Satish Chandra Garkoti
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Tropical Ecosystems: Structure, Functions and Challenges in the Face of Global Change

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Preface

Today, the world is facing a labyrinth of environmental problems. Unprecedented increases in human population, consumerism, infrastructure development and a myriad of other anthropogenic activities are bringing about widespread changes in the ecosystem structure and processes throughout the globe. An entire range of human-driven ecological and socio-economic changes often termed ‘global change’ are complex, deep and accelerating, and the impacts of these on ecosystems and people, in the majority of cases, remain unpredictable. The global change factors impacting all regions in general and tropics, where the ecosystems are known for nurturing the highest level of biodiversity, in particular, are discernible in terms of the degradation of ecosystems, loss of biodiversity, habitat fragmentation, biological invasion and changes in climate. These global change factors are redefining the human-nature relationship in recent times. Societies, especially those living in developing countries in the tropics, will require multiple yet urgent responses to address as well as to adapt to global change affecting them directly as well as indirectly. Amidst the increasing magnitude and pace of the various global change factors, scientists are compelled to chart unfamiliar frontiers of research in ecology and environmental sciences to develop the necessary knowledge base. The understanding will help in taking measures, well in advance, to minimise the cumulative impacts of global change on the entire web of life that exists on the planet.

Among the global change factors, climate change is affecting almost all aspects of the earth system. Being the storehouse of biodiversity, the impacts of the factors, such as land-use change, induced deforestation and biological invasion, and most importantly that of climate change are now well recorded on tropical ecosystems, most notably on forests. Tropical forests, besides supporting the livelihood needs of a huge number (~40% of the total global population) of impoverished people, are also endowed with the largest reservoir of carbon pool than any other terrestrial biomes. Considering the important role of tropical forests in providing a range of vital ecosystem services, including carbon sequestration and at the same time the great amount of pressure on their very existence, these have been drawing much attention from the ecologists and resource managers worldwide.

Over the last two decades, enormous global efforts have gone into understanding the causes and impacts of global change on ecosystems and human societies. Efforts have also been made in developing strategies for mitigation and adaptation of the impacts. Some of the key global change programmes include the International

Council for Science (ICSU)-sponsored International Programme of Biodiversity Science (DIVERSITAS), International Geosphere-Biosphere Programme (IGBP), International Human Dimensions Programme of Global Environmental Change, World Climate Research Programme, Earth System Science Partnership and Programme on Ecosystem Change and Society. On the other hand, the International Society for Tropical Ecology (ISTE) and the Association for Tropical Biology and Conservation (ATBC) have been promoting research and education on the tropical ecosystems of the world, while the European Commission has been supporting Erasmus Mundus Masters Course in Tropical Biodiversity and Ecosystems, to name a few among other programmes focussing on tropical ecosystems internationally. All these programmes have greatly enhanced our scientific understanding about several complex and interrelated facets of the earth system and the global change impacts on it. However, we still have much to understand about critical tipping points across ecosystem types, particularly in the wake of projected scenarios for tropical ecosystems in IPCC's most recently released report in October 2018. Hence, there is much to be learned to in order to respond to the multiple challenges that the various components of global change present to sustainable management of ecosystems.

With the above backdrop, Jawaharlal Nehru University (JNU) in collaboration with the International Society for Tropical Ecology (ISTE) organised the 14th International Tropical Ecological Congress (ITEC-14) on the theme 'Tropical Ecosystems in a Changing World' at Jawaharlal Nehru University (JNU), New Delhi, during 10 and 12 December 2014. The key objectives of the Congress were to critically evaluate the status of the current research, develop a broad understanding on tropical ecosystems in the face of global change and suggest a road map for future research.

