy, T., Chancerel, E., Romero-Severson, J., Borkowski, D.S., Ducousso, A., Monty, A., Porté, A.J. and Kremer, A. 2017. Back to America: tracking the origin of European introduced populations of *Quercus rubra* L. *Genome*, 60(9): 778-790.

Meyer, J.Y. 2000. Preliminary review of the invasive plants in the Pacific islands (SPREP Member Countries). Invasive species in the Pacific: A technical review and draft regional strategy, +85pp.

Milton, K., Windsor, D.M., Morrison, D.W. and Estribi, M.A. 1982. Fruiting phenologies of two neotropical Ficus species. *Ecology*, 63(3): 752-762.

Ming, L.C. 1999. *Ageratum conyzoides*: A tropical source of medicinal and agricultural products. *Perspectives on new crops and new uses*, 1988: 469-473.

Misra, R.1968. Ecology Work Book. Oxford and IBH Publishing Co., New Delhi.

Mittermeier, R.A., Hoffmann, M., Pilgrim, J., Brooks, T., Lamoreux, J., Mittermeier, C.G., Gil, P.R., Da Fonseca, G.A.B. 2004. Hotspots revisited: Earth's biologically richest and most endangered terrestrial ecoregions. CEMEX, Mexico City

Mittermeier, R.A., Myers, N., Thomsen, J.B., Da Fonseca, G.A. and Olivieri, S. 1998. Biodiversity hotspots and major tropical wilderness areas: approaches to setting conservation priorities. *Conservation Biology*, 12: 516-520.

Mohanta, M.R., Suresh, H. and Sahu, S.C. 2020. A Review on Plant Phenology Study in Different Forest Types of India. *Indian Forester*, 146: 1137–1148.

Mohandass, D., Campbell, M.J., Chen, X.S. and Li, Q.J., 2018. Flowering and fruiting phenology of woody trees in the tropical-seasonal rainforest, Southwestern China. Cu*rrent* science, pp.2313-2322.

Mollot, G., Pantel, J.H. and Romanuk, T.N. 2017. The effects of invasive species on the decline in species richness: a global meta-analysis. *In Advances in Ecological Research*, 56: 61-83.

Monlai, S. 2013. Diversity and distribution of ethno-medicinal plants in Lohit and Anjaw Districts of Arunachal Pradesh, Ph.D. thesis, Mizoram University.

Moody, K., 1989. Weeds reported in rice in South and Southeast Asia. International Rice Research Institute. +442 pp.

Moody, K., Munroe, C.E., Lubigan, R.T. and Paller, E.C. 1984. Major weeds of the Philippines, University of the Philippines College, +328pp.

Mooney, A. and Hobbs, H. 2000. Invasive species in a changing world. Island press. +457pp.

Mooney, H.A. and Cleland, E.E. 2001. The evolutionary impact of invasive species. *Proceedings of the National Academy of Sciences*, 98(10): 5446-5451.

Mooney, H.A., 2005. Invasive alien species: the nature of the problem. Scope-scientific committee on problems of the environment international council of scientific unions, 63, p.1.

Moore, D., Gange, A.C., Gange, E.G. and Boddy, L., 2008, January. Fruit bodies: their production and development in relation to environment. In British Mycological Society Symposia Series (Vol. 28, pp. 79-103). Academic Press.

Mortensen, L.M., 2000. Effects of air humidity on growth, flowering, keeping quality and water relations of four short-day greenhouse species. *Scientia horticulturae*, 86(4), pp.299-310.

Mortensen, L.M., 1986. Effect of relative humidity on growth and flowering of some greenhouse plants. *Scientia horticulturae*, 29(4), pp.301-307.

Morais, M.C. and Freitas, H. 2015. Phenological dynamics of the invasive plant *Acacia longifolia* in Portugal. *Weed Research*, 55(6): 555-564.

Moravcova, L., Pyšek, P., Jarošík, V., Havlíčková, V. and Zákravský, P. 2010. Reproductive characteristics of neophytes in the Czech Republic: traits of invasive and non-invasive species. *Preslia*, 82(4): 365-390.

Morellato, L.P.C., Camargo, M.G.G., D'Eça Neves, F.F., Luize, B.G., Mantovani, A. and Hudson, I.L. 2010. The influence of sampling method, sample size, and frequency of observations on plant phenological patterns and interpretation in tropical forest trees, 99-121. In Phenological Research: Methods for Environmental and Climate Change Analysis, Hudson, I.L. and Keatley, M.R. (eds.), Springer Netherlands.

Morisette, J.T., Richardson, A.D., Knapp, A.K., Fisher, J.I., Graham, E.A., Abatzoglou, J., Wilson, B.E., Breshears, D.D., Henebry, G.M., Hanes, J.M. and Liang, L. 2009. Tracking the rhythm of the seasons in the face of global change: phenological research in the 21st century. *Frontiers in Ecology and the Environment*, 7(5): 253-260.

Motooka, P. 2003. Weeds of Hawaii's pastures and natural areas; an identification and management guide. In: *Weeds of Hawaii's pastures and natural areas; an identification and management guide*. USA: College of Tropical Agriculture and Human Resources, University of Hawaii.

Mousavifard, S.M., Momtaz, H., Sepehr, E., Davatgar, N. and Sadaghiani, M.H.R. 2013. Determining and mapping some soil physico-chemical properties using geostatistical and GIS techniques in the Naqade region, Iran. *Archives of Agronomy and Soil Science*, 59(11): 1573-1589.

Mueller-Dombois, D. and Ellenberg, H. 1974. Aims and methods of vegetation ecology. John Willey and Sons. Inc., New York.

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

Mukherjee, I. 1969. Studies on flowering responses of *Urena lobata*. *Plant physiology*, 44(12): 1749-1751.

Mullahey, J.J. and Akanda, R.U. 1996. Reproductive biology and control of tropical soda apple (*Solanum viarum*). Proceed. South. *Weed Science Society*. 49: 146 – 147

Mullahey, J.J., 2012. Biology ecology and control of tropical soda apple (Sola*num viarum*). *Pakistan Journal of Weed Science Research*. 18: 447-456.

Mullahey, J.J., Nee, M., Wunderlin, R.P. and Delaney, K.R. 1993. Tropical soda apple (*Solanum viarum*): a new weed threat in subtropical regions. *Weed Technology*, 7(3): 783-786.

Mungi, N.A., Qureshi, Q. and Jhala, Y.V. 2021. Role of species richness and human impacts in resisting invasive species in tropical forests. *Journal of Ecology*, 109(9): 3308-3321.

Muniappan, R. and Bamba, J. 1999, July. Biological control of Chromolaena odorata: successes and failures, 81-85. In Proceedings of the X international symposium on biological control of weeds, Bozeman, Montana, USA: Montana State University.

Muniappan, R., Reddy, G.V. and Raman, A. 2009. Biological control of tropical weeds using arthropods. Cambridge University Press.

Muniappan, R., Reddy, G.V.P. and Lai, P.Y. 2005. Distribution and biological control of *Chromolaena odorata*, 223-233. In Invasive plants: ecological and agricultural aspects, Inderjit (eds.), Basel: Birkhäuser.

Murtem, G. 2000. Common wild vegetables of Nyishi tribe of Arunachal Pradesh. *Bulletin of Arunachal Forest Research*, 18(1&2): 66-77.

Mustaqeem Ahmad, Sanjay, K.U. and Singh, R.D. 2018. Patterns of alien plant species richness across gradients of altitude: analyses from the Himalayan state of Himachal Pradesh. *Tropical Ecology*, 59(1): 35-43.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., Da Fonseca, G.A. and Kent, J. 2000. Biodiversity hotspots for conservation priorities. *Nature*, 403(6772): 853-858.

Myśliwy, M. 2014. Plant invasions across different habitat types at floristic survey. *Applied Ecology and Environmental Research*, 12(1): 193-207.

Nable, R.O., Bañuelos, G.S. and Paull, J.G. 1997. Boron toxicity. *Plant and Soil*, 193: 181-198.

Naidu, V.G.S.R., Yaduraju, N.T. and Gogoi, A.K. 2005. Weed that Heal, National Research Centre for Weed Science, Jabalpur, India,+120pp.

Nanda, A., Suresh, H.S. and Krishnamurthy, Y.L., 2014. Phenology of a tropical dry deciduous forest of Bhadra wildlife sanctuary, southern India. *Ecological processes*, 3(1), p.1.

Nakar, R.N. and Jadeja, B.A., 2015. Flowering and fruiting phenology of some herbs, shrubs and undershrubs from Girnar Reserve Forest, Gujarat, India. Current Science, pp.111-118.

Nasja, M.P., Razak, I.P. and Rafeeq, K.U.M.A. 2024. Effect of recent seasonal rainfall pattern on the phenology of a common fig plant from north Kerala, South India. *Ecology, Environment & Conservation* (0971765X), 30: 492-501.

National Focal Point for APFISN. 2005. Stocktaking of National Forest Invasive Species Activities. Ministry of Environment and Forests, Delhi, India.

Naude, M. 2012. Fynbos riparian biogeochemistry and invasive Australian acacias Ph.D. thesis, Stellenbosch: Stellenbosch University.

Nayak, B.G. and Patil, S.K. 2001. Variation of solasodine in *Solanum viarum* Dunal with maturity stages. *Karnataka Journal of Agriculture Science*, 14: 185 – 186.

Nayak, R., Verma, A.K., Manika, N., Chaudhary, L.B., Behera, S.K., Bargali, K. and Pandey, V.N. 2024. Distribution pattern and management of invasive alien plant species in Sikkim Himalaya, India. *Plant Science Today*, 11(2): 205-213.

Nee, M. 1991. Synopsis of *Solanum* Section Acanthophora: A group of interest for glycoalkaloids, 257-266. In Solanaceae III, *Taxonomy, Chemistry, Evolution*, In Solanaceae III: taxonomy, chemistry, evolution, Hawkes, J.G., Lester, R.N., Nee, M and Estrada-R, N. (eds.), Kew Publishing.

Negi, G.C.S. 1989. Phenology and nutrient dynamics of tree leaves in Kumaun Himalayan forests. Ph.D. Thesis, Kumaun University, p.243.

Negi, G.C.S., Joshi, S., Singh, P. and Joshi, R. 2022. Phenological response patterns of forest communities to annual weather variability at long-term ecological monitoring sites in Western Himalaya. *Trees, Forests and People*, 8: 100237.

Negi, G.C.S., Si Ralhan, P.K., Khanna, R.K., Singh, S.P. and Singh, J.S. 1985. Phenological characteristics of the tree layer of Kumaun Himalayan forests. *Vegetatio*, 60: 91-101.

Negi, G.C.S., Singh, P. and Singh, S.P. 2021. Atmospheric warming-associated phenological earliness does not increase the length of growing season in himalayan trees. *Forest Science*, 67(6): 694-700.

Neilson, R.P. 1995. A model for predicting continental-scale vegetation distribution and water balance. *Ecological Applications*, 5(2): 362-385.

Neto, E.M.L., Almeida, A.L., Peroni, N., Castro, C.C. and Albuquerque, U.P. 2013. Phenology of *Spondias tuberosa* Arruda (Anacardiaceae) under different landscape management regimes and a proposal for a rapid phenological diagnosis using local knowledge. *Journal of Ethnobiology and Ethnomedicine*, 9: 1-13.

Ng, C.C., Wu, S.J., Wang, C.Y., Tzeng, W.S. and Shyu, Y.T. 2011. Emergence and growth of beggarticks (*Bidens pilosa* var. *radiata*) in different plant communities under experimental field conditions. *Journal of Agricultural Science and Technology A*, 1: 950-962.

Ni, G., Zhao, P., Huang, Q., Zhu, L., Hou, Y., Yu, Y., Ye, Y. and Ouyang, L. 2020. Mikania micrantha invasion enhances the carbon (C) transfer from plant to soil and mediates the soil C utilization through altering microbial community. *Science of the Total Environment*, 711: 135020.

Nikolic, M. and Pavlovic, J. 2018. Plant responses to iron deficiency and toxicity and iron use efficiency in plants, 55-69. In Plant micronutrient use efficiency, Hossain, M.A., Kamiya, T., Buritt, D.J, Tran, L.P. and Fujiwara, T. (eds.), Academic Press. Elsevier: Amsterdam, The Netherlands.

Nimachow, G., Taga, T., Tag, H. and Dai, O. 2008. Linkages between bio-resources and human livelihood: a case study of Adi tribes of Mirem Village, Arunachal Pradesh (India). *Initiation*, 2(1): 183-98.

Nord, E.A. and Lynch, J.P. 2009. Plant phenology: a critical controller of soil resource acquisition. *Journal of Experimental Botany*, 60(7): 1927-1937.

Novoa, A., Dehnen-Schmutz, K., Fried, J. and Vimercati, G. 2017. Does public awareness increase support for invasive species management? Promising evidence across taxa and landscape types. *Biological Invasions*, 19(12): 691-3705.

Nozzolillo, C. 1970. A Mineral Nutrition Experiment with *Bryophyllum daigremontianum*. *Bioscience*, 20(16): 916.

O’Connell, E. and Savage, J. 2020. Extended leaf phenology had limited benefits for invasive species growing at northern latitudes. *Biological Invasions*, 22(10): 2957-2974.

O’Hare, T.J. 2002. Interaction of temperature and vegetative flush maturity influences shoot structure and development of lychee (*Litchi chinensis* Sonn.). *Scientia Horticulturae*, 95(3): 203-211.

Ohsawa, M., Shakya, P.R. and Numata, M. 1986. Distribution and succession of west Himalayan Forest types in the eastern part of the Nepal Himalaya. *Mountain Research and Development*, 6: 143-157.

Ohtsuka, T. 1999. Early stages of secondary succession on abandoned cropland in north-east Borneo Island. *Ecological Research*, 14(3): 281-290.

Okalebo, J.R., Gathua, K.W. and Woomer, P.L. 2002. Laboratory methods of soil and plant analysis: a working manual second edition. Sacred Africa, Nairobi.

Okunade, A.L. 2002. *Ageratum conyzoides* L.(asteraceae). *Fitoterapia*, 73(1): 1-16.

Olenin, S., Alemany, F., Cardoso, A.C., Gollasch, S., Goulletquer, P., Lehtiniemi, M., McCollin, T., Minchin, D., Miossec, L., Ambrogi, A.O. and Ojaveer, H. 2010. Marine strategy framework directive. Task Group, 2.

Olenin, S., Gollasch, S., Lehtiniemi, M., Sapota, M. and Zaiko, A. 2017. Biological invasions, 193-232. In Biological oceanography of the Baltic Sea, Snoeijs-Leijonmalm, P., Schubert, H. and Radziejewskapp, T. (eds.), Springer Dordrecht.

Ollerton, J. and Lack, A.J. 1992. Flowering phenology: an example of relaxation of natural selection?. *Trends in Ecology and Evolution*, 7(8): 274-276.

Olsen, S.R. 1954. Estimation of available phosphorus in soils by extraction with sodium bicarbonate (No. 939). US Department of Agriculture.

Olson DM, Dinerstein E. 1998. The Global 200: a representation approach to conserving the Earth's most biologically valuable ecoregions. *Conservation Biology*. 1998 12(3): 502-15.

Oludare, A. and Muoghalu, J.I. 2014. Impact of *Tithonia diversifolia* (Hemsly) A. Gray on the soil, species diversity and composition of vegetation in Ile-Ife (Southwestern Nigeria), Nigeria. *International Journal of Biodiversity and Conservation*, 6(7): 555-562.

Ong, H.C. 2001. *Urena lobata* L, 575-577. In: Plant Resources of South-East Asia No. 12(2): Medicinal and poisonous plants, Valkenburg, J. L. C. H. and van Bunyapraphatsara, N. (eds.), Leiden, The Netherlands: Backhuys Publisher.

Opler, P.A., Frankie, G.W. and Baker, H.G. 1976. Rainfall as a factor in the release, timing, and synchronization of anthesis by tropical trees and shrubs. *Journal of Biogeography,* 3: 231-236.

Orlandi, F., Bonofiglio, T., Ruga, L., Sgromo, C., Romano, B. and Fornaciari, M. 2007. Phenological investigations of different winter-deciduous species growing under Mediterranean conditions. *Annals of Forest Science*, 64(5): 557-568.

Osaki, S., Wasaki, J. and Nakatsubo, T. 2022. Phenological shifts of the invasive annual weed *Bidens pilosa* var. *pilosa* in response to warmer temperature. *Plant Ecology*, 223(10): 1155-1165.

Osik, N.N. 1996. A Brief History of Arunachal Pradesh. Omson, New Delhi.

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

Osunkoya, O.O., Akinsanmi, O.A., Lim, L.S., Perrett, C., Callander, J. and Dhileepan, K. 2017. *Parthenium hysterophorus* L.(Asteraceae) invasion had limited impact on major soil nutrients and enzyme activity: Is the null effect real or reflects data insensitivity?. *Plant And Soil*, 420: 177-194.

Ovaskainen, O., Skorokhodova, S., Yakovleva, M., Sukhov, A., Kutenkov, A., Kutenkova, N., Shcherbakov, A., Meyke, E. and Delgado, M.D.M. 2013. Community-level phenological response to climate change. *Proceedings of the National Academy of Sciences*, 110(33): 13434-13439.

Ozolincius, R., Lekevičius, E., Stakėnas, V., Galvonaitė, A., Samas, A. and Valiukas, D. 2014. Lithuanian forests and climate change: possible effects on tree species composition. *European Journal of Forest Research*, 133(1): 51-60.

Page, N., Datta, A. and Basu, B. 2021 Trees of Arunachal Pradesh: A Field Guide. Nature Conservation Foundation, Mysore, +591 pp.

Paini, D.R., Sheppard, A.W., Cook, D.C., De Barro, P.J., Worner, S.P. and Thomas, M.B. 2016. Global threat to agriculture from invasive species. *Proceedings of the National Academy of Sciences*, 113(27): 7575-7579.

Parker, C. 1992. Weeds of Bhutan New York, USA: Springer.

Parker, C. 1972. The Mikania problem. *Tropical Pest Management*, 18(3): 312-315.

Parker, I.M., Simberloff, D., Lonsdale, W.M., Goodell, K., Wonham, M., Kareiva, P.M., Williamson, M.H., Von Holle, B.M.P.B., Moyle, P.B., Byers, J.E. and Goldwasser, L. 1999. Impact: toward a framework for understanding the ecological effects of invaders. *Biological Invasions*, 1: 3-19.

Parry, M. L., Canziani, O. F., Palutikof, J. P., van der Linden, P. J., & Hanson, C. E. 2007. Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working

Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change, Cambridge: Cambridge University Press, +976p.

Pathak, R., Negi, V.S., Rawal, R.S. and Bhatt, I.D. 2019. Alien plant invasion in the Indian Himalayan Region: state of knowledge and research priorities. *Biodiversity and Conservation*, 28(12): 3073-3102.

Pauchard, A., Kueffer, C., Dietz, H., Daehler, C.C., Alexander, J., Edwards, P.J., Arévalo, J.R., Cavieres, L.A., Guisan, A., Haider, S. and Jakobs, G. 2009. Ain't no mountain high enough: plant invasions reaching new elevations. *Frontiers in Ecology and the Environment*, 7(9): 479-486.

Paul, T.K. and Kumar, A. 2023. Flora of Mahananda Wildlife Sanctuary, Darjeeling, West Bengal. Botanical Survey of India, Kolkata. +171pp.

Peet, R.K., Wentworth, T.R. and White, P.S. 1998. A flexible, multipurpose method for recording vegetation composition and structure. *Castanea*, 63: 262-274.

Pejchar, L. and Mooney, H.A. 2009. Invasive species, ecosystem services and human well-being. *Trends In Ecology & Evolution*, 24(9): 497-504.

Peltzer, D.A., Allen, R.B., Lovett, G.M., Whitehead, D. and Wardle, D.A. 2010. Effects of biological invasions on forest carbon sequestration. *Global Change Biology*, 16(2): 732-746.

Penuelas, J. and Filella, I. 2001. Responses to a warming world.(Perspectives: Phenology). *Science*, 294(5543): 793-795.

Penuelas, J., Sardans, J, Llusia J, Owen, S.M., Carnicer, J., Giambelluca, T.W., Rezende, E.L., Waite, M and Ninemets, U. 2010. Faster returns on ‘leaf economics’ and different biogeochemical niche in invasive compared with native plant species. *Global Change Biology* 16(8): 2171–2185.

Pereira, H.M., Ferrier, S., Walters, M., Geller, G.N., Jongman, R.H., Scholes, R.J., Bruford, M.W., Brummitt, N., Butchart, S.H., Cardoso, A.C. and Coops, N.C. 2013. Essential biodiversity variables. *Science*, 339(6117): 277-278.

Peres-Neto, P.R., Jackson, D.A. and Somers, K.M. 2005. How many principal components? Stopping rules for determining the number of non-trivial axes revisited. *Computational Statistics & Data Analysis*, 49(4): 974-997.

Perrett, C., Osunkoya, O.O. and Clark, C. 2012. Cat's claw creeper vine, *Macfadyena unguis-cati* (Bignoniaceae), invasion impacts: comparative leaf nutrient content and effects on soil physicochemical properties. *Australian Journal of Botany*, 60(6): 539-548.

Piao, S., Liu, Q., Chen, A., Janssens, I.A., Fu, Y., Dai, J., Liu, L., Lian, X.U., Shen, M. and Zhu, X. 2019. Plant phenology and global climate change: Current progresses and challenges. *Global Change Biology*, 25(6): 1922-1940.

Pickering, C., Hill, W. and Green, K. 2008. Vascular plant diversity and climate change in the alpine zone of the Snowy Mountains, Australia. *Biodiversity and Conservation*, 17: 1627-1644.

PIER. 2012. Pacific Islands Ecosystems at Risk. Honolulu, USA: HEAR, University of Hawaii. http://www.hear.org/pier/index.html

Pimentel, D. 2005. Environmental consequences and economic costs of alien species, 269-276. In Invasive plants: Ecological and agricultural aspects, inderjit (eds.), Birkhäuser Basel.

Pimentel, D., McNair, S., Janecka, J., Wightman, J., Simmonds, C., O'connell, C., Wong, E., Russel, L., Zern, J., Aquino, T. and Tsomondo, T. 2001. Economic and environmental threats of alien plant, animal, and microbe invasions. *Agriculture, Ecosystems & Environment*, 84(1): 1-20.

Plos, C., Hensen, I., Korell, L., Auge, H. and Römermann, C. 2024. Plant species phenology differs between climate and land-use scenarios and relates to plant functional traits. *Ecology and Evolution*, 14(5): e11441.

Polgar, C., Gallinat, A. and Primack, R.B. 2014. Drivers of leaf-out phenology and their implications for species invasions: insights from T horeau's C oncord. *New Phytologist*, 202(1): 106-115.

Post, E. and Stenseth, N.C. 1999. Climatic variability, plant phenology, and northern ungulates. *Ecology*, 80(4): 1322-1339.

Potgieter, L.J., Gaertner, M., O'Farrell, P.J. and Richardson, D.M. 2019. Perceptions of impact: invasive alien plants in the urban environment. *Journal Of Environmental Management*, 229: 76-87.

Potter, K.M., Giardina, C., Hughes, R.F., Cordell, S., Kuegler, O., Koch, A. and Yuen, E. 2023. How invaded are Hawaiian forests? Non-native understory tree dominance signals potential canopy replacement. *Landscape Ecology*, 38(12): 3903-3923.

Powell, K.I., Chade, J.M. and Knight, T.M. 2011. A synthesis of plant invasion effects on biodiversity across spatial scales. *American Journal Of Botany*, 98(3): 539-548.

Prakash, L. 2023. A Checklist of Flowering Plants Of Sree Ramu College Of Arts And Science, Pollachi, Coimbatore, Tamil Nadu, India. *Indian Journal of Plant Sciences*, 12: 68-90.

Prasad, U.K., Muniappan, R., Ferrar, P., Aeschliman, J.P., de Foresta, H. 1996. Proceedings of the third international workshop on biological control and management of Chromolaena odorata, Abidjan, Côte d'Ivoire, November 1993. Publication No. 202, Agricultural Experimental Station, University of Guam, USA, +213pp.

Prashanthi, S.K. and Kulkarni, S. 2005. Aureobasidium pullulans, a potential mycoherbicide for biocontrol of *eupatorium* [Chromolaena odorata (L.) King and Robinson] weed. *Current Science*, 88(1): 18-21.

Prentis, P.J., Wilson, J.R., Dormontt, E.E., Richardson, D.M. and Lowe, A.J. 2008. Adaptive evolution in invasive species. *Trends In Plant Science*, 13(6): 288-294.

Putterill, J., Laurie, R. and Macknight, R. 2004. It's time to flower: the genetic control of flowering time. *Bioessays*, 26(4): 363-373.

Pysek, P. and Prach, K. 1995. Invasion dynamics of Impatiens glandulifera - a century of spreading reconstructed. *Biological Conservation*, 74(1): 41-48.

Pysek, P. and Richardson, D.M. 2007. Traits associated with invasiveness in alien plants: where do we stand?, 97-125. In Biological invasions, Nentwig, W. (eds.), Springer Berlin, Heidelberg.

Pysek, P., Blackburn, T.M., García-Berthou, E., Perglová, I. and Rabitsch, W. 2017. Displacement and local extinction of native and endemic species, 157-175. In: Impact of biological invasions on ecosystem services, Vila, M and Hulme, P.E. (eds.), Springer Cham.

Pysek, P., Genovesi, P., Pergl, J., Monaco, A. and Wild, J. 2013. Plant invasions of protected areas in Europe: an old continent facing new problems, 209-240. In Plant invasions in protected areas: patterns, problems and challenges, Foxcroft, L.C., Pyšek, P., Richardson, D.M. and Genovesi, P. (eds.), Springer Dordrecht.

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

Pyšek, P., Jarošík, V. and Kučera, T. 2002. Patterns of invasion in temperate nature reserves. *Biological Conservation*, 104(1): 13-24.

Pyšek, P., Jarošík, V., Hulme, P.E., Pergl, J., Hejda, M., Schaffner, U. and Vilà, M. 2012. A global assessment of invasive plant impacts on resident species, communities and ecosystems: the interaction of impact measures, invading species' traits and environment. *Global Change Biology*, 18(5): 1725-1737.

Qi, Y., Xian, X., Zhao, H., Yang, M., Zhang, Y., Yu, W. and Liu, W., 2023. World Spread of Tropical Soda Apple (*Solanum viarum*) under Global Change: Historical Reconstruction, Niche Shift, and Potential Geographic Distribution. *Biology*, 12(9): 1179.

Qiang, S. and Cao, X.Z. 2001. Harmfulness of exotic weeds in China and for their management. *Biodiversity Science*, 9(2): 188.

Qu, T., Du, X., Peng, Y., Guo, W., Zhao, C. and Losapio, G., 2021. Invasive species allelopathy decreases plant growth and soil microbial activity. *PloS one*, 16(2): e0246685.

Rafferty, N.E., Diez, J.M. and Bertelsen, C.D. 2020. Changing climate drives divergent and nonlinear shifts in flowering phenology across elevations. *Current Biology*, 30(3): 432-441.

Raheem, A., Yohanna, P., Li, G., Noh, N.J., Iqbal, B., Tang, J., Du, D., Alahmadi, T.A., Ansari, M.J., Zhan, A. and Son, Y. 2024. Unraveling the ecological threads: How invasive alien plants influence soil carbon dynamics. *Journal of Environmental Management,* 356: 120556.

Raheem, A.M. and Omar, N.Q. 2021. Investigation of distinctive physico-chemical soil correlations for Kirkuk city using spatial analysis technique incorporated with statistical modeling. *International Journal of Geo-Engineering,* 12(1): 18.

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

Rai, P.K. and Singh, J.S. 2021. Plant invasion in protected areas, the Indian Himalayan region, and the North East India: progress and prospects. *Proceedings of the Indian National Science Academy*, 87(1): 19-35.

Rai, P.K. and Singh, J.S. 2024. Ecological insights and environmental threats of invasive alien plant *Chromolaena odorata*: Prospects for sustainable management. *Weed Biology and Management*, 24(1): 15-37.

Rai, P.K. 2022. Environmental degradation by invasive alien plants in the anthropocene: challenges and prospects for sustainable restoration. *Anthropocene Science*, 1(1): 5-28.

Rai, P.K., Kumar, V., Lee, S., Raza, N., Kim, K.H., Ok, Y.S. and Tsang, D.C. 2018. Nanoparticle-plant interaction: Implications in energy, environment, and agriculture. *Environment International*, 119: 1-19.

Rai, U., 2006. Characterisation of plant biodiversity in Darjiling Hills using remote sensing techniques, Ph.D. thesis, University of North Bengal.

Raimundo, R.L.G., Fonseca, R.L., Schachetti-Pereira, R., Peterson, A.T. and Lewinsohn, T.M. 2007. Native and exotic distributions of siamweed (Chromolaena odorata) modeled using the genetic algorithm for rule-set production. *Weed Science*, 55(1): 41-48.

Raizada, P., Raghubanshi, A.S. and Singh, J.S. 2008. Impact ofinvasive alien plant species on soil processes: A review. *Proceedings of the National Academy of Sciences India Section B–Biological Sciences*, 78(PART 4): 288-298.

Raju, A.S. and Rani, D.S. 2017. Pollination ecology of *Triumfetta rhomboidea* (Tiliaceae). *Annali di Botanica*, 7: 33-41.

Ralhan, P.K., Khanna, R.K., Singh, S.P. and Singh, J.S. 1985. Phenological characteristics of the tree layer of Kumaun Himalayan forests. *Vegetatio*, 60: 91-101.

Rambia, K and Rathore, S. 2022. Snakes of Pakke Tiger Reserve. Jyotsna Prakashan, +104pp. https://books.google.co.in/books?id=k0TMzwEACAAJ

Rameshprabu, N. and Swamy, P.S. 2015. Prediction of environmental suitability for invasion of *Mikania micrantha* in India by species distribution modelling. *Journal of Environmental Biology*, 36(3): 565.

Ramirez, K.S., Snoek, L.B., Koorem, K., Geisen, S., Bloem, L.J., Ten Hooven, F., Kostenko, O., Krigas, N., Manrubia, M., Caković, D. and van Raaij, D. 2019. Range-expansion effects on the belowground plant microbiome. *Nature Ecology & Evolution*, 3(4):604-611.

Rana, C.S. and Gairola, S. 2009. Forest community structure and composition along an elevational gradient of Parshuram Kund area in Lohit district of Arunachal Pradesh, India. *Nature and Science*, 1(8): 44-52.

Rana, S.S., Sharma, R., Singh, A. and Kumar, S. 2019. Studies on shifts in weed flora in maize (Zea mays L.) in Kangra district of Himachal Pradesh. Journal of Research in *Weed Science*, 2: 230-240.

Randall, R.P. 2012. A Global Compendium of weeds. Perth, Australia: Department of Agriculture and Food, Western Australia. +1124pp.

Rangwala, I. and Miller, J.R. 2012. Climate change in mountains: a review of elevation-dependent warming and its possible causes. *Climatic Change*, 114: 527-547.

Rao, R.R. 1977. Changing pattern in the Indian flora. *Nelumbo-The Bulletin of the Botanical Survey of India*, 19: 156-166.

Rathcke, B. and Lacey, E.P. 1985. Phenological patterns of terrestrial plants. *Annual Review Of Ecology and Systematics*, 16(1); 179-214.

Rathi, J. and Gopalakrishnan, S. 2005. Insecticidal activity of aerial parts of *Synedrella nodiflora* Gaertn (Compositae) on *Spodoptera litura* (Fab.). Jou*rnal of Central European Agriculture.* 6: 223-228.

Rathnayake, D.K. and Wijetunga, W.A.S. 2016. Species composition and visiting frequencies of flower visitors of *Chromolaena odorata* in a dry zone forest patch of Sri Lanka. *Psyche: A Journal of Entomology*, 2016(1): 8746251.

Rawal, R.S., Bankoti, N.S., Samant, S.S. and Pangtey, Y.P.S. 1991. Phenology of tree layer species from the timber line around Kumaun in Central Himalaya, India. *Vegetatio*, 93(2); 108-118.

Ray, D., Behera, M.D. and Jacob, J. 2019. Comparing invasiveness of native and non-native species under changing climate in North-East India: ecological niche modelling with plant types differing in biogeographic origin. *Environmental Monitoring and Assessment,* 191: 1-13.

Reddy, C.S. and Raju, V.S., 2009. *Aeschynomene americana* L. and *Mikania micrantha* Kunth–new invasive weeds in flora of Andhra Pradesh, India. *Journal of Economic and Taxonomic Botany,* 33(3): 540-541.

Reddy, C.S. 2008. Catalogue of invasive alien flora of India. *Life science journal*, 5(2); 84-89.

Reeves, P.H. and Coupland, G. 2000. Response of plant development to environment: control of flowering by daylength and temperature. *Current opinion in plant biology*, 3(1): 37-42.

Reinhart, K.O. and Callaway, R.M. 2006. Soil biota and invasive plants. *New phytologist,* 170(3): 445-457.

Rejmanek, M. and Richardson, D.M. 1996. What attributes make some plant species more invasive?. *Ecology*, 77(6): 1655-1661.

Rejmánek, M., Richardson, D.M., Higgins, S.I., Pitcairn, M.J. and Grotkopp, E. 2005. Ecology of invasive plants: state of the art. *Scope-scientific committee on problems of the environment international council of scientific unions*, 63; 104.

Renault, D., Hess, M.C., Braschi, J., Cuthbert, R.N., Sperandii, M.G., Bazzichetto, M., Chabrerie, O., Thiébaut, G., Buisson, E., Grandjean, F. and Bittebiere, A.K. 2022. Advancing biological invasion hypothesis testing using functional diversity indices. *Science of the Total Environment*, 834: 155102.

Renault, D., Leclerc, C., Colleu, M.A., Boutet, A., Hotte, H., Colinet, H., Chown, S.L. and Convey, P. 2022. The rising threat of climate change for arthropods from Earth's cold regions: Taxonomic rather than native status drives species sensitivity. *Global Change Biology*, 28(20): 5914-5927.

Ricciardi, A. 2012. Invasive species, 161-178. In Ecological systems: selected entries from the Encyclopedia of sustainability science and technology, Meyers, R. (eds.). New York, NY: Springer New York.