The present volume represents a small but important step towards bringing together the current research findings on various global change themes presented and discussed during ITEC-14 by a number of scientists working on some lesser explored tropical/subtropical ecosystems in India, Nepal, Indonesia and more specifically in the Himalayan region which is identified as one the most vulnerable ecosystems of the world from the climate change standpoint. A total of 16 contributions included in this volume cover a diverse range of global change themes, such as the impacts of changing temperature and precipitation on soil ecosystem, forest degradation, extent and impacts of invasive species, response of plants to pollution, climate change impacts on biodiversity and tree phenology, land use-led environmental changes, importance of traditional knowledge in climate change adaptation, timberline ecosystems and role of integrated landscape modelling for sustainable management of natural resources. Most of the research topics included in the volume present state-of-the-art information on global change impacts on the terrestrial ecosystems in the tropics and most prominently on the Himalayan ecosystem. Our challenge was to present as many as 16 discrete contributions in terms of research themes and cosmopolitan essence in one volume without losing coherence. The impacts of one or the other global change factor, particularly climate change, on a range of tropical ecosystems described in majority of the chapters helped us in establishing the desired linkage.

The book is a collective endeavour of an international multidisciplinary group of scientists made to serve the purpose of improving our understanding on the impacts of global change on the structure and functioning of tropical ecosystems and hence towards addressing the challenges of their sustainable management. We hope that the book will help the researchers working in the areas of ecology and environmental science to update their knowledge and natural resource managers and policy planners to find explanations to some of their assumptions and observations on multiple global change factors impacting tropical ecosystems inclusive of Himalayan ecosystem. For us, presenting the book in this form would not have been possible without the support received from a number of individuals and organisations. We sincerely thank all the contributors for patiently cooperating through the editorial process to meet the requirements of the publisher. With great pleasure, we would like to thank all the organisations in India whose financial support enabled us in organising ITEC 2014: National Biodiversity Authority and Mountain Division of the Ministry of Environment, Forest and Climate Change, Department of Science and Technology, Indian Council of Agricultural Research, Indian National Science Academy and Council of Scientific and Industrial Research. We thank Shri Hem Pande, former secretary, Government of India, and to the authorities of Jawaharlal Nehru University for the vital support in various ways in organising the Congress. Thanks are also due to the students at the School of Environmental Sciences, JNU, specially Binay Sa, Rajendra Joshi, Bikas Mishra, Mayank Krishna, Dr. Sonam Wangmo, Dr. Anju Kumari, Ms. Chandrabala, Vivek Singh and Abhishek Verma, for their cooperation and assistance during ITEC-2014 and to Dr. Nepolian Borah and Ambuj Mishra for their assistance in preparation of the book. We are deeply indebted to Ms. Akansha, Mr. Shukla and Mr. John Ramkumar at Springer, India, for their support and forbearance. The views expressed and findings presented in this volume are those of the respective authors and do not reflect in any manner to those of the organisations they are affiliated to.

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Contents

Introduction

1

Satish Chandra Garkoti and Rajeev L. Semwal

Worldwide today we live in an era often termed as “Anthropocene” which has no antecedence considering the extraordinary human footprints on ecosystems, particularly that on biodiversity-rich tropical ecosystems, in the entire history of mankind. Understanding the implications of such impacts on the structure and functioning of ecosystems is important from the standpoint of developing appropriate strategies that help address the pressing issues impinging on the state of sustainable management of ecosystems in the tropics.

The tropical region of the globe located between vast swathes of trpoics of Cancer and Capricon accounts ~40% of the Earth’s surface area encompasses a range of ecosystem types. These ecosystems such as rain forests, seasonal forests, dry deciduous forests, spiny forests, savannah, deserts, marine, and other habitat types including mountains support the highest level of biodiversity globally and play an important role in global processes and socioeconomic concerns (Whitmore 1990; Frankie et al. 2004). Across ecosystem types, tropical forests though covering only 15% of the world’s surface area represent two-thirds of the plant biomass, over half of the total global biomass carbon which is equivalent to one-third of global net primary productivity. The aforementioned attributes enable tropical forests to contribute significantly in atmospheric CO_2 exchange and balance in functional terms than any other terrestrial biome (Singh and Singh 1991; Beer et al. 2010; Pan et al. 2011, 2013). In addition to carbon sequestration, tropical forests provide a range of other important ecosystem services such as serving habitats for myriad plants and animal species including the endangered ones, helping in conserving important features of the global hydrological cycle, and cushioning extensive catchment areas of rivers in the tropics (Gibbs et al. 2010). The biological wealth of these forests not only contributes in sustaining the livelihoods of nearly