Richardson, A.D., Hufkens, K., Milliman, T., Aubrecht, D.M., Furze, M.E., Seyednasrollah, B., Krassovski, M.B., Latimer, J.M., Nettles, W.R., Heiderman, R.R. and Warren, J.M. 2018. Ecosystem warming extends vegetation activity but heightens vulnerability to cold temperatures. *Nature,* 560(7718): 368-371.

Richardson, D.M. and Pyšek, P., 2006. Plant invasions: merging the concepts of species invasiveness and community invasibility. *Progress in physical geography*, 30(3): 409-431.

Richardson, D.M. and Pysek, P. 2007. Classics in physical geography revisited: Elton, C.S. 1958. The Ecology of invasions by animals and plants. *Progress in Physical Geography*, 31: 659-666.

Richardson, D.M. and Pysek, P. 2008. Fifty years of invasion ecology–the legacy of Charles Elton. *Diversity And Distributions*, 14(2): 161-168.

Richardson, D.M. and Van Wilgen, B.W. 2004. Invasive alien plants in South Africa: how well do we understand the ecological impacts? working for water. *South African Journal of Science*, 100(1): 45-52.

Richardson, D.M. 2004. Plant invasion ecology - dispatches from the front line. *Diversity and Distributions*, 10(5-6): 315-319.

Richardson, D.M., Holmes, P.M., Esler, K.J., Galatowitsch, S.M., Stromberg, J.C., Kirkman, S.P., Pyšek, P. and Hobbs, R.J. 2007. Riparian vegetation: degradation, alien plant invasions, and restoration prospects. *Diversity and Distributions*, 13(1); 126-139.

Richardson, D.M., Pyšek, P., Rejmanek, M., Barbour, M.G., Panetta, F.D. and West, C.J. 2000. Naturalization and invasion of alien plants: concepts and definitions. *Diversity And Distributions,* 6(2); 93-107.

Roba, A.D., Mohammed, M. and Nigatu, L. 2017. Evaluation of soil physicochemical properties under the canopy of coffee shade trees effect (*Cordia africana* and *Erythrina abyssinica*) in arsi golelcha district, Ethiopia. *Journal of Resources Development and Management*, 32: 80-91.

Robertson, M.P., Caithness, N. and Villet, M.H. 2001. A PCA-based modelling technique for predicting environmental suitability for organisms from presence records. *Diversity and Distributions*, 7(1-2); 15-27.

Robertson, M.P., Villet, M.H., Fairbanks, D.H., Henderson, L., Higgins, S.I., Hoffmann, J.H., Le Maitre, D.C., Palmer, A.R., Riggs, I., Shackleton, C.M. and Zimmermann, H.G. 2003. A proposed prioritization system for the management of invasive alien plants in South Africa: research in action. *South African Journal of Science,* 99(1): 37-43.

Roddick, J. 1991. The importance of the Solanaceae in medicine and drug therapy, 7-23. In Solanaceae III: taxonomy, chemistry, evolution, Hawkes, J.G., Lester, R.N., Nee, M and Estrada-R, N. (eds.), Kew Publishing.

Roleček, J., Chytrý, M., Hájek, M., Lvončík, S. and Tichý, L. 2007. Sampling design in large-scale vegetation studies: Do not sacrifice ecological thinking to statistical purism! *Folia Geobotanica*, 42: 199-208.

Ronnenberg, K., Hensen, I. and Wesche, K. 2011. Contrasting effects of precipitation and fertilization on seed viability and production of Stipa krylovii in Mongolia. *Basic and Applied Ecology*, 12(2); 141-151.

Roques, A., Rabitsch, W., Rasplus, J.Y., Lopez-Vaamonde, C., Nentwig, W. and Kenis, M. 2009. Alien terrestrial invertebrates of Europe, 63-79. In Handbook of alien species in Europe, Dordrecht: Springer Netherlands.

Roy, P.S. and Behera, M.D. 2005. Assessment of biological richness in different altitudinal zones in the Eastern Himalayas, Arunachal Pradesh, India. *Current science*, 88; 250-257.

Ruhi, A., Catford, J.A., Cross, W.F., Escoriza, D. and Olden, J.D. 2019. Understanding the nexus between hydrological alteration and biological invasions, 45-64. In Multiple stressors in river ecosystems, Sabater, S., Elosegi, A. and Ludwig R. (eds.). Amsterdam, the Netherland:Elsevier.

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

Ruwanza, S., Gaertner, M., Richardson, D.M. and Esler, K.J. 2013. Soil water repellency in riparian systems invaded by *Eucalyptus camaldulensis*: a restoration perspective from the Western Cape Province, South Africa. *Geoderma*, 200: 9-17.

Saha, D. and Sundriyal, R.C. 2010. Stand Structure, Phenology and Fruit yield Of *Illicium Griffithii* in western Arunachal Pradesh, Northeast India. *Indian Journal of Forestry*, 33(4), pp.475-488.

Saikia, P., Deka, J., Bharali, S., Kumar, A., Tripathi, O.P., Singha, L.B., Dayanandan, S. and Khan, M.L. 2017. Plant diversity patterns and conservation status of eastern Himalayan forests in Arunachal Pradesh, Northeast India. *Forest Ecosystems*, 4; 1-12.

Sakachep, Z.K. and Rai, P.K. 2021. Influence of invasive alien plants on vegetation of Hailakandi district, Assam, North-East, India. *Indian Journal of Ecology*, 48(1): 261-266.

Sakai, A.K., Allendorf, F.W., Holt, J.S., Lodge, D.M., Molofsky, J., With, K.A., Baughman, S., Cabin, R.J., Cohen, J.E., Ellstrand, N.C. and McCauley, D.E. 2001. The population biology of invasive species. *Annual Review Of Ecology And Systematics*, 32(1): 305-332.

Sala, O.E., Stuart Chapin, F.I.I.I., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A. and Leemans, R. 2000. Global biodiversity scenarios for the year 2100. *Science*, 287(5459): 1770-1774.

Sallabanks, R.E.X. 1993. Hierarchical mechanisms of fruit selection by an avian frugivore. *Ecology*, 74(5): 1326-1336.

Sanders, N.J., Gotelli, N.J., Heller, N.E. and Gordon, D.M. 2003. Community disassembly by an invasive species. *Proceedings of the National Academy of Sciences*, 100(5): 2474-2477.

Sandilyan, S. 2015. Highways of India: A heaven to invasive alien plants. Science India, 18(12): 33-36.

Sankara, R., Swamy, R.K., kumar, D., Singh, A.and Bhat, K.G. 2019. Flora of peninsular India. Indian Institute of Science. http://peninsula.ces.iisc.ernet.in. (Accesses on 25 December 2024).

Sankaran, K.V. 2015. Invasive species pest fact: *Mikania micrantha*. Asia-Pacific Forest Invasive Species Network. (Accessed 02.02.2024).

Santos, R.F., Nunes, B.M., Sá, R.D., Soares, L.A. and Randau, K.P. 2016. Morpho-anatomical study of *Ageratum conyzoides*. *Revista Brasileira de Farmacognosia*, 26: 679-687.

Sardans, J., Bartrons, M., Margalef, O., Gargallo-Garriga, A., Janssens, I.A., Ciais, P., Obersteiner, M., Sigurdsson, B.D., Chen, H.Y. and Peñuelas, J. 2017. Plant invasion is associated with higher plant–soil nutrient concentrations in nutrient-poor environments. *Global Change Biology*, 23(3): 1282-1291.

Sarma, A.K. and Bhattacharjya, D.K. 2016. Flowering and fruiting calendar of the Angiospermic weeds in and around Guwahati city of Assam, India. *Pleione*, 10(2); 333 - 343.

Sarmah, A., Haridasan, K. and Bisht, N.S. 2000. Development of medicinal plants as an economic venture in Arunachal Pradesh: Prospects and constraints. *Arunachal Forest News,* 18(1-2): 85-90.

Satyabrata, S.M., Ram Chandra, R.C., Geetha, K.A. and Kunal Mandal, K.M. 2000. Production technology of some important medicinal and aromatic crops developed under the all India coordinated research project. *Indian Journal of Arecanut, Spices, Medicinal Plants,* 2: 88 – 98.

Saunders, D.A., Hobbs, R.J. and Margules, C.R. 1991. Biological consequences of ecosystem fragmentation: a review. *Conservation Biology,* 5(1): 18-32.

Sax, D.F. 2002. Native and naturalized plant diversity are positively correlated in scrub communities of California and Chile. *Diversity and Distributions,* 8(4): 193-210.

Schippers, R.R. 2004. *Solanum torvum* SW. Vegetables. *Plant resources of Tropical Africa,* 2: 501-503.

Schirmel, J., Bundschuh, M., Entling, M.H., Kowarik, I. and Buchholz, S. 2016. Impacts of invasive plants on resident animals across ecosystems, taxa, and feeding types: a global assessment. *Global Change Biology,* 22(2): 594-603.

Schmitt, C.B., Burgess, N.D., Coad, L., Belokurov, A., Besançon, C., Boisrobert, L., Campbell, A., Fish, L., Gliddon, D., Humphries, K. and Kapos, V. 2009. Global analysis of the protection status of the world's forests. *Biological Conservation,* 142(10): 2122-2130.

Schnelle, F. 1955. Pflanzen-Phänologie. Probleme der Bioklimatologie 3. Leipzig: Akademische Verlagsgesellschaft Geest & Portig K.-G

Schwartz, M.D. 2003. Phenology: an integrative environmental science (Vol. 132). Dordrecht: Kluwer Academic Publishers.

Schwartz, M.D. 1998. Green-wave phenology. *Nature,* 394(6696): 839-840.

Scott, J.K. and Panetta, F.D. 1993. Predicting the Australian weed status of southern African plants. *Journal of Biogeography,* 20(1): 87-93.

Scott, N.A., Saggar, S. and McIntosh, P.D. 2001. Biogeochemical impact of *Hieracium* invasion in New Zealand's grazed tussock grasslands: sustainability implications. *Ecological Applications*, 11(5): 1311-1322.

Seebens, H., Bacher, S., Blackburn, T.M., Capinha, C., Dawson, W., Dullinger, S., Genovesi, P., Hulme, P.E., Van Kleunen, M., Kühn, I. and Jeschke, J.M. 2021. Projecting the continental accumulation of alien species through to 2050. *Global Change Biology,* 27(5): 970-982.

Seebens, H., Blackburn, T.M., Dyer, E.E., Genovesi, P., Hulme, P.E., Jeschke, J.M., Pagad, S., Pyšek, P., Winter, M., Arianoutsou, M. and Bacher, S. 2017. No saturation in the accumulation of alien species worldwide. *Nature communications*, 8(1): 14435.

Sekar, K.C. 2012. Invasive alien plants of Indian Himalayan region - diversity and implication. *American Journal of Plant Sciences,* 3(2): 177-184.

Selvan, K.M., Lyngdoh, S., Veeraswami, G.G. and Habib, B. 2013. An assessment of abundance, habitat use and activity patterns of three sympatric pheasants in an eastern Himalayan lowland tropical forest of Arunachal Pradesh, India. *Asian Journal of Conservation Biology*, 2(1): 52-60.

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

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

Shackleton, R.T., Witt, A.B., Nunda, W. and Richardson, D.M. 2017. *Chromolaena odorata* (Siam weed) in eastern Africa: distribution and socio-ecological impacts. *Biological Invasions,* 19(4): 1285-1298.

Shankar, U., Yadav, A.S., Rai, J.P.N. and Tripathi, R.S. 2012. Status of Alien Plant Invasions in the North-eastern Region of India. In Invasive Alien Plants an Ecological Appraisal for the Indian Subcontinent, Bhatt, J.R., Singh, J.S., Singh, S.P., Tripathi, R.S. and Kohli, R.K. (eds.), Wallingford UK: CABI.

Shannon, C.E and Wiener, W. 1963. The mathematical theory of communication. Urbana II: University of Illinois Press.

Sharma, B.D. and Gupta, I.C. 1989. Effect of trees cover on soil fertility in western Rajasthan. *Indian Forester*, 115(5): 348–354.

Sharma, G.P. and Raghubanshi, A.S. 2009. Lantana invasion alters soil nitrogen pools and processes in the tropical dry deciduous forest of India. *Applied Soil Ecology*, 42(2): 134-140.

Sharma, G.P. and Raghubanshi, A.S. 2012. Invasive species: ecology and impact of Lantana camara invasions, 19-42. In Invasive alien plants: an ecological appraisal for the Indian subcontinent, Bhatt, J.R., Singh, J.S., Singh, S.P., Tripathi, R.S. and Kohli, R.K. (eds.), Wallingford UK: CABI.

Sharma, P.D. 2005. Ecology and environment. Rastogi Publications, Meerat.

Sharp, R.L., Larson, L.R. and Green, G.T. 2011. Factors influencing public preferences for invasive alien species management. *Biological Conservation*, 144(8): 2097-2104.

Sheam, M., Haque, Z. and Nain, Z. 2020. Towards the antimicrobial, therapeutic and invasive properties of *Mikania micrantha* Knuth: a brief overview. *Journal of Advanced Biotechnology Experimnetal. Therapeutics*, 3(2): 92-101.

Shen, S., Xu, G., Clements, D.R., Jin, G., Liu, S., Yang, Y., Chen, A., Zhang, F. and Kato-Noguchi, H. 2016. Suppression of reproductive characteristics of the invasive plant *Mikania micrantha* by sweet potato competition. *BMC Ecology*, 16: 1-9.

Shen, Y., Liu, W., Baskin, J.M., Baskin, C.C. and Cao, M. 2006. Persistent soil seed banks of the globally significant invasive species, *Eupatorium adenophorum*, in Yunnan Province, south-western China. *Seed Science Research*, 16(2); 157-162.

Sheoran, P., Sardana, V., Chander, S., Kumar, A., Meena, M.D., Bali, A. and Sharma, P. 2018. Sulphur, boron and zinc nutrition to improve productivity, profitability and oil quality in sunflower (*Helianthus annuus*). *Indian Journal of Agricultural.Science*, 88: 1746-1754.

Sherry, R.A., Zhou, X., Gu, S., Arnone III, J.A., Schimel, D.S., Verburg, P.S., Wallace, L.L. and Luo, Y. 2007. Divergence of reproductive phenology under climate warming. *Proceedings of the National Academy of Sciences*, 104(1): 198-202.

Shiferaw, W., Bekele, T., Demissew, S. and Aynekulu, E., 2020. Phenology of the alien invasive plant species *Prosopis juliflora* in arid and semi-arid areas in response to climate variability and some perspectives for its control in Ethiopia. *Polish Journal of Ecology*, 68(1): 37-46.

Shiferaw, W., Demissew, S., Bekele, T., Aynekulu, E. and Pitroff, W. 2021. Invasion of *Prosopis juliflora* and its effects on soil physicochemical properties in Afar region, Northeast Ethiopia. *International Soil and Water Conservation Research*, 9(4): 631-638.

Shivers, S.D., Golladay, S.W., Waters, M.N., Wilde, S.B., Marzolf, N.S. and Covich, A.P. 2023. Invasive species interactions affect nutrient cycling in a shallow reservoir: a mesocosm experiment. *Lake and Reservoir Management*, 39(4): 278-290.

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

Shukla, A.K., Behera, S.K., Tripathi, R., Prakash, C., Nayak, A.K., Kumar, P.S., Chitdeshwari, T., Kumar, D., Nayak, R.K., Babu, P.S. and Katkar, R.N., 2021. Evaluation of spatial spreading of phyto-available sulphur and micronutrients in cultivated coastal soils. *PLoS One*, 16(10); e0258166.

Shukla, R.P. and Ramakrishnan, P.S. 1982. Phenology of trees in a sub-tropical humid forest in north-eastern India. *Vegetatio*, 49(2): 103-109.

Silva, F.L., Fischer, D.C.H., Fechine Tavares, J., Sobral Silva, M., Filgueiras de Athayde-Filho, P. and Barbosa-Filho, J.M. 2011. Compilation of secondary metabolites from *Bidens pilosa* L. *Molecules*, 16(2): 1070-1102.

Simba, Y.R., Kamweya, A.M., Mwangi, P.N. and Ochora, J.M. 2013. Impact of the invasive shrub, *Lantana camara* L. on soil properties in Nairobi National Park, Kenya. *International Journal of Biodiversity and Conservation*, 5(12): 803–809.

Simberloff, D. and Von Holle, B. 1999. Positive interactions of nonindigenous species: invasional meltdown? *Biological Invasions,* 1: 21-32.

Simberloff, D. 1996. Impacts of introduced species in the United States. Consequences, 2(2): 13-22.

Simberloff, D. 2009. The role of propagule pressure in biological invasions. *Annual Review of Ecology, Evolution, and Systematics,* 40: 81-102.

Simberloff, D., Martin, J.L., Genovesi, P., Maris, V., Wardle, D.A., Aronson, J., Courchamp, F., Galil, B., García-Berthou, E., Pascal, M. and Pyšek, P. 2013. Impacts of biological invasions: what's what and the way forward. *Trends in Ecology & Evolution*, 28(1): 58-66.