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40% of the peoples living in developing tropical countries but also to many more all over the globe. However, forests throughout the globe are dwindling fast under human pressure. The tropical forests notably the rainforests which are significant from biodiversity and species endemism viewpoint are declining at alarming rates. In 1990 the global forest area was 4128 million ha, corresponding approximately 30% of the Earth's land area, which has declined to 3999 million ha in 2015. The area under natural forests, accounting for little over 90% of the total forest area, has been recorded decreasing at the rate of 6.5 million ha per year between 2010 and 2015 mainly due to expanding agricultural land use in the tropics (FAO 2015). While forest area continues to decline, the human population is increasing at a rapid pace putting subsistence and livelihoods of more than 200 million of the world's impoverished people in jeopardy (sCBD 2010).

Taking into account that the global requirement for food is expected to rise nearly 50% by 2050, the agricultural expansion trends observed in the period of 1980–2000 are likely to remain unchanged in present times as well implying that the forests all over the tropics will continue to be cleared at much higher rate for such conversion. This can be corroborated from the fact that between 1995 and 2007 period, transformation of tropical forests for agriculture increased by 17% in developing tropics, whereas it decreased by 34% in developed countries (FAO 2009; Mollicone et al. 2007). Thus, large-scale agricultural expansion has been identified as the major driver of deforestation throughout the tropics (Morton et al. 2006; Nepstad et al. 2008; Hansen et al. 2008; Butler and Laurance 2009). The need of the hour is to urgently find a balance between meeting the growing global requirement of food and the services provided by intact tropical forests in this fast-changing world when local and regional economies are merging with global economy at equal pace.

In addition to land use change-induced deforestation, several global change drivers such as climate change, biological invasion, and habitat fragmentation have also been identified as responsible for drastically altering the dynamics of the tropical ecosystems (Millennium

Ecosystem Assessment 2005; Morris 2010). Among these, from the climate change perspective alone, the carbon stocks in the global forest area have already declined by 11 gigatons in the last 25 years (FAO 2015). Anthropogenic emissions of greenhouse gases are responsible for the warming of the climate system over preindustrial levels. The global mean surface temperature (GMST) has been recorded 0.87°C higher for the period 2006–2015 than it was between 1850 and 1900 period. In the last 1400 years, the recent three decades from 1983 to 2012 represent the warmest period in the entire northern hemisphere leading to the decline in the amounts of snow/ice and sea level rise (IPCC 2014).

As the warming is currently increasing at 0.2 °C per decade due to historical and continuing emissions, it will continue and keep impacting the climate system for centuries to come. And if the GMST were to touch 1.5 °C above preindustrial average, the situation will lead to warming up in midlatitude regions by 3 °C (IPCC 2018). The impacts from global warming on ecosystems have already been visible in terms of changing/diminishing trends in some of the services provided by these and causing irreversible disintegration in some of the ecosystem types (IPCC 2018). With global warming of 1.5 °C, the projected manifestations include wide-ranging changes in weather patterns, shifts in climatic zones and ecosystems, habitat destruction, water scarcity, flooding, intense heat waves and extreme precipitation, increase in the intensity and frequency of wild fires, and decline in crop yields leading to cascading risks to health, livelihoods, food security, water supply, human displacement, and economic growth (IPCC 2018). Change in conditions linked to warming of climate will threaten ecosystems with narrow geographic ranges and those with other peculiar attributes such as biodiversity hotspots, coral reefs, mountain glaciers, small islands, etc. If increase in global average temperature exceeds 1.5 °C, this would influence plant and wildlife species as a sizable number of these are most likely to lose nearly half of their natural distribution ranges rendering approximately 20–30% species at the risk of extinction (IPCC 2018). Apprehensions continue to escalate about the growing emissions of GHGs when tropical forests are constantly degrading, cleared, and frequently burned for agricultural extensification responsible for about 10% of net global carbon emissions (Gibbs et al. 2010).