Singh, C., Gupta, M, Gupta, A.K. and Sharma, M. 2020. *Triumfetta rhomboidea* Jacq.-An overview. *International Journal of Current Pharmaceutical Review and Research*, 12(1): 01-08.

Singh, H.B., Kumar, A. and Adhikari, B.S. 2021. Status of Invasive Alien Plant Species (IAPs) In Arunachal Pradesh, India: A Review. *I J R B A T*, Issue (Special-17); 604-613.

Singh, K.P. and Kushwaha, C.P. 2006. Diversity of flowering and fruiting phenology of trees in a tropical deciduous forest in India. *Annals of botany,* 97(2): 265-276.

Singh, P. 1991. Avian and mammalian evidences in Pakhui Wildlife Sanctuary in East Kameng district, Arunachal Pradesh. *Arunachal Forest News*, 9(2): 1-10.

Singh, P. 1994. Recent bird records from Arunachal Pradesh. *Forktail*, 10: 65-104.

Singh, S.P. 2004. Some success stories in classical biological control of agricultural pests in India. Asia-Pacific Association of Agricultural Research Institutions, FAO Regional Office for Asia and Pacific, Bangkok, Thailand, +73 pp.

Singh, S.P., Khanna, K.R. and Sudhir, S. 1998. Breeding of *Solanum viarum*: current status as steroid bearing plant. *Journal of Medicinal and Aromatic Plant Sciences*, 20(2): 423-431.

Singh, T.P., Singh, S., Roy, P.S. and Rao, B.S.P. 2002. Vegetation mapping and characterization in West Siang District of Arunachal Pradesh, India–a satellite remote sensing-based approach. *Current science*,83(10): 1221-1230.

Singh, R., Rawat, M., Chand, T., Tripathi, S.K. and Pandey, R., 2023. Phenological variations in relation to climatic variables of moist temperate forest tree species of western Himalaya, India. Heliyon, 9(6).

Singha, H.R., Sinha, S. and Sinha, R.K. 2016. Karyomorphology of *Solanum viarum* Dunal–an ethnomedicinal species of Tripura. *Vegetos*, 29: 65-68.

Sirinthipaporn, A. and Jiraungkoorskul, W. 2017. Wound healing property review of siam weed, *Chromolaena odorata*. *Pharmacognosy Reviews*, 11(21): 35.

Sizer, N. and Tanner, E.V. 1999. Responses of woody plant seedlings to edge formation in a lowland tropical rainforest, Amazonia. *Biological Conservation*, 91(2-3): 135-142.

Smith, J.G., Sconiers, W., Spasojevic, M.J., Ashton, I.W. and Suding, K.N. 2012. Phenological changes in alpine plants in response to increased snowpack, temperature, and nitrogen. *Arctic, Antarctic, and Alpine Research*, 44(1); 135-142.

Smythe, N. 1970. Relationships between fruiting seasons and seed dispersal methods in a neotropical forest. *The American Naturalist*, 104(935); 25-35.

Sobrinho, M.S., Tabatinga, G.M., Machado, I.C., and Lopes, A.V. 2013. Reproductive phenological pattern of Calotropis procera (Apocynaceae), an invasive species in Brazil: annual in native areas; continuous in invaded areas of caatinga, *Acta Bot. Brasil.*, 27, pp. 456–459.

Sondhi, S. and Kunte, K. 2014. Butterflies and Moths of Pakke Tiger Reserve. Samrakshan Trust.

Song, Z., Fu, Y.H., Du, Y., Li, L., Ouyang, X., Ye, W. and Huang, Z. 2020. Flowering phenology of a widespread perennial herb shows contrasting responses to global warming between humid and non-humid regions. *Functional Ecology*, 34(9): 1870-1881.

Soti, P., Purcell, M.F. and Jayachandran, K. 2019. Data on foliar nutrient concentration of invasive plants in the recipient habitat and their native habitat. *Data in brief*, 25.

Soti, P.G. and Jayachandran, K. 2016. Effect of exotic invasive old world climbing fern (*lygodium microphyllum*) on soil properties. *Journal of Soil Science and Plant Nutrition*, 16(4): 930-940.

Soti, P.G., Purcell, M. and Jayachandran, K. 2020. Soil biotic and abiotic conditions negate invasive species performance in native habitat. *Ecological Processes*, 9(1): 1-9.

Sousa, W.P. 1984. The role of disturbance in natural communities. *Annual Review of Ecology and Systematics*, 15(1): 353-391.

Souza, I.M. and Funch, L.S. 2017. Synchronization of leafing and reproductive phenological events in *Hymenaea* L. species (Leguminosae, Caesalpinioideae): the role of photoperiod as the trigger. *Brazilian Journal of Botany*, 40; 125-136.

Sparks, T.H., Menzel, A. and Stenseth, N.C. 2009. European cooperation in plant phenology. *Climate Research*, 39(3): 175-177.

Stamets, P., Growing gourmet and medicinal mushrooms, Ten Speed Press, 2011.

Stanek, M., Piechnik, Ł. and Stefanowicz, A.M. 2020. Invasive red oak (*Quercus rubra* L.) modifies soil physicochemical properties and forest understory vegetation. *Forest Ecology and Management,* 472: 118253.

Stanley, M.C., Ifeanyi, O.E., Nwakaego, C.C. and Esther, I.O., 2014. Antimicrobial effects of Chromolaena odorata on some human pathogens. *International Journal of Current Microbiology and Applied Sciences*, 3(3): 1006-1012.

Stefanowicz, A.M., Majewska, M.L., Stanek, M., Nobis, M. and Zubek, S. 2018. Differential influence of four invasive plant species on soil physicochemical properties in a pot experiment. *Journal of Soils and Sediments*, 18: 1409-1423.

Stephenson, A.G. 1981. Flower and fruit abortion: proximate causes and ultimate functions. *Annual Review of Ecology And Systematics*, 12; 253-279.

Strauss, S.Y., Lau, J.A. and Carroll, S.P. 2006. Evolutionary responses of natives to introduced species: what do introductions tell us about natural communities? Ecology letters, 9(3): 357-374.

Subbiah, B.V. and Asija, G.L. 1956. A rapid procedure for the estimation of available nitrogen in soils. *Current Science* 25: 259-260.

Suding, K.N., Stanley Harpole, W., Fukami, T., Kulmatiski, A., MacDougall, A.S., Stein, C. and van der Putten, W.H. 2013. Consequences of plant–soil feedbacks in invasion. *Journal of Ecology*, 101(2): 298-308.

Suksamrarn, A., Chotipong, A., Suavansri, T., Boongird, S., Timsuksai, P., Vimuttipong, S. and Chuaynugul, A. 2004. Antimycobacterial activity and cytotoxicity of flavonoids from the flowers of *Chromolaena odorata*. *Archives of Pharmacal Research*, 27: 507-511.

Sun, Y., Müller-Schärer, H. and Schaffner, U. 2014. Plant neighbours rather than soil biota determine impact of an alien plant invader. *Functional Ecology*, 28(6): 1545-1555.

Swacha, G., Botta-Dukát, Z., Kacki, Z., Pruchniewicz, D. and Zolnierz, L. 2017. A performance comparison of sampling methods in the assessment of species composition patterns and environment–vegetation relationships in species-rich grasslands. *Acta Societatis Botanicorum Poloniae,* 86(4).

Swaine, M.D., Agyeman, V.K., Kyere, B., Orgle, T.K., Thomson, J. and Veenendaal, E.M. 1997. Ecology of Forest Trees in Ghana. ODA Forestry Series No. 7, London, +64pp.

Tag, H and Das, A.K. 2004. Ethnobotanical notes on Hill Miri Tribe of Arunachal Pradesh. *Indian Journal of Traditional Knowledge*, 3: 80-85.

Tag, H., Kalita, P., Singh, R.K. and Das, A.K. 2021. Biocultural Resources and Traditional Food Systems of Nyishi Tribe of Arunachal Pradesh (India): An Empirical Learning on the Role of Mythology and Folklore in Conservation, 155-198. In Social-Ecological Diversity and Traditional Food Systems, Singh, R.K., Turner, N., Reyes-Garcia, V and Pretty, J. (eds.), CRC Press.

Tang, Y., Zhou, W. and Du, Y. 2023. Effects of temperature, precipitation, and CO2 on plant phenology in China: a circular regression approach. *Forests*, 14(9): 1844.

Tasrif, A., Sahid, I.B., Sastroutomo, S.S. and Latiff, A. 1991. Purity study of imported leguminous cover crops. *Plant Protection Quarterly*, 6(4): 190-193.

Taylor, S., Kumar, L., Reid, N. and Kriticos, D.J. 2012. Climate change and the potential distribution of an invasive shrub, *Lantana camara* L. *Plos One*, 7(4): e35565.

Taylor, G., Tallis, M.J., Giardina, C.P., Percy, K.E., Miglietta, F., Gupta, P.S., Gioli, B., Calfapietra, C., Gielen, B., Kubiske, M.E., and Scarascia-Mugnozza, G.E. 2008. Future atmospheric CO_2 leads to delayed autumnal senescence, *Glob. Change Biol.*, 14(2), pp. 264–275.

Te Beest, M., Esler, K.J. and Richardson, D.M. 2015. Linking functional traits to impacts of invasive plant species: a case study. *Plant Ecology*, 216: 293-305.

Tekiela, D.R. and Barney, J.N. 2015. System-level changes following invasion caused by disruption of functional relationships among plant and soil properties. *Ecosphere*, 6(12): 1-16.

Tekiela, D.R. and Barney, J.N. 2017. Invasion shadows: The accumulation and loss of ecological impacts from an invasive plant. Invasive *Plant Science and Management*, 10(1): 1-8.

Ter Braak, C.J. 1986. Canonical correspondence analysis: a new eigenvector technique for multivariate direct gradient analysis. *Ecology*, 67(5): 1167-1179.

Ter Braak, C.J. 1987. The analysis of vegetation-environment relationships by canonical correspondence analysis. *Vegetatio*, 69: 69-77.

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

Terra, J.A., Shaw, J.N., Reeves, D.W., Raper, R.L., Van Santen, E. and Mask, P.L. 2004. Soil carbon relationships with terrain attributes, electrical conductivity, and a soil survey in a coastal plain landscape. *Soil Science*, 169(12): 819-831.

Tesfaye, G.T., Demel Teketay, D.T., Masresha Fetene, M.F. and Beck, E. 2011. Phenology of seven indigenous tree species in a dry Afromontane forest, southern Ethiopia. *Trpoical Ecology*, 52(3): 229-241.

Testoni, S., Dawson, L., Melo, V., Lopes-Mazzetto, J., Ramalho, B. and Salvador, F. 2022. Soil colour and plant-wax markers: Application in forensic investigations under urban subtropical environments. *Forensic Sciences*, 2(1): 57-71.

Thackeray, S.J., Henrys, P.A., Hemming, D., Bell, J.R., Botham, M.S., Burthe, S., Helaouet, P., Johns, D.G., Jones, I.D., Leech, D.I. and Mackay, E.B. 2016. Phenological sensitivity to climate across taxa and trophic levels. *Nature*, 535(7611): 241-245.

Thapa, C.B. and Baburam, K.C. 2023. Phenology of Major Weeds in Paddy Field of Pokhara, Nepal. *Himalayan Biodiversity*, 9: 58-62.

Thapa, S., Chitale, V., Rijal, S.J., Bisht, N. and Shrestha, B.B. 2018. Understanding the dynamics in distribution of invasive alien plant species under predicted climate change in Western Himalaya. *PloS one*, 13(4): e0195752.

Theoharides, K.A. and Dukes, J.S. 2007. Plant invasion across space and time: factors affecting nonindigenous species success during four stages of invasion. *New Phytologist*, 176(2): 256-273.

Thiele, J., Isermann, M., Otte, A. and Kollmann, J. 2010. Competitive displacement or biotic resistance? Disentangling relationships between community diversity and invasion success of tall herbs and shrubs. *Journal of Vegetation Science*, 21(2): 213-220.

Thompson, T.E. 1983. Genetic variability in pecan fruit development. *HortScience*, 18(6): 955-957.

Thuiller, W., Richardson, D.M. and Midgley, G.F., 2007. Will Climate Change Promote Alien Plant Invasions?. *Ecological Studies*,193: 197.

Thuiller, W., Richardson, D.M., Pyšek, P., Midgley, G.F., Hughes, G.O. and Rouget, M. 2005. Niche-based modelling as a tool for predicting the risk of alien plant invasions at a global scale. *Global Change Biology*, 11(12); 2234-2250.

Tiwari, R.K.S. and Chandra, K.K., 2017. Flowering and Fruiting Dynamics of Medicinal Plants in Relation to Climatic Conditions. International Journal of Ecology and Environmental Sciences, 43(1), pp.25-34.

Timbilla, J.A. 1996, October. Effect of biological control of *Chromolaena odorata* on biodiversity: a case study in the Ashanti region of Ghana, 97-101. In Proceedings of the fourth international workshop on the biological control and management of *Chromolaena odorata*,

Timsina, B., Shrestha, B.B., Rokaya, M.B. and Münzbergová, Z. 2011. Impact of *Parthenium hysterophorus* L. invasion on plant species composition and soil properties of grassland communities in Nepal. *Flora-Morphology, Distribution, Functional Ecology of Plants*, 206(3): 233-240.

Tondoh, J.E., Koné, A.W., N'Dri, J.K., Tamene, L. and Brunet, D. 2013. Changes in soil quality after subsequent establishment of *Chromolaena odorata* fallows in humid savannahs, Ivory Coast. *Catena*, 101: 99-107.

Tripathi, R.S. 2009. Alien plant invasion: A hot ecological issue. *International Society of Environmental Botanist*, 15(3).

Tripathi, R.S., Khan, M.L. and Yadav, A.S. 2012. Biology of *Mikania micrantha* HBK: a Review, 9-107. In Invasive alien plants: An ecological appraisal for the Indian subcontinent, Bhatt, J.R., Singh, J.S., Singh, S.P., Tripathi, R.S. and Kohli, R.K. (eds.), Wallingford UK: CABI.

Tripathi, R.S., Kushwaha, S.P.S. and Yadav, A.S. 2006. Ecology of three invasive species of *Eupatorium*: A review. *International Journal of Ecology and Environmental Sciences*, 32(4): 301-326.

Trunschke, J. and Stöcklin, J. 2017. Plasticity of flower longevity in alpine plants is increased in populations from high elevation compared to low elevation populations. *Alpine Botany*, 127: 41-51

Tsuji, H. 2017. Molecular function of florigen, *Breed. Sci*., 67(4), pp. 327–332.

Tukey, L.D. 1952. Effect of night temperature on growth of the fruit of the sour cherry. *Botanical Gazette*, 114(2): 155-165.

Tun, S..S. 2020. Study on Morphology, Histology of Leaves and Preliminary Phytochemical Test of *Synedrella nodiflora* (L.) Gaertn. (Bizat-Hpo). *3rd Myanmar Korea Conference Research Journa*l,3(2); 729-736.

Turner, M.G., 2005. Landscape ecology: what is the state of the science. *Annual Review of Ecology, Evolution, and Systematics*, 36: 319-344.

Ullah, M.S., Farooque, A.A., Javed, Q., Ullah, I., Bo, Y., Jabran, K. and Sun, J. 2024. Under Biological Invasion: Impacts of litter decomposition mediated by invasive plant species on soil nutrients and functional growth traits of both invasive and native plant species. *Russian Journal of Ecology*, 55(2): 89-100.

Ullah, R., Khan, N. and Ali, K. 2022. Which factor explains the life-history of *Xanthium strumarium* L., an aggressive alien invasive plant species, along its altitudinal gradient?. *Plant Direct*, 6(1): e375.

Usharani, B. and Raju, A.S., 2018. Pollination ecology of *Synedrella nodiflora* (l.) Gaertn.(Asteraceae). *Journal of Threatened Taxa*, 10(11): 12538-12551.

UTPD, 2014. Useful Tropical Plants Database: Urena lobata. https://tropical.theferns.info/viewtropical.php?id=Urena+lobata. (Accessed on 05.09.2024).

Uyi, O.O., Ekhator, F., Ikuenobe, C.E., Borokini, T.I., Aigbokhan, E.I., Egbon, I.N., Adebayo, A.R., Igbinosa, I.B., Okeke, C.O., Igbinosa, E.O. and Omokhua, G.A., 2014. *Chromolaena odorata* invasion in Nigeria: A case for coordinated biological control. *Management of Biological Invasions*, 5(4): 377.

Valkenburg, J.V. and Bunyapraphatsara, N. 2001. Plant resources of South-East Asia. In Medicinal and Poisonous plants, Valkenburg, JLCH, van and Bunyapraphatsara, N. (eds.), Leiden, Netherlands: Backhuys publishers. 782 pp.

Von Holle, B., Wei, Y. and Nickerson, D., 2010. Climatic variability leads to later seasonal flowering of Floridian plants. PLoS One, 5(7), p.e11500.

Van der Pijl, L. 1972. Principles of Dispersal in Higher Plants. Berlin: Springer-Verlag.

Van der Putten, W.H., Kowalchuk, G.A., Brinkman, E.P., Doodeman, G.T.A., Van der Kaaij, R.M., Kamp, A.F.D., Menting, F.B.J. and Veenendaal, E.M. 2007. Soil feedback of exotic savanna grass relates to pathogen absence and mycorrhizal selectivity. *Ecology*, 88(4): 978-988.

Van Kleunen, M., Weber, E. and Fischer, M., 2010. A meta-analysis of trait differences between invasive and non-invasive plant species. *Ecology Letters*, 13: 235-245.

Van Schaik, C.P., Terborgh, J.W. and Wright, S.J. 1993. The phenology of tropical forests: adaptive significance and consequences for primary consumers. *Annual Review of ecology and Systematics*, 24(1); 353-377.

van Wilgen B.W and Wilson J.R. 2018. The status of biological invasions and their management in South Africa in 2017. South African National Biodiversity Institute, Kirstenbosch and DST-NRF Centre of Excellence for Invasion Biology, Stellenbosch.

Vanderhoeven, S., Dassonville, N. and Meerts, P. 2005. Increased topsoil mineral nutrient concentrations under exotic invasive plants in Belgium. *Plant and Soil*, 275: 169-179.

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

Velho, N. and Krishnadas, M. 2011. Post-logging recovery of animal-dispersed trees in a tropical forest site in north-east India. *Tropical Conservation Science*, 4(4): 405-419.

Verheyen, K., Bossuyt, B., Honnay, O. and Hermy, M., 2003. Herbaceous plant community structure of ancient and recent forests in two contrasting forest types. *Basic and Applied Ecology*, 4(6): 537-546.

Verma, A.K., Nayak, R., Manika, N., Bargali, K., Pandey, V.N., Chaudhary, L.B. and Behera, S.K. 2023. Monitoring the distribution pattern and invasion status of *Ageratina adenophora* across elevational gradients in Sikkim Himalaya, India. *Environmental Monitoring and Assessment*, 195(1): 152.

Verma, G., Sharma, R.P., Sharma, S.P., Subehia, S.K. and Shambhavi, S., 2012. Changes in soil fertility status of maize-wheat system due to long-term use of chemical fertilizers and amendments in an alfisol. *Plant, Soil and Environment*, 58(12): 529-533.

Verma, S and Joshi, S.P. 2021. Floristic of a riverine ecosystem along Siwaliks, Dehradun, India. *IJESR*, 9(11): 1-12.

Verstraeten, G., Baeten, L., Van den Broeck, T., De Frenne, P., Demey, A., Tack, W., Muys, B. and Verheyen, K. 2013. Temporal changes in forest plant communities at different site types. *Applied Vegetation Science*, 16(2): 237-247.

Vestergard, M., Rønn, R. and Ekelund, F. 2015. Above–belowground interactions govern the course and impact of biological invasions. *AoB Plants*, 7: plv025.

Vilà, M. and Hulme, P.E. 2017. Impact of biological invasions on ecosystem services (Vol. 12). Cham, Switzerland: Springer International Publishing.

Vilà, M., Basnou, C., Pyšek, P., Josefsson, M., Genovesi, P., Gollasch, S., Nentwig, W., Olenin, S., Roques, A., Roy, D. and Hulme, P.E. 2010. How well do we understand the impacts of alien species on ecosystem services? A pan-European, cross-taxa assessment. *Frontiers in Ecology and the Environment*, 8(3): 135-144.

Vilà, M., Espinar, J.L., Hejda, M., Hulme, P.E., Jarošík, V., Maron, J.L., Pergl, J., Schaffner, U., Sun, Y. and Pyšek, P. 2011. Ecological impacts of invasive alien plants: a meta-analysis of their effects on species, communities and ecosystems. *Ecology letters*, 14(7): 702-708.

Vilà, M., Pino, J. and Font, X. 2007. Regional assessment of plant invasions across different habitat types. *Journal of Vegetation Science*, 18(1): 35-42.

Vila, M., Trillo, A., Castro-Díez, P., Gallardo, B. and Bacher, S. 2024. Field studies of the ecological impacts of invasive plants in Europe. *NeoBiota*, 90: 139–159.

Vitasse, Y., Porté, A.J., Kremer, A., Michalet, R. and Delzon, S., 2009. Responses of canopy duration to temperature changes in four temperate tree species: relative contributions of spring and autumn leaf phenology. *Oecologia*, 161(1), pp.187-198.

Vitousek, P.M. and Walker, L.R. 1989. Biological invasion by Myrica faya in Hawai'i: plant demography, nitrogen fixation, ecosystem effects. *Ecological monographs*, 59(3): 247-265.

Vitousek, P.M. 1990. Biological invasions and ecosystem processes: towards an integration of population biology and ecosystem studies, 183-191. In Ecosystem Management, Samson, F.B and Knopf, F.L (eds.), Springer-Verlag new York.

Vitousek, P.M. 1994. Beyond global warming: ecology and global change. *Ecology*, 75(7): 1861-1876.

Vitousek, P.M., D'Antonio, C.M., Loope, L.L. and Westbrooks, R. 1996. Biological invasions as global environmental change. *American scientist*, 84(5): 468-478.

Vitousek, P.M., D'antonio, C.M., Loope, L.L., Rejmanek, M. and Westbrooks, R. 1997. Introduced species: a significant component of human-caused global change. *New Zealand Journal of Ecology*, 21(1); 1-16.

Volin, J.C., Kruger, E.L., Volin, V.C., Tobin, M.F. and Kitajima, K. 2010. Does release from natural belowground enemies help explain the invasiveness of *Lygodium microphyllum*? A cross-continental comparison. *Plant Ecology*, 208: 223-234.

Volis, S. 2007. Correlated patterns of variation in phenology and seed production in populations of two annual grasses along an aridity gradient. Evolutionary *Ecology*, 21: 381-393.

Waggy, M.A. 2009. *Solanum viarum*. Fire Effects Information System,[Online]., USA: US Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory.

Wagh, V.V. and Jain, A.K. 2018. Status of ethnobotanical invasive plants in western Madhya Pradesh, India. *South African Journal of Botany*, 114: 171-180.

Wagner, V., Chytrý, M., Jiménez-Alfaro, B., Pergl, J., Hennekens, S., Biurrun, I., Knollová, I., Berg, C., Vassilev, K., Rodwell, J.S. and Škvorc, Ž. 2017. Alien plant invasions in European woodlands. *Diversity and distributions*, 23(9): 969-981.

Wagner, W.L., Herbst, D.R. and Sohmer, S.H. 1999. Manual of the flowering plants of Hawaii: University of Hawaii Press.

Wahlsteen, E. and Borah, D. 2022. A checklist with three new species and two new country records of Begonia from Dibang Valley, Arunachal Pradesh, India. *Brittonia*, 74(4): 403-418.

Wahyuni, I., Sulistijorini, S., Setiabudi, S., Meijide, A., Nomura, M., Kreft, H., Rembold, K., Tjitrosoedirdjo, S.S. and Tjitrosoedirdjo, S. 2016. Distribution of Invasive Plant Species in Different Land-Use Systems in Sumatera, Indonesia. *Biotropia*, 23(2): 127-135.

Walker, L.R. and Vitousek, P.M. 1991. An invader alters germination and growth of native dominant tree in Hawai'i. *Ecology*, 72(4): 1449-1455.

Walkley, A. and Black, I.A. 1934. An examination of the Degtjareff method for determining soil organic matter, and a proposed modification of the chromic acid titration method. *Soil Science*, 37(1): 29-38.

Walther, G.R., Roques, A., Hulme, P.E., Sykes, M.T., Pyšek, P., Kühn, I., Zobel, M., Bacher, S., Botta-Dukát, Z., Bugmann, H. and Czucz, B. 2009. Alien species in a warmer world: risks and opportunities. *Trends in Ecology & Evolution*, 24(12): 686-693.

Wang, B.S., Wang, Y.J., Liao, W.B., Zhan, Q.J., Li, M.G. and Peng, S.L. 2004. The Invasion Ecology and Management of Alien Weed *Mikania micrantha* H.B.K. Science Press, Beijing, China:+1-177pp.

Wang, X., Wang, X., Wang, W., Wang, J. and Yu, F. 2022. Effects of invasive plant diversity on soil microbial communities. *Diversity*, 14(11): 992.

Wang, Y., Wang, K.L., Zhou, D.S., Li, L. and Chen, Z.H. 2007. Effects of vegetation succession on soil quality in karst region of Guangxi. China. *Journal of Soil, Water and Conservation*, 21(6): 130-134.

Wangpan, T., Tasar, J., Taka, T., Giba, J., Tesia, P. and Tangjang, S. 2019. Traditional use of plants as medicine and poison by Tagin and Galo Tribe of Arunachal Pradesh. *Journal of Applied Pharmaceutical Science,* 9(9): 98-104.

Wardle, D.A., Bardgett, R.D., Klironomos, J.N., Setala, H., Van Der Putten, W.H. and Wall, D.H. 2004. Ecological linkages between aboveground and belowground biota. *Science,* 304(5677): 1629-1633.

Waring, R.H. and Schlesinger, W.H. 1985. Forest ecosystems. *Analysis at multiples scales*, 55.

Waser, A.M., Splinter, W. and Van der Meer, J. 2015. Indirect effects of invasive species affecting the population structure of an ecosystem engineer. *Ecosphere*, 6(7): 1-12.

Waterhouse, D.F. 1997. The major invertebrate pests and weeds of agriculture and plantation forestry in the southern and western Pacific. AClAR *Monograph* No. 44:99.

Waterhouse, D.F. 1993. The major arthropod pests and weeds of agriculture in Southeast Asia: distribution, importance and origin. *Monograph*, 21: 144.

Weber, E. 1998. The dynamics of plant invasions: a case study of three exotic goldenrod species (*Solidago L.)* in Europe. *Journal of Biogeography*, 25(1): 147-154.

Wei, H., Xu, J., Quan, G., Zhang, J. and Qin, Z. 2017. Invasion effects of *Chromolaena odorata* on soil carbon and nitrogen fractions in a tropical savanna. Ecosphere, 8(5): e01831.

Wei, W.H. and Jing, Z. 1990. Effect of particle size on transition metal concentrations in the Changjiang (Yangtze River) and the Huanghe (Yellow River), China. *Science of The Total Environment*, 94(3): 187-207.

Weidenhamer, J.D. and Callaway, R.M. 2010. Direct and indirect effects of invasive plants on soil chemistry and ecosystem function. *Journal of chemical ecology*, 36: 59-69.

Wells, M.J., Balsinhad, V.M., Joffe, H., Engelbrecht, V.M., Harding, G. and Stirton, C.H. 1986. A Catalogue of Problem Plants in Southern Africa, incorporating The National Weed List of South Africa. Memoirs of the Botanical Survey of South Africa, No. 53. Botanical Research Institute, Pretoria, South Africa, +658pp.

Welman, W.G. 2003. The genus *Solanum* (Solanaceae) in southern Africa: subgenus *Leptostemonu*m, the introduced sections Acanthophora and Torva. *Bothalia*, 33(1): 1-18.

Wender, B.W., Harrington, C.A. and Tappeiner, J.C. 2004. Flower and fruit production of understory shrubs in western Washington and Oregon. *Northwest Science*, 78(2): 124-140.

Werner, C., Zumkier, U., Beyschlag, W. and Máguas, C. 2010. High competitiveness of a resource demanding invasive acacia under low resource supply. *Plant Ecology*, 206: 83-96.

Whitehouse, C, Cheek, M, Andrews, SM, Verdcourt, B. 2001. Tiliaceae and Muntingiaceae. In: *Flora of Tropical East Africa*, Beentje, H.J. and Smith, S.A.L. (eds.) Royal Botanical Garden, Kew, Rotterdam.

Whitford, P.B. 1949. Distribution of woodland plants in relation to succession and clonal growth. *Ecology*, 30(2): 199-208.

Widmann, K., Gebauer, G., Rehder, H. and Ziegler, H. 1990. Biomass production and nitrogen contents of the CAM plants *Kalanchoe daigremontiana* and *K. tubiflora* in cultures with different nitrogen and water supply. *Oecologia*, 82: 478-483.

Wijebandara, D.M.D.I., Dasog, G.S., Patil, P.L. and Hebbar, M. 2011. Zinc fractions and their relationships with soil properties in paddy-growing soils of northern dry and hill zones of Karnataka. *Journal of the Indian Society of Soil Science*, 59(2): 141-147.

Williams, C.H. and Steinbergs, A. 1959. Soil sulphur fractions as chemical indices of available sulphur in some Australian soils. *Australian Journal of Agricultural Research*, 10(3): 340-352.

Williams, T.A. and Abberton, M.T., 2004. Earlier flowering between 1962 and 2002 in agricultural varieties of white clover. *Oecologia*, 138(1), pp.122-126.

Williams, F., Eschen, R., Harris, A., Djeddour, D., Pratt, C., Shaw, R.S., Varia, S., Lamontagne-Godwin, J., Thomas, S.E. and Murphy, S.T. 2010. The economic cost of invasive non-native species on Great Britain. *CABI Proj No VM10066*: 1-99.

Williams, T.A. and Abberton, M.T. 2004. Earlier flowering between 1962 and 2002 in agricultural varieties of white clover. *Oecologia*, 138: 122-126.

Williamson, M. 1999. Invasions. *Ecography*, 22(1): 5-12.

Williamson, M.H. and Fitter, A. 1996. The characters of successful invaders. *Biological conservation*, 78(1-2): 163-170.

Willis, C.G., Ruhfel, B.R., Primack, R.B., Miller-Rushing, A.J., Losos, J.B. and Davis, C.C. 2010. Favorable climate change response explains non-native species' success in Thoreau's woods. PloS one, 5(1): e8878.

Wilsey, B.J. and Wayne Polley, H. 2006. Aboveground productivity and root–shoot allocation differ between native and introduced grass species. *Oecologia*, 150: 300-309.

Wilsey, B.J., Barber, K. and Martin, L.M. 2015. Exotic grassland species have stronger priority effects than natives regardless of whether they are cultivated or wild genotypes. *New Phytologist*, 205(2): 928-937.

Williams-Linera, G., Tolome, J., and Forest, L.M. 1996. Litterfall, temperate and tropical dominant trees, and climate in a Mexican lower montane forest, *Biotropica*, 28(4), pp. 649–656.

Williams-Linera, G. 1997. Phenology of deciduous and broadleaved-evergreen tree species in a Mexican tropical lower montane forest, *Glob. Ecol. Biogeogr. Lett.*, 6(3), pp. 115–127.

Win, S.S. and Hnin, N.N. 2022. Taxonomic Study on Some species of Family Asteraceae found in Thone Taung Village, Pyin Oo Lwin Township, Mandalay Region, *Yadanabon University Research Journal*, 12(2): 350-359.

Windham, L. and Lathrop, R.G. 1999. Effects of *Phragmites australis* (common reed) invasion on aboveground biomass and soil properties in brackish tidal marsh of the Mullica River, New Jersey. *Estuaries*, 22: 927-935.

Wolfe, B.E. and Klironomos, J.N. 2005. Breaking new ground: soil communities and exotic plant invasion. *Bioscience*, 55(6): 477-487.

Wolkovich, E.M. and Cleland, E.E. 2011. The phenology of plant invasions: a community ecology perspective. *Frontiers in Ecology and the Environment*, 9(5): 287-294.

Wolkovich, E.M., Cook, B.I. and Davies, T.J. 2014. Progress towards an interdisciplinary science of plant phenology: building predictions across space, time and species diversity. *New Phytologist*, 201(4): 1156-1162.

Wolkovich, E.M., Davies, T.J., Schaefer, H., Cleland, E.E., Cook, B.I., Travers, S.E., Willis, C.G. and Davis, C.C. 2013. Temperature-dependent shifts in phenology contribute to the success of exotic species with climate change. *American Journal of Botany*, 100(7): 1407-1421.

Wright, S.J. and Calderon, O. 1995. Phylogenetic patterns among tropical flowering phenologies. *Journal of Ecology,* 83: 937-948.

Wu, S.H., Hsieh, C.F. and Rejmánek, M. 2004. Catalogue of the Naturalized Flora of Taiwan. *Taiwania*, 49(1): 16-31.

Xie, Y., Huang, H., Xie, X., Ou, J., Chen, Z., Lu, X., Kong, D., Nong, L., Lin, M., Qian, Z. and Mao, Y. 2024. Landscape, human disturbance, and climate factors drive the species richness of alien invasive plants on subtropical islands. *Plants*, 13(17): 2437.

Xu, C.Y., Griffin, K.L. and Schuster, W.S.F. 2007. Leaf phenology and seasonal variation of photosynthesis of invasive *Berberis thunbergii* (Japanese barberry) and two co-occurring native understory shrubs in a northeastern United States deciduous forest. *Oecologia*, 154: 11-21.

Xu, H., Liu, Q., Wang, S., Yang, G. and Xue, S. 2022. A global meta-analysis of the impacts of exotic plant species invasion on plant diversity and soil properties. *Science of the Total Environment, 8*10: 152286

Xu, H.G. and Ding, H. 2003. Countermeasures for the prevention of invasive alien species. Conserving Biodiversity and Strengthening Nature, 128-139. In Conserving Biodiversity and Strengthening Nature Reserve Management, Wang, D.H and Fang, C. (eds.). China Environment Sciences Press, Beijing.

Xu, J., Grumbine, R.E., Shrestha, A., Eriksson, M., Yang, X., Wang, Y.U.N. and Wilkes, A. 2009. The melting Himalayas: cascading effects of climate change on water, biodiversity, and livelihoods. *Conservation Biology*, 23(3): 520-530.