Being fragile, the forest ecosystems are facing the greatest anthropogenic pressure and threats from changing climate on the whole life support systems in the tropical regions. The United Nations Sustainable Development Goals (SDGs), 2015, emphasize on poverty alleviation, minimizing social and economic disparities, and climate action as indispensables for responding to the challenges that climate change presents to human societies across the globe. To minimze risks to natural and human- managed ecosystems arising from multiple global change factos, measures that help avoid deforestation, support restoration and rehabilitation of degraded ecosystems, manage biodiversity, and take advantage of traditional knowledge systems have been advocated in recent IPCC special report (IPCC 2018). REDD+ (Reducing Emissions from Deforestation and Forest Degradation, Sustainable Management of Forests), an incentive-based framework, is already in place under the United Nations Framework Convention on Climate Change (UNFCCC) since 2005 to support sustainable forest management in developing countries (ReAchard et al. 2007; Parker et al. 2009; Brunner et al. 2010); such a multifaceted crisis requires developing holistic understanding on the structural and functional attributes of the tropical ecosystems on a constant basis. However, developing such a research framework is a challenge for the ecologists in general and those working in the tropics in particular. Fortunately, the International Society for Tropical Ecology (ISTE) was established way back in 1960. The ISTE is exclusively dedicated to promote ecological research in the tropics and has been making concerted efforts to help ecologists to respond to the challenge in substantial terms. Since its establishment, the ISTE routinely organizes the Tropical Ecology Congress (TEC) in which leading ecologists interact, present their research findings, and deliberate on the future research agenda on tropical ecosystems. Jawaharlal Nehru University (JNU), New Delhi, hosted the 14th session of the TEC on the theme "Tropical Ecosystems in a Changing World" between 10 and 12 December 2014.

This volume has its genesis in TEC-14 attended by over 500 delegates representing as many as 17 countries. The volume presents state-of-the-art information derived from a select number of papers presented mainly during the TEC on a wide range of topics falling well within the theme of the congress. Furthermore, during the past four or more decades, ecology has already evolved as an integrated discipline where the emphasis is to link the natural science components with socioeconomic processes so as to find enduring answers to the critical problems. The chapters amply reflect on this approach while covering varied dimensions in tropical ecology across diverse ecosystems particularly that of the most sensitive ecosystem on the face of the Earth: the Himalayas. Moreover, when anthropogenic activities are bringing about drastic changes in tropical ecosystems, issues related to the genesis of the crisis are also dealt by many authors in their respective chapters. Therefore, the volume attempts to bring together research topics having a broad focus on human and climate change influence on the terrestrial ecosystems in the tropics in general and more specifically from the most significant and vulnerable Himalayan ecosystem. The chapters, all 16 in number, are presented in a coherent manner where various dimensions including adaptation and mitigation related to climate change serve as the common denominator across majority of them.

Accordingly, the long-term consequences of global warming induced changes in the temperature and precipitation regimes on the ecological functioning of tropical soil ecosystems, and hence forests were reviewed by Afreen and coauthors in Chap. 2. The authors highlighted that decrease in precipitation would increase the deciduousness in the tropical vegetation which may lead to significant changes in nutrient cycling. Under such a situation, though nutrient use efficiency of the plants may possibly increase, there would be declines in litter decomposition, nutrient mineralization rates, soil carbon, concentration of nitrogen and phosphorus, and fluxes of CO_2, N, and P including microbial population and its activity.

In Chap. 3, Singh and others explored how the relentless collection of firewood, tree leaf fodder, and leaf litter from forest floor by local people for subsistence is one of the main causes of degradation of forests in the Himalayas. Citing specific data from oak (*Quercus leucotrichophora*) and chir pine (*Pinus roxburghii*) forests of central Himalaya, the authors described that such degradation leads to 40–50% decline in biomass and about 80% decline in net primary productivity responsible for disrupting structural and functional attributes of these forests. Terming the constant removal of biomass as “chronic forest disturbance,” with the process of recovery yet to be investigated, the authors surmise that it is likely to be slower than that of clear-felled forests, described as forests subjected to “acute disturbance.” The study has policy implications for the Himalayan forests where green felling has been banned for about the last four decades and therefore warrants detailed scientific enquiry.