Xu, Q.Y., Wang, D., Quan, G.M., Xiang, H.M. and Zhang, J.E. 2020. Invasive *Chromolaena odorata* species specifically affects growth of its co-occurring weeds. Annals of the New York Academy of Sciences, 1470(1): 57-66.

Xuan, T.D., Shinkichi, T., Hong, N.H., Khanh, T.D. and Min, C.I. 2004. Assessment of phytotoxic action of *Ageratum conyzoides* L.(billy goat weed) on weeds. Crop Protection, 23(10): 915-922.

Yang, Q., Carrillo, J., Jin, H., Shang, L., Hovick, S.M., Nijjer, S., Gabler, C.A., Li, B. and Siemann, E. 2013. Plant–soil biota interactions of an invasive species in its native and introduced ranges: Implications for invasion success. *Soil Biology and Biochemistry*, 65: 78-85.

Yang, X., Cheng, Y.F., Deng, C., Ma, Y., Wang, Z.W., Chen, X.H. and Xue, L.B. 2014. Comparative transcriptome analysis of eggplant (*Solanum melongena* L.) and turkey berry (*Solanum torvum* Sw.): phylogenomics and disease resistance analysis. *BMC Genomics*, 15: 1-13.

Yelenik, S.G., Stock, W.D. and Richardson, D.M. 2004. Ecosystem level impacts of invasive *Acacia saligna* in the South African fynbos. *Restoration Ecology,* 12(1): 44-51.

Yin, S., Wang, X., Meng, Z., Mo, X., He, M. and Liu, J. 2024. Mitigating invasive species using biochar derived from invasive plants in saline-alkaline wetland plant communities dominated by *Amaranthus palmeri*. *Applied Vegetation Science,* 27(3): e12801.

Young, P.H., Hsu, Y.J. and Yang, W.C. 2010. *Bidens pilosa* and its medicinal use. In Recent Progress in Medicial Plants/Drug plants II, Awaad, A.S., Singh, V.K andGovil, J.N.(eds.),Standium Press, Houston, Tex, USA.

Yu, X., He, T., Zhao, J. and Li, Q. 2014. Invasion genetics of *Chromolaena odorata* (Asteraceae): extremely low diversity across Asia. *Biological Invasions,* 16: 2351-2366.

Yue, M., Yu, H., Li, W., Yin, A., Cui, Y. and Tian, X. 2019. Flooding with shallow water promotes the invasiveness of *Mikania micrantha*. *Ecology and Evolution,* 9(16): 9177-9184.

Zachariades, C., Day, M., Muniappan, R. and Reddy, G.V.P. 2009. *Chromolaena odorata* (L.) king and robinson (Asteraceae), 130-162. In Biological control of tropical weeds using arthropods, Zachariades, C., Day, M., Muniappan, R. and Reddy, G.V.P. (eds). Cambridge University Press, Cambridge.

Zahara, M. 2019. Description of *Chromolaena odorata* LRM King and H. Robinson as medicinal plant: A Review. In IOP Conference Series: Materials Science and Engineering. IOP Publishing. 506(1): 012022.

Zalamea, M., and González, G. 2008. Leaf-fall phenology in a subtropical wet forest in uerto Rico: from species to community patterns, *Biotropica*, 40(3), pp. 295–304.

Zani, D., Crowther, T.W., Mo, L., Renner, S.S. and Zohner, C.M. 2020. Increased growing-season productivity drives earlier autumn leaf senescence in temperate trees. *Science,* 370(6520): 1066-1071.

Zettlemoyer, M.A., Schultheis, E.H. and Lau, J.A. 2019. Phenology in a warming world: differences between native and non-native plant species. *Ecology letters,* 22(8): 1253-1263.

Zhang, C.B., Wang, J., Qian, B.Y. and Li, W.H. 2009. Effects of the invader *Solidago canadensis* on soil properties. *Applied Soil Ecology*, 43(2-3): 163-169.

Zhang, H., Yuan, W., Liu, S. and Dong, W. 2015. Divergent responses of leaf phenology to changing temperature among plant species and geographical regions. *Ecosphere*, 6(12): 1-8.

Zhang, L.Y. and Ye, W.H. 2002. Community invasibility and its influencing factors. *Chinese Journal of Plant Ecology*, 26(1): 109-114.

Zhang, L.Y., Ye, W.H., Cao, H.L. and Feng, H.L. 2004. *Mikania micrantha* HBK in China–an overview. *Weed Research*, 44(1): 42-49.

Zhang, Q., Yang, R., Tang, J., Yang, H., Hu, S. and Chen, X. 2010. Positive feedback between mycorrhizal fungi and plants influences plant invasion success and resistance to invasion. *PLoS one*, 5(8): e12380.

Zhang, W., Yin, D., Huang, D., Du, N., Liu, J., Guo, W. and Wang, R. 2015. Altitudinal patterns illustrate the invasion mechanisms of alien plants in temperate mountain forests of northern China. *Forest Ecology and Management*, 351: 1-8.

Zhang, Y.G., Zhang, Y.Y., Cai, J.P., Zhu, P., Gao, H.J. and Jiang, Y. 2014. Variation in available micronutrients in black soil after 30-year fertilization treatment. *Plant, Soil and Environment*, 60(9): 387-393.

Zhao, W., Liu, X., Huang, Q. and Cai, P. 2015. *Streptococcus suis* sorption on agricultural soils: Role of soil physico-chemical properties. *Chemosphere*, 119: 52-58

Zohner, C.M. and Renner, S.S. 2019. Ongoing seasonally uneven climate warming leads to earlier autumn growth cessation in deciduous trees. *Oecologia*, 189(2): 549-561.

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RESEARCH ARTICLE

Unveiling the ecological footprint of dominant invasive species in Pakke Tiger Reserve, North-East Himalaya

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ABSTRACT

The present study investigated the distribution patterns and abundance of nine dominant Invasive Alien Plant Species (IAPS) in the Pakke Tiger Reserve, Arunachal Pradesh. The study was conducted to explore their proliferation across the various forest types in the region from 2019 to 2022. The result of the study revealed that *Chromolaena odorata* was widely distributed across all the forest types, exhibiting the highest relative density and relative abundance. Moreover, the distribution pattern of all the selected invasive species showed contagious distribution across the forest types, except for *U. lobata* displayed random distribution in non-forest areas. The level of invasiveness across the forest types also revealed that *C. odorata* exhibited the maximum proliferation, followed by *A. conyzoides*, *M. micrantha* and *U. lobata*. Furthermore, areas such as non-forest, Riverine Forest and Assam Alluvial Plain Semi Evergreen Forests, located close to human settlements and roads, showed a high level of invasion. If prompt and effective management strategies are not implemented, the Pakke Tiger Reserve may end up serving as a habitat for globally detrimental IAPS, according to the current study.

INTRODUCTION

Invasive alien plant species (IAPS) are one of the factors responsible for global environmental changes, biodiversity loss, species extinction, and disruption of ecosystem processes critical to human well-being worldwide (Charles & Dukes, 2007). IAPS can alter community composition, induce local extinctions and the loss of native genotypes, alter habitats, and have an impact on food-web properties, ecosystem processes, and functioning (Bellard, Cassey & Blackburn, 2016). Their increased expansion in new region and capability to strike losses on ecological as well as economy has become major concern for the policymakers, thus requiring urgent management (Early et al., 2016). The first and most important stage in developing a management strategy for an alien plant is determining its distribution within its newly extended range (Auld & Johnson, 2014). Furthermore, research conducted on various forest types, elevation gradients and disturbances related to IAPS invasion, may be very important in identifying trends and aiding to develop management strategies and planning (Rahbek, 2005). Additionally, it is well known that disturbances affect a forest's pattern and community structure (Pathak et al., 2021). To support the management authority, it is essential to assess the degree of invasion and the distribution pattern of IAPS across different types of forests.

Many researches on biological invasion in various types of forests have been undertaken worldwide (Dyderski & Jagodzinski, 2019), but few have been conducted in the forests regions of the eastern Himalaya region (Kosaka et al., 2010; Singh et al., 2021). The Eastern Himalaya region is one of thirty-six global biodiversity hotspots and is distinguished by remarkable biodiversity of major ecological and global significance, such as global 200 ecoregions (Brooks et al., 2006). The region is a convergence of two biogeographic origins, namely the Palearctic and Indo-Malayan realm (Hua, 2012). In addition, its forest resources provide a range of ecosystem goods and services that support human life, making it the primary regulator of life (Saha & Sundriyal, 2012). The eastern Himalayan region is among the planet's most delicate ecosystems and is vulnerable to natural disasters (Diaz, Grosjean & Graumlich, 2003), particularly from IAPS that are spreading across the area and endangering its mountainous forest environment. Despite the eastern Himalayan region's unique biodiversity, little is known about biological invasion and its effects on regional biodiversity. The paucity of empirical data in many sections of the region makes the management information unattainable. This condition necessitates thorough research on biological intrusions and evaluation of the impact they have on the surrounding ecosystem.

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The current research aims to evaluate the degree of invasiveness and the patterns of distribution of most common alien plant invaders, i.e., *Ageratum conyzoides, Bidens pilosa, Chromolaena odorata, Mikania micrantha, Solanum torvum, Solanum viarum, Synedrella nodiflora, Triumfetta rhomboidea* and *Urena lobata* across various forest types of Pakke Tiger Reserve in the eastern Himalaya. As a consequence, the findings of this study may add to our understanding of the effects of plant invasion and provide practical management strategies for both the specific area and the Indian Himalayan Region (IHR) as a whole.

MATERIALS AND METHODS

Study Area

With an extraordinarily rich biodiversity, the Pakke Tiger Reserve (PTR) is located in the Pakke Kessang district (erstwhile East Kameng) of Arunachal Pradesh, in the eastern Himalayan region. The PTR covers an area of 861.95 km^2 with an elevational range from 100 to 2040 m asl. The vegetation of PTR varies from riverine forest to Assam's alluvial plain semi-evergreen forest, with a variety of intermixing forest types in between. In the lower plains and foothills, *Pterospermum acerifolium, Sterculia villosa, Monoon simiarum, Duabanga grandiflora* and *Stereospermum chelonoides* predominate, whilst the primary canopy-forming species in the higher region include *Castanopsis tribuloides, C. indica* and *Mesua ferrea,* in the higher region. Furthermore, the study area is disturbed by anthropogenic activities in the vicinities (Bapu & Nimasow, 2017), thus making it susceptible to disturbance and invasion by alien species.

Vegetation sampling

Prior to vegetation sampling, a reconnaissance survey was conducted to identify the forest types following Champion and Seth (1968) and FSI (2009) i.e., the Assam Alluvial Plain Semi Evergreen Forest (AAPSEF), the East Himalayan Mixed Coniferous Forest (EHMCF), the East Himalayan Subtropical Wet Hill Forest (EHSTWHF), the Non-Forest (NF) and the Riverine Forest (RF). Within each forest types, sites infested with dominant invasive alien plant species, i.e., *Ageratum conyzoides, Bidens pilosa, Chromolaena odorata, Mikania micrantha, Solanum torvum, Solanum viarum, Synedrella nodiflora, Triumfetta rhomboidea* and *Urena lobata* were selected. An intensive sampling was carried out between September, 2021 to February, 2023 in order to determine the overall diversity spectrum of plant species in the PTR. A spatial grid (25 km^2) was created using ArcGIS 10.6.1 and the five forest types were identified using the GIS domain in PTR (Figure 1). In order to document the vegetation, the plots were laid at random within the vicinity of the accessible, maintaining a minimum distance of 400 m. A total of 208 plots (20×20 m) were laid in various forest types (65 in AAPSEF, 84 in EHMCF, 27 in EHSTWH, 16 each in NF and RF; Table 1).

The diversity of IAPS and their relative contribution to overall species richness and the percentage of the entire community cover that they comprise, were measured to determine the level of invasion following Chytry et al. (2005), Medvecka et al. (2018) and Chaudhary et al. (2019). For the purpose of gathering data, a random sampling procedure was carried out at each of the research plots in five distinct forest types. The nested quadrat design was used for collecting the vegetation data following Rai (2006) by laying 1, 20×20 m plot for the trees, and within that 2, 5×5 m plots for shrubs and saplings and 5, 1×1 m plots for herbs and seedlings. The data was analyzed for obtaining density, abundance and their relative values following Mishra (1968) and Mueller-Dombois and Ellenberg (1974). Relative density helps in understanding contribution of each species to the plant community highlighting the ecological value of the community, whereas relative abundance integrates each species output contribution to the plant community. Both were considered for the study because they present various perspectives on community composition and structure that affects conservation and management and usually share more about the dynamics of plant populations in the whole community. The nativity of a species was determined following Index Kewensis Plantarum Phanerogamarum and supplements (Anonymous 1883-1885, 1886-1970). The distribution pattern of each species was obtained by calculating abundance (A) and Frequency (F) ratio following Curtis and Cottam (1956) as regular (< 0.025), random (0.025-0.05) and contagious (>0.05). Paleontological Statistics (PAST) software, version 4.17 was used to analyze the data (Hammer, Harper & Ryan, 2001).

RESULTS

Floristic diversity

A total of 151 species, consisting of 46.4% tree species, 15.9% shrub species, 31.8% herb species, 2.6% each climber and grasses, while 0.7% sedges representing 57 different families and 124 different genera, were recorded through plot sampling. The family Asteraceae had the highest number of species (18), making it the most dominant family followed by Malvaceae (11), Fabaceae (10) and Acanthaceae (8). Of the total number of species, 63 species were native to the Himalayan and Indian Oriental (Indo-Himalaya) regions. The total species richness across various forest types was highest in AAPSEF (137) and lowest in NF (77). The tree species richness was recorded maximum in EHMCF (67) and minimum in NF (33), shrub species richness was maximum in AAPSEF (23) and minimum in EHSTWH (15) and the herbaceous species richness was maximum in AAPSEF (55) and minimum in NF and RF (28 species each; Table 1).

Density and abundance of dominant species

Ageratum conyzoides was present in all forest types with highest relative abundance (3.9%) and relative density (21.7%) in EHSTWH forest (Figure 2 and 3), while the species had lowest relative abundance (2.6%) and relative density (19.1%) in AAPSEF forest. The density of *A. conyzoides* ranged from 30888 (RF) to 37931 individuals ha^{-1} (EHSTWH) with an average density 34340±12781 individuals ha^{-1} in PTR. However,

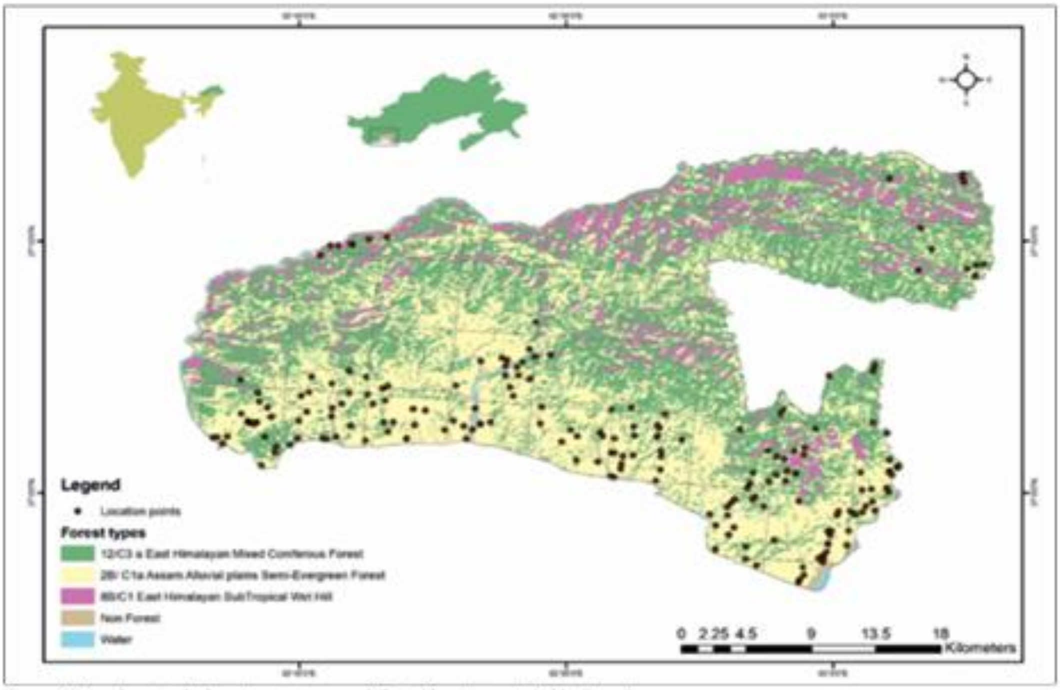

Figure 1. Map showing the location point across different forest types in Pakke Tiger Reserve

Table 1. Area, sampling effort and individuals of species within sampled area across various forest types in PTR. Values in parenthesis are number of IAPS

Forest types	Area (km²)	Plots	Total individuals	Number of taxa Tree	Shrubs	Herbs
Assam Alluvial Plain Semi Evergreen Forest (AAPSEF)	390.7	65	137 (41)	59 (0)	23 (8)	55 (33)
East Himalaya Mixed Coniferous Forest (EHMCF)	364.5	84	133 (26)	67 (0)	21 (4)	45 (22)
East Himalaya Subtropical Wet Hill Forest (EHSWHF)	79.8	27	81 (13)	36 (0)	15 (4)	30 (9)
Non-Forest (NF)	17.5	16	77 (21)	33 (0)	16 (5)	28 (16)
Riverine Forest (RF)	9.4	16	80 (20)	37 (0)	15 (3)	28 (17)

*AAPSEF- Assam Alluvial Plain Semi Evergreen Forest; EHMCF- East Himalaya Mixed Coniferous Forest; EHSTWH- East Himalaya Subtropical Wet Hill; NF- Non-Forest and RF- Riverine Forest.

Bidens pilosa had the highest relative abundance (3.5%) and relative density (10.2%) in the NF forest type where it was the dominant herb layer and the lowest relative abundance (2.2%) and relative density (7.1%) in the AAPSEF forest (Figure 2 & 3). The average density of *Bidens pilosa* was 29332±708 individuals ha^{-1}, varying from 26875 (EHSTWH) to 31250 individuals ha^{-1} (NF) in PTR. *Chromolaena odorata* was present along the roadside all across the forest types and formed a dominant shrub species layer, with highest relative abundance (10%) and relative density (46%) in NF and the lowest relative abundance (6.5%) and relative density (39%) was in EHMCF (Figure 2 & 3). The density of *C. odorata* ranged from 1815 (RF) to 2533 individuals ha^{-1} (EHSTWH), with an average of 2054±133 individuals ha^{-1}. *Mikania Micrantha* was present in all forest types with *Chromolaena odorata* in close proximity forming a bonding pair and had the highest relative density (16.8%) and relative abundance (4.5%) in EHSTWH and had similar values for relative abundance with NF and RF but different in relative density (Figure 2 & 3). The average density of *Chromolaena odorata* was 40995±1549 individuals ha^{-1}, ranging from 37087 (NF) to 44761 individuals ha^{-1} (EHMCF). *Solanum torvum* had the highest relative density (5.5%) and relative abundance (6.7%) in NF (Figure 2 & 3). The density of the species ranged from 1000 (RF) to 1263 individuals ha^{-1} (EHMCF) with an

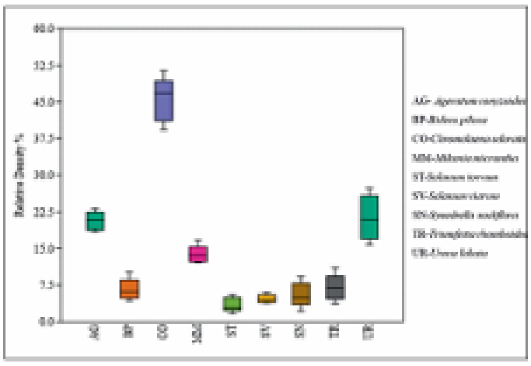

Figure 2. Relative density (%) of dominant invasive alien plant species in PTR.

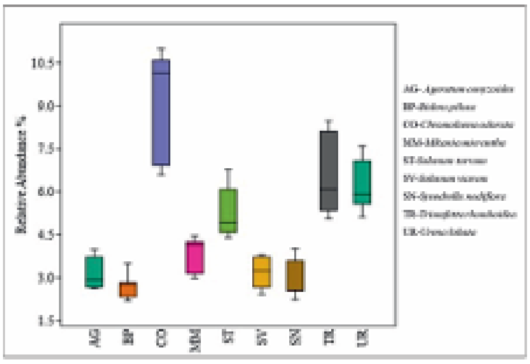

Figure 3. Relative Abundance (%) of dominant invasive alien plant species in PTR.

average density of 1163±47 individuals ha^{-1}. *Solanum viarum* had the highest relative density (5.9%) and relative abundance (3.7%) was observed in NF and lowest relative density and relative abundance in EHSTWH (Figure 2 & 3). The average density was 34050±1006 individuals ha^{-1}, varying from 30512 (AAPSEF) to 36440 individuals ha^{-1} (EHMCF) across the forest types. *Synedrella nodiflora* was present more predominantly in EHMCF, NF and RF and has the highest relative abundance (4%) and relative density (9.2%) in NF while the lowest relative abundance (2%) and relative density (2%) was in EHSTWH (Figure 2 & 3). The average species density was 32176±2528 individuals ha^{-1} and ranged from 24444 (RF) to 37692 individuals ha^{-1} (EHSTWH). *Triumfetta rhomboidea* has the highest relative abundance (8.5%) and relative density (5.5%) in NF and lowest relative abundance (5%) and relative density (7%) in EHMCF (Figure 2 & 3), with an average density 1430±71 individuals ha^{-1}, ranged from 1200 (EHSTWH) to 1650 individuals ha^{-1} (RF). *Urena lobata* had the highest relative abundance (7.5%) and relative density (27.3%) in NF and lowest relative abundance (5.5%) and relative density (15.7%) in AAPSEF (Figures 2 & 3). The average density across forest types was 1367±66 individuals ha^{-1}, ranged from 1165 (EHSTWH) to 1505 individuals ha^{-1} (AAPSEF).

Species distribution pattern

The distribution pattern of dominant species in PTR is shown in Figure 4. The species like, *Ageratum conyzoides*, *Bidens pilosa*, *Chromolaena odorata*, *Mikania micrantha*, *Solanum torvum*, *Solanum viarum*, *Synedrella nodiflora* and *Triumfetta rhomboidea* showed contagious or clustered distribution in all

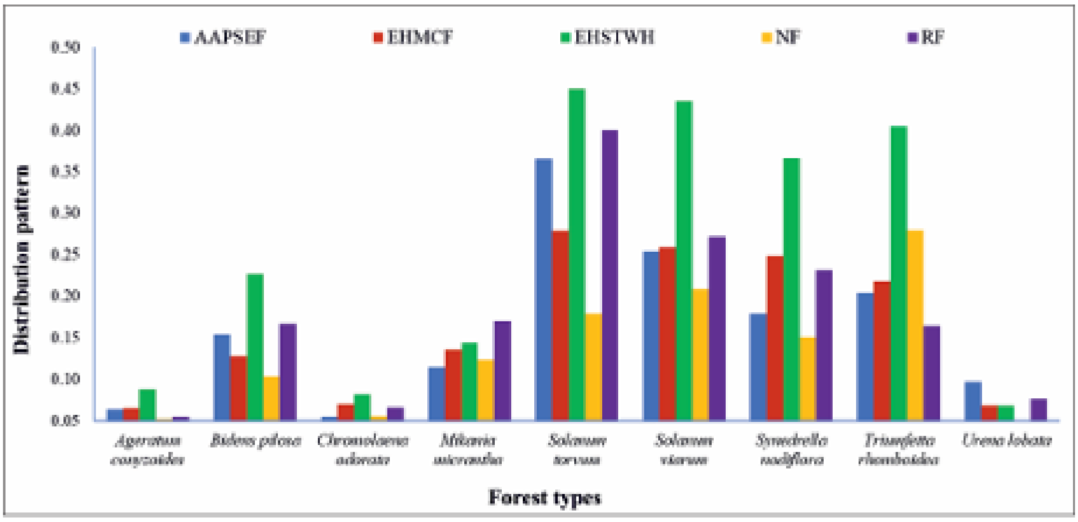

Figure 4. Distribution pattern of dominant invasive alien plants in different forest types in PTR. AAPSEF- Assam Alluvial Plain Semi Evergreen Forest; EHMCF- East Himalaya Mixed Coniferous Forest; EHSTWH- East Himalaya Subtropical Wet Hill, NF- Non-Forest and RF- Riverine Forest

the forest types. However, none of the species were found to be regular distribution which is a good sign as the naturalization process has not taken place in PTR. However, for *Urena lobata*, native to tropical Africa (Reddy, 2008), the distribution pattern demonstrated its contagiousness in four forest types (AAPSEF, EHMCF, EHSTWH and RF), but in NF, it was randomly distributed (Figure 4).

Invasion by dominant species in various forest types

The invasion of *Ageratum conyzoides* was 19, 20, 18, 10 and 11% in AAPSEF, EHMCF, EHSTWH, NF and RF, respectively (Figure 5). In terms of area, *Ageratum* occupied 77.3 km^2 in AAPSEF, 74.9 km^2 in EHMCF, 14.8 km^2 in EHSTWH, 1.8 km^2 in NF and 1.0 km^2 in RF, of the respective total forest cover area. The invasion of *Bidens pilosa* was 10% in AAPSEF (40.4 km^2), 12% in EHMCF (44.9 km^2), 11% in EHSTWH (9.5 km^2), 8% in NF (1.5 km^2) and 6% in RF (1.1 km^2; Figure 5). *Chromolaena odorata* invasion was 20% each in EHMCF (72.9 km^2) and RF (1.9 km^2), while minimum (13.6%) was in EHSTWH (10.8 km^2; Figure 5). The level of invasion by *Mikania micrantha* was highest (18%) in EHMCF (68.5 km^2) followed by EHSTWH (16%; 12.8 km^2) and AAPSEF (15%; 58.8 km^2), RF (13%; 1.2 km^2) and NF (9%; 1.7 km^2) of the respective total forest cover (Figure 5). *Solanum torvum* covers an area of 49.72 km^2 in AAPSEF (12%), 35.5 km^2 in EHMCF (9%), 6.65 km^2 in EHSTWH (8%), 1.7 km^2 in NF (10%) and 0.4 km^2 in RF (5%; Figure 5). *Solanum viarum* has invaded 40.6 km^2 (10.4%) in AAPSEF, 54.03 km^2 (14.8%) in EHMCF, 7.6 km^2 (9.5%) in EHSTWH, 1.6 km^2 (9.2%) in NF and 0.7 km^2 (7%) in RF (Figure 5) of the total forest cover. *Synedrella nodiflora* covered 38.6 km^2 (9.9%) in AAPSEF, 41.1 km^2 (11.8%) in EHMCF, 4.9 km^2 (6.1%) in EHSTWH, 1.9 km^2 (10.8%) in NF and 0.9 km^2 (10%) in RF, of the total forest cover (Figure 5). *Triumfetta rhomboidea* had highest level of invasion in AAPSEF (16.3%; 63.7 km^2) followed by EHMCF (11.6%; 42.3 km^2), EHSTWH and NF (8.7% each) with (7.0 and 1.5 km^2, respectively) and RF (9.3%, 0.9 km^2; Figure 5). The total area of invasion by *Urena lobata* across the forest types was 17% in AAPSEF (67.0 km^2), 16.4% in EHMCF (59.7 km^2), 10.7% in RF (1.1 km^2), 9.8% in EHSTWH (7.8 km^2) and 8.4% in NF (1.4 km^2; Figure 5).

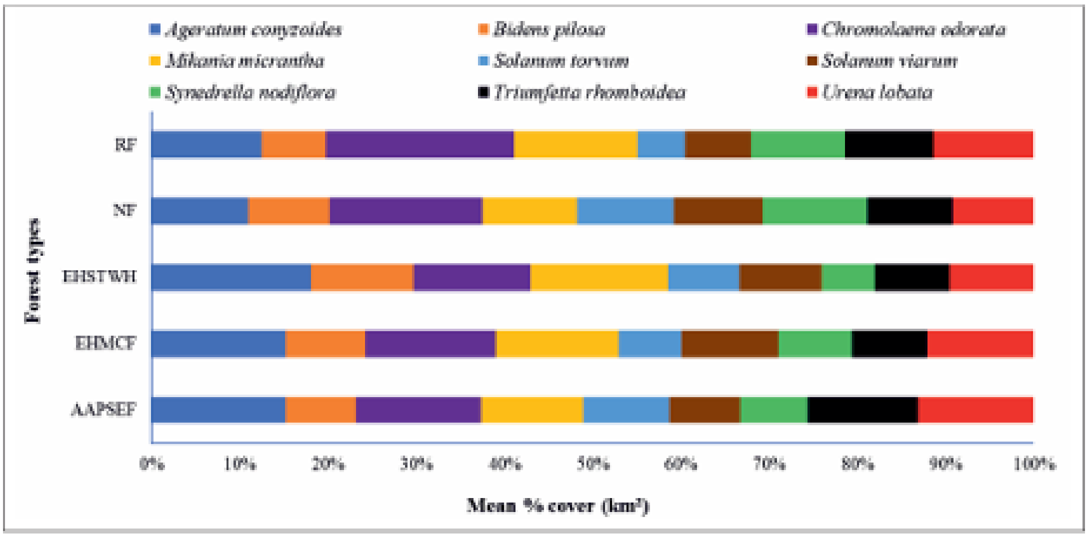

Figure 5. Mean percent cover of dominant invasive alien plants across forest types in PTR.

DISCUSSION

One of the most significant global issues facing natural ecosystems is the invasion of alien plant species. There is widespread recognition of the detrimental effect that IAPS have on the world's biodiversity, which have led to the loss of several plant and animal species in world various ecosystems (Baillie, Hilton-Taylor, & Stuart, 2004; Early et al., 2016; Sandilyan, 2015). Whether they are in terrestrial, aquatic, or island ecosystems, IAPS have seriously threatened native biodiversity worldwide (Enserink, 1999). The Himalayan region which has been considered as one of the biodiversity hotspot regions of the world is at an increased risk of invasion by IAPS (Khuroo et al., 2021). IHR is facing unprecedented anthropogenic pressure brought on by the economy's rapid expansion in the region which makes up major portion of it (Chitale, Behera & Roy, 2014) and eventually leading to spread and proliferation of IAPS (Pathak et al., 2021). Moreover, ecologically sensitive mountainous regions of IHR are continuously facing the seriousness of climate changes and invasion of IAPS which are becoming more challenging at every level (Mack et al., 2000; Tripathi, Khan & Yadav, 2012). Therefore, the present focused on the distribution pattern of invasive alien plants invaded forest ecosystems of PTR.

Ageratum conyzoides, "billy goat weed" is one of the troublesome and rapidly colonizing weeds in tropical and subtropical countries (Batish et al., 2009). The species is reported as one of the main invaders in Indian Himalayan region, establishing up to the elevation of 3000 m (Dogra, 2007) with the ability to transitioned, in the tropical forest of the eastern Himalayan region (Singh et al., 2002). In the present study the species was found more predominantly in NF and EHSTWH and showed an increasing trend with increased in elevation, however, it showed reverse trend in the level of invasion. Dogra et al. (2009) also reported the decline in native species richness in the forested areas of IHR following invasion by *Ageratum conyzoides*. Lamsal et al. (2018) and Ray, Behera and Jacob (2019) reported that the species has shown more potentiality to invade rigorously in the lower parts of Arunachal Pradesh. *Bidens pilosa*, native to tropical America, has successfully invaded subtropical, tropical and temperate regions with its greater phenotypic plasticity and adaptability (Bartolome, Villaseñor & Yang, 2013; Malik, Pandey & Bhatt, 2016). Moreover, the species has reached subalpine from subtropical region of the Indian Himalayan regions (Kosaka et al., 2010; Khatri et al., 2022), it favours disturbed habitats but has been observed in undisturbed forests and grasslands areas (Osaki, Wasaki & Nakatsubo, 2022), which was similarly observed in the present study, i.e., EHSTWH and EHMCF. The level of invasiveness was more in NF, which is open forest with minimum canopy cover above. Despite its widespread prevalence, research on invasion dynamics of *Bidens pilosa* is limited to Himalayan ecosystems.

Chromolaena odorata is one of the world's 100 worst invasive alien plant species according to Lowe et al. (2000) because of its unique characteristic features. High reproductive capacity, dispersal mechanism and competitive ability make this plant a successful invader in ecosystem (Joshi, 2001). In the present study, the species was found to be one of the most dominant invaders and observed establishing in the accessible trails as well as the motorable roads in the lower zones of AAPSEF, NF and RF. This pattern was also observed by Rao (1977) in the lower belt of eastern Himalaya. However, the east Himalaya are more susceptible to invasion in future scenario (Barik & Adhikari, 2012). *Mikania micrantha* is regarded as the most notorious invader (Cronk & Fuller, 2014) which poses serious threat to the biodiversity (Wang et al., 2004). In the present study, *M. micrantha* was seen covering the canopy of small trees, shrubs in EHSTWH, RF and NF. Pronounced invasiveness of *Mikania micrantha* in riverine and moist forested areas was observed. The eastern Himalayan region exhibits a high probability of invasion by this species, was also reported by Rameshprabu and Swamy (2015).

Solanum torvum, a successful colonizer because of its invasive trait (Welman, 2003), has become naturalized throughout the tropics and subtropics worldwide (CABI, 2024), where it is known to be invasive weed of forested areas, often along river sides, forest clearings and other disturbed habitats (Francis, 2004). The study showed that abundance decreases with the increase in elevation and invasive in lower part of PTR (AAPSEF and NF), which might be due to openness with anthropogenic activities as it cannot survive under a closed forest canopy (Francis, 2004). *Solanum viarum* is constantly associated with the anthropogenic disturbances (Medal et al., 2012) and grows in the understorey of rainforest, grasslands and dry forests (Nee, 1991). In eastern Himalayan region, this species occurs mainly below 2000 m and invades natural habitats including forests edges and riverine area (Parker, 1992), same observation has been confirmed in the present study. The present study observed the similarity in pattern of invasion in the lower foothill forest types in the Himalayan region as reported by Singh, Khanna, and Sudhir (1998). *Synedrella nodiflora* can tolerate broad range of environmental condition and can grow profusely in all disturbed tropical and subtropical habitats (CABI, 2024). The species is not well studied with respect to its invasiveness. The species was found growing in abundance in the lower foothills of the present study area (NF and RF) as the species prefers highly disturbed areas.