Biological invasion is an integral component of global change posing serious threat to the integrity of ecosystems in the tropics. Shrestha has synthesized the existing knowledge and practices on the management of invasive alien plants (IAPs) in Nepal in Chap. 4. He provided a list of 183 exotic vascular plant species naturalized in Nepal among which 26 have turned invasive. Until now the invasive species have invaded varied land use types extensively from tropical to temperate zones except *Myriophyllum aquaticum*, *Mikania micrantha* and *Chromolaena odorata* still confined to specific geographical locations of the country. IAPs have already been causing habitat degradation, species displacement and impacting livelihoods of farming communities in Nepal. Among several methods employed for controlling the proliferation of IAPs globally including in Nepal, the author has specifically laid emphasis on regional collaboration to enhance the effectiveness of other control measures. Similarly, adoption of ecosystem approach to understand the phenomenon of plant invasion and to tackle the spread of alien invasive species across central India rather than species-specific approach has been suggested in Chap. 5 by Dar et al. The authors have provided a conceptual basis for improving general understanding on invasive species and their impact on tropical forest ecosystems of central India.

Borah and coauthors elucidated how aboveground carbon stocks varied under different levels of anthropogenic disturbance in wet tropical forests of Assam, India, in Chap. 6. Older trees of higher girth class were the major contributors in building carbon stock in wet tropical forests rather than the young trees of smaller girth class irrespective of level of disturbance. The study revealed that the removal of trees of higher girth class affected carbon stock significantly negatively, while the young trees helped in carbon sequestration across disturbance regimes. The authors demonstrated that from the carbon stock point of view in wet tropical forests among the phytosociological attributes studied, basal cover of trees influenced carbon stock more pronouncedly than tree density, diversity, and species richness.

In Chap. 7, Mukherjee et al. analyzed responses of tropical and subtropical plants to air pollution. They showed that in addition to other anthropogenic pressures on resources, air pollution has drastically altered the dynamics of the tropical ecosystems. In most of the tropical regions, air pollutants, viz. particulate matter, ozone, nitrogen deposition, and fossil hydrocarbons, have been identified impairing the growth and development of plants while forcing change in their morphology, physiology, and functional traits. The authors emphasized upon the need of developing future air quality scenarios as per the projected emissions patterns in order to devise suitable response for minimizing the impacts of air pollution on tropical vegetation.

In the tropical dry deciduous forests of central India, sal (*Shorea robusta*) is a highly valued timber species. However, sal forests in the vicinity of human settlements are often overexploited for fuel wood and construction material leading to their degradation. Knauf et al. in Chap. 8 described the impacts of harvesting of such forest goods in a few sal forest sites in the central Indian state of Chhattisgarh. The authors found that *S. robusta* failed to regenerate on highly impacted sites and suggested environmental education, provision of alternative of forest resources, and testing of assisted natural regeneration and incentive-based mechanisms for the sustainable management of sal forests located specifically close to human habitations in India.

Adopting a landscape approach and combining field survey and remote sensing methods, Rai and coauthors reported total area, dominant tree species, and altitudinal range of the treeline ecotone in western Himalaya in Chap. 9. Despite a reported upward shift in treeline/timberline species at few locations in the Himalayas in the wake of climate change, the study recoded no altitudinal change in the upper treeline during the last 40 years. Considering the immense ecological role of birch (*Betula utilis*) as a prominent timberline tree species in the Himalayas in terms of environmental stability and providing habitat for several endangered species of fauna, the authors stressed on the need of participatory monitoring of the species using citizen science approach.

Treeline or timberline, the high-altitude limit of forests and trees, despite subtle difference in them, are often referred to interchangeably in the scientific literature, contended Singh and Singh in Chap. 10. In the Himalayan region, though spread over several hundred km in length and being the highest in the entire Northern Hemisphere, treelines/timberlines are least studied due to remoteness. Based on existing literature and experiences gained through an extensive ongoing study on mapping the treelines in the Indian Himalayas at a regional scale, the authors stated that treelines are higher, generally by 50–200 m, than timberline in the region. They also proposed methods and ideas which could be applied to study treelines/timberline in the context of climate change in the Himalayas. In addition to giving importance to temperature lapse rate in treeline/timberline studies, the authors underlined the need of including aspects such as tree ring width changes in relation to climate, tree water relations, phenological changes in plants and species distribution, and mapping of population dynamics of not only trees but associated herbaceous species for developing comprehensive understanding on the climate change impacts in the Himalayas.