Triumfetta rhomboidea is listed as invasive in 12 countries including India (CABI, 2024) because of their competitive nature of invasion in pastures and disturbed areas in forests, thereby preventing the establishment of native species (Motooka, 2003). The present study observed that this species has started to form dense stands, thereby limiting the growth of native flora which is also reported by Motooko (2003). *Urena lobata*, an aggressive noxious weed (Randall, 2012) and fast-growing plant is capable of forming dense patches and monospecific stand (Langeland, 2008). The species invasiveness in certain habitats such as disturbed forest, shrubland, riparian areas, forest margins, roadsides was reported by Francis (2004). The species invasion has been found in the riverine areas as well as the interim forest of the study area, along trails or roadsides. These species are also expected to expand in the lower foothill's regions of the eastern Himalaya region with the probability of 0.5 to 0.75% (Ray, Behera and Jacob, 2019).

The species like *Ageratum conyzoides, Chromolaena odorata, Mikania micrantha* and *Urena lobata* were found closely associated with each other in the study area. Although *Ageratum conyzoides* and *Urena lobata* have their major invasion near the roadside and disturbed areas, *Chromolaena odorata* and *Mikania micrantha* were found inside the forested areas, even in the undisturbed areas. In the riverine, non-forested and lower plain region of PTR, *Chromolaena odorata* and *Mikania micrantha* have predominance, with full ground cover by *C. odorata* and *M. micrantha* covering the full matured trees. The current study revealed that PTR is at the risk of becoming habitat for some of the world most harmful IAPS in the future, unless effective strategies and management plans are implemented promptly.

CONCLUSIONS

The study concludes that the forests in the study area are being invaded by the aggressive and noxious IAPS. These species are preferring the disturbed forest area especially AAPSEF, NF and RF due to increased anthropogenic activities. Moreover, it is required to minimize the disturbances in these forests in order to control the invasion of alien species. However, the EHMCF and EHSTWH with higher elevation have less invasion as compared to other forest types. Furthermore, indigenous native species integrating with IAPS should be taken into consideration while preparing management strategies as this could be the solution for controlling the invaded forest habitats in the region. Across the different forest types, various native species were recorded that have high abundance and densities close to that of the invasive alien plant species. These plants can be used to control the invaded area in the study area. Furthermore, Singh et al. (2021) also state that in order to counter the threat of IAPS, preparation of inventories, early detection and better planning are utmost necessity. Therefore, there need an urgent work to control proliferation of these species in the eastern Himalayan region, through developing proper management planning and control strategies.

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REFERENCES

Anonymous., 1883-1885 and 1886-1970. *Index Kewensis Plantarum Phanerogamarum. Oxford:* Clarendon Press, **1-2** (15 suppl.).

Auld, B.A. and Johnson, S.B., 2014. Invasive alien plant management. *CABI Reviews*, pp.1-12. https://doi.org/10.1079/PAVSNNR20149037

Baillie, J., Hilton-Taylor, C. and Stuart, S.N. (eds), 2004. *2004 IUCN red list of threatened species: a global species assessment*. IUCN.

Bapu, T.D. and Nimasow, G., 2017. Land cover change assessment of Pakke Tiger Reserve (PTR), East Kameng district of Arunachal Pradesh. *Methodology*, **136**, p.41.

Barik, S.K. and Adhikari, D., 2012. Predicting the geographical distribution of an invasive species (*Chromolaena odorata* L. (King) & H.E. Robins) in the Indian subcontinent under climate change scenarios. In Invasive alien plants: an ecological appraisal for the Indian subcontinent. Wallingford UK: CABI, pp.77-88. https://doi.org/10.1079/9781845939076.0077

Bartolome, A.P., Villaseñor, I.M. and Yang, W.C., 2013. *Bidens pilosa* L. (Asteraceae): Botanical Properties, Traditional uses, Phytochemistry, and Pharmacology. *Evidence-Based Complementary and Alternative Medicine*, **1**, p.340215. https://doi.org/10.1155/2013/340215

Batish, D.R., Kaur, S., Singh, H.P. and Kohli, R.K., 2009. Nature of interference potential of leaf debris of *Ageratum conyzoides*. *Plant Growth Regulation*, **57**, pp.137-144. https://doi.org/10.1007/s10725-008-9329-9

Bellard, C., Cassey, P. and Blackburn, T.M., 2016. Alien species as a driver of recent extinctions. *Biology Letters*, **12**(2), p.20150623. https://doi.org/10.1098/rsbl.2015.0623

Brooks, T.M., Mittermeier, R.A., Da Fonseca, G.A., Gerlach, J., Hoffmann, M., Lamoreux, J.F., Mittermeier, C.G., Pilgrim, J.D. and Rodrigues, A.S., 2006. Global biodiversity conservation priorities. *Science*, **313**(5783), pp.58-61. https://doi.org/10.1126/science.1127609

CABI., 2024. *CABI Compendium*. Wallingford, UK: CAB International. https://www.cabidigitallibrary.org/. (Accessed on 29.05.2024)

Champion, H.G. and Seth, S.K., 1968. *A revised survey of the forest types of India*. Manager of publications, Delhi.

Charles, H. and Dukes, J.S., 2007. Impacts of invasive species on ecosystem services. *Biological Invasions*, pp.217-237. https://doi.org/10.1007/978-3-540-36920-2_13

Chaudhary, A., Adhikari, B.S., Joshi, N.C. and Rawat, G.S., 2019. Patterns of invasion by Crofton weed (*Ageratina adenophora*) in Kailash sacred landscape region of western Himalaya (India). *Environment Conservation Journal*, **20**(3), pp.9-17. https://doi.org/10.36953/ECJ.2019.20302

Chitale, V.S., Behera, M.D. and Roy, P.S., 2014. Future of endemic flora of biodiversity hotspots in India. *Plos One*, **9**(12), p.e115264. https://doi.org/10.1371/journal.pone.0115264

Chytry, M., Pysek, P., Tichy, L., Knollova, I. and Danihelka, J., 2005. Invasions by alien plants in the Czech Republic: a quantitative assessment across habitats. *Preslia*, **77**(4), pp.339-354.

Cronk, Q.C. and Fuller, J.L., 2014. *Plant invaders: The Threat to natural ecosystems*. Routledge. https://doi.org/10.4324/9781315071831

Curtis, J.T and Cottom, G., 1956. Plant ecology workbook. A laboratory, field and reference manual, p.86.

Diaz, H.F., Grosjean, M. and Graumlich, L., 2003. Climate variability and change in high elevation regions: past, present and future. *Climatic Change*, **59**(1), pp.1-4. https://doi.org/10.1023/A:1024416227887

Dogra, K.S., 2007. *Impact of some invasive species on the structure and composition of natural vegetation of Himachal Pradesh*. PhD thesis, Panjab University, Chandigarh, India.

Dogra, K.S., Kohli, R.K., Sood, S.K. and Dobhal, P.K., 2009. Impact of *Ageratum conyzoides* L. on the diversity and composition of vegetation in the Shivalik hills of Himachal Pradesh (Northwestern Himalaya), India. *International Journal of Biodiversity and Conservation*, **1**(5), pp.135-145.

Dyderski, M.K. and Jagodzinski, A.M., 2019. Context-dependence of urban forest vegetation invasion level and alien species' ecological success. *Forests*, **10**(1), p.26. https://doi.org/10.3390/f10010026

Early, R., Bradley, B.A., Dukes, J.S., Lawler, J.J., Olden, J.D., Blumenthal, D.M., Gonzalez, P., Grosholz, E.D., Ibanez, I., Miller, L.P. and Sorte, C.J., 2016. Global threats from invasive alien species in the twenty-first century and national response capacities. *Nature Communications*, **7**(1), p.12485. https://doi.org/10.1038/ncomms12485

Enserink, M., 1999. Biological invaders sweep in. *Science*, **285**, pp.1834-1836. https://doi.org/10.1126/science.285.5435.1834

Francis, J.K., 2004. *Wildland Shrubs of the United States and its Territories: Thamnic Descriptions*, USDA Forest Service, International Institute of Tropocal Forestry, San Juan PR; Rocky Mountain Research Station, Fort Collins CO; General Technical Report. **I**. https://doi.org/10.2737/IITF-GTR-26

FSI., 2009. *State of Forest Report*. Published by Forest Survey of India. https://fsi.nic.in/forest-report-2009. (Accessed on 29.05.2024)

Hammer, Ø., Harper, D.A.T and Ryan, P.D., 2001. Past: paleontological statistics software package for education and data analysis. *Palaeontologia Electronica*, **4**(1), p.1.

Hua, Z., 2012. Biogeographical divergence of the flora of Yunnan, southwestern China initiated by the uplift of Himalaya and extrusion of Indochina block. *PLoS One*, **7**(9): p.e45601. https://doi.org/10.1371/journal.pone.0045601

Joshi, C., 2001. *Invasive Banmara (Chromolaena odorata), Spatial detection and prediction. MSc. Thesis*. International Institute for Aerospace Survey and Earth Sciences, Enschede, The Netherlands, p.53.

Khatri, K., Negi, B., Bargali, K. and Bargali, S.S., 2022. Trait variability in co-occurring invasive and native plant species in road side population of Kumaun Himalaya. *Brazilian Journal of Botany*, **45**(3), pp.1099-1110. https://doi.org/10.1007/s40415-022-00827-y

Khuroo, A.A., Ahmad, R., Hamid, M., Ruquia, G., Malik, A.H. and Rashid, I., 2021. Alien flora of Indian Himalayan Region: taxonomic, biogeographical and ecological assessment. *New Vistas in Indian Flora*, **2**, pp.497-518.

Kosaka, Y., Saikia, B., Mingki, T., Tag, H., Riba, T. and Ando, K., 2010. Roadside distribution patterns of invasive alien plants along an altitudinal gradient in Arunachal Himalaya, India. *Mountain Research and Development*, **30**(3), pp.252-258. https://doi.org/10.1659/MRD-JOURNAL-D-10-00036.1

Lamsal, P., Kumar, L., Aryal, A. and Atreya, K., 2018. Invasive alien plant species dynamics in the Himalayan region under climate change. *Ambio*, **47**(6), pp.697-710. https://doi.org/10.1007/s13280-018-1017-z

Langeland, K.A., 2008. *Identification and biology of non-native plants in Florida's natural areas*. Gainesville, Florida, USA: University of Florida IFAS Extension.

Lowe, S., Browne, M., Boudjelas, S. and De Poorter, M., 2000. *100 of the world's worst invasive alien species: a selection from the global invasive species database*. Auckland: Invasive Species Specialist Group, **12**.

Mack, R.N., Simberloff, D., Mark Lonsdale, W., Evans, H., Clout, M. and Bazzaz, F.A., 2000. Biotic invasions: causes, epidemiology, global consequences, and control. *Ecological Applications*, **10**(3), pp.689-710. https://doi.org/10.1890/1051-0761(2000)010[0689:BICEGC]2.0.CO;2

Malik, Z.A., Pandey, R. and Bhatt, A.B., 2016. Anthropogenic disturbances and their impact on vegetation in Western Himalaya, India. *Journal of Mountain Science*, **13**(1), pp.69-82. https://doi.org/10.1007/s11629-015-3533-7

Medal, J., Overholt, W., Charudattan, R., Mullahey, J., Gaskala, R., Diaz, R. and Cuda, J., 2012. Tropical soda apple management plan. *UF/IFAS and FDACS/DPI, Gainesville FL*.

Medvecka, J., Jarolimek, I., Hegedusova, K., Skodova, I., Bazalova, D., Botkova, K. and Sibikova, M., 2018. Forest habitat invasions–Who with whom, where and why. *Forest Ecology and Management*, **409**, pp.468-478. https://doi.org/10.1016/j.foreco.2017.08.038

Mishra, R., 1968. *Ecology Work Book*. Oxford and IBH Publishing Co. *New Delhi*, p.624.

Motooka, P., 2003. Weeds of Hawaii's pastures and natural areas; an identification and management guide. In Weeds of Hawaii's pastures and natural areas; an identification and management guide. USA: College of Tropical Agriculture and Human Resources, University of Hawaii.

Mueller-Dombois, D. and Ellenberg, H., 1974. *Aims and methods of vegetation ecology*. New York: Wiley, p.547.

Nee, M., Hawkes, J.G., Lester, R.N. and Estrada, N., 1991. Synopsis of *Solanum* Section Ancanthophora: a revision of interest for glyco-alkaloids. *Solanaceae III: Taxonomy, Chemistry, Evolution. Kew, Richmond, Surrey, UK: Royal Botanic Gardens*, pp.258-266.

Osaki, S., Wasaki, J. and Nakatsubo, T., 2022. Phenological shifts of the invasive annual weed *Bidens pilosa* var. *pilosa* in response to warmer temperature. *Plant Ecology*, **223**(10), pp.1155-1165. https://doi.org/10.1007/s11258-022-01262-5

Parker, C., 1992. *Weeds of Bhutan*. Economic Botany. New York, USA: Springer. https://doi.org/10.5962/bhl.title.120425

Pathak, R., Negi, V.S., Yadava, A.K. and Bhatt, I.D., 2021. Distribution pattern of dominant invasive alien plants in forests of Kumaon region in West Himalaya. *International Journal of Ecology and Environmental Sciences*, **47**(4), pp.325-332.

Rahbek, C., 2005. The role of spatial scale and the perception of large-scale species-richness patterns. *Ecology letters*, **8**(2), pp.224-239. https://doi.org/10.1111/j.1461-0248.2004.00701.x

Rai, U., 2006. *Characterisation of plant biodiversity in Darjiling Hills using remote sensing techniques*. PhD thesis, University of North Bengal, West Bengal, India.

Rameshprabu, N. and Swamy, P.S., 2015. Prediction of environmental suitability for invasion of Mikania micrantha in India by species distribution modelling. *Journal of Environmental Biology*, **36**(3), p.565.

Randall, R.P., 2012. *A global compendium of weeds*, **2**, p.1124.

Rao, R.R., 1977. Changing pattern in the Indian flora. *Nelumbo-The Bulletin of the Botanical Survey of India*, pp.156-166. https://doi.org/10.20324/nelumbo/v19/1977/75568

Ray, D., Behera, M.D. and Jacob, J., 2019. Comparing invasiveness of native and non-native species under changing climate in North-East India: ecological niche modelling with plant types differing in biogeographic origin. *Environmental Monitoring and Assessment*, **191**, pp.1-13. https://doi.org/10.1007/s10661-019-7685-8

Reddy, C.S., 2008. Catalogue of invasive alien flora of India. *Life science journal*, **5**(2), pp.84-89.

Saha, D. and Sundriyal, R.C., 2012. Utilization of non-timber forest products in humid tropics: Implications for management and livelihood. *Forest Policy and Economics*, **14**(1), pp.28-40. https://doi.org/10.1016/j.forpol.2011.07.008

Sandilyan, S., 2015. Highways of India: A heaven to invasive alien plants. *Science India*, **18**(12), pp.33-36.

Singh, H.B., Kumar, A. and Adhikari, B.S., 2021. Status of invasive alien plant species (IAPs) in Arunachal Pradesh, India: a review. *Int J Researches Biosci Agric Technol*, pp.604-613.

Singh, S.P., Khanna, K.R. and Sudhir, S., 1998. Breeding of *Solanum viarum*: current status as steroid bearing plant. *Journal of Medicinal and Aromatic Plant Sciences*, **20**(2), pp.423-431.

Singh, T.P., Singh, S., Roy, P.S. and Rao, B.S.P., 2002. Vegetation mapping and characterization in West Siang District of Arunachal

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THIS IS TO CERTIFY THAT, **MR. HUIDROM BABLIN SINGH** OF **WILDLIFE INSTITUTE OF INDIA, UTTARAKHAND, INDIA,** HAS ACTIVELY PARTICIPATED IN INTERNATIONAL CONFERENCE ON CLIMATE CHANGE AND ITS EFFECT ON ENVIRONMENT, AGRICULTURE, MANAGEMENT, HEALTH AND SOCIETY HELD ON 13^{th}-14^{TH} NOV. 2024 AND PRESENTED ARTICLE ENTITLED **IMPACT OF INVASIVE ALIEN PLANT SPECIES ON SOIL PROPERTIES IN THE PAKKE TIGER RESERVE, ARUNACHAL PRADESH**, VIA **ORAL** PRESENTATION.

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Date: 16th March, 2024

BEST PAPER AWARD

This is to certify that Dr. / Mr. / Mrs./ Ms. Huidrom Bablin Singh from Wildlife Institute of India participated & presented a [✓ paper] / poster entitled "Dominant invasive plant distribution in different forest types of Pakke Tiger Reserve, Arunachal Pradesh" in the Technical Session III He/She secured Best Position in [✓ Oral] / Poster Presentation in the Male/Female category in the National Seminar held on 16th March 2024 at D.A.V. (P.G.) College, Dehradun.

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