Maikhuri et al. provided insights on the importance of traditional knowledge to successfully adapt to climate change impacts on traditional farming, the key livelihood earning activity of more than two-thirds of the population of Uttarakhand Himalaya in Chap. 11. Based on extensive survey, the authors demonstrated that integration of local perceptions and traditional farming knowledge in formal adaptation and mitigation frameworks would be more effective to respond to climate change-induced weather variability. They expressed the need of supplementing the efforts through capacity building of farmers and value addition in their crop products for the sustainability of the farming systems in Uttarakhand Himalaya.

In Nepal the tropical zone comprising foothills and Gangetic Plains, once known for its extensive biodiversity-rich dense tropical forests, has been witnessing drastic environmental degradation in recent times. In Chap. 12 Jha and colleagues pointed out change in land use, unprecedented increase in human population, and climate change-induced temperature rise and erratic rainfall patterns during the last four decades as the major factors responsible for the degradation being witnessed in the zone. The authors recommended a stronger political will, sound environment, and development policies and their effective implementation for maintaining ecological balance with development in the tropical zone of Nepal.

Rawat and Rawal in Chap. 13 highlighted the extent and diversity of Himalayan forests and argued that as such these cannot be categorized either with tropical or temperate forests of the world. The Himalayan forests, though structurally somewhat resembling tropical forests, show affinity with temperate forests in their functioning. Based on a study spanning over more than two decades on diverse forest communities in parts of Nanda Devi Biosphere Reserve in Uttarakhand, the authors reported developing several sensitivity indices to depict change in vegetation composition signifying the need for differential conservation and management planning of Himalayan forests which are generally treated uniformly with other forest types in the country. Diverse conservation and management scenarios for regional forests have been discussed in the chapter.

It has been anticipated that the richness of endemic species with restricted distribution of the Himalayan region, which is 1 of 34 global biodiversity hotspots, is highly vulnerable under the changing climate. However, deficiency in long-term data sets from the region poses a serious limitation in developing suitable conservation strategies to respond to the emerging challenges. In Chap. 14, Negi and Rawal have provided an overview of research-based evidences of the impacts of climate change on flora and fauna of the Himalayan region and suggested some mitigation measures for biodiversity conservation.

In Chap. 15, Hartiningtias and others reported phenological patterns and community structure of tropical lowland dipterocarp forests of Bukit Barisan Selatan National Park of Indonesia. While highlighting the peculiar phenological trait of sub-annual flowering observed in these forests of the park from other dipterocarp forests, the authors identified excessive logging and frequent forest fires as major threats to their existence. They underlined the need and importance of conservation of the unique populations of dipterocarp species lacking the evolutionary advantage of mass flowering and fruiting to minimize mortality when some of these have already been categorized as critically endangered and vulnerable.

In order to address the complexities involved in conservation of natural resources and socioeconomic development planning at landscape scale, modelling has emerged as an important tool in recent times.

In Chap. 16, Goswami and others have examined the feasibility of using various spatiotemporal models in microscale sustainable landscape planning in Indian context. The authors have identified LandSHIFT, a spatially explicit model that uses mid-to-long-term scenarios of land use and land cover changes involving multiple biophysical and socioeconomic factors with some suggested minor modifications, as the most suitable among the existing models for the purpose in Indian context.

Chapter 17 summarized the new findings of the authors which may help in minimizing the impact of global change and also help in prioritizing future research in the tropics.

Lastly we hope that the book should help readers such as environmental science researchers, natural resource managers, and policy planners in enriching and updating their knowledge on multiple ecological issues facing tropical ecosystems especially the Himalayan ecosystem in most recent times.

References

Beer C, Reichstein M, Tomelleri E, Ciais P, Jung M, Carvalhais N, Rodenbeck C, Arain AM, Baldocchi D, Bonan GB, Bondeau A, Cescatti A, Lasslop G, Lindroth A, Lomas M, Luyssaert S, Margolis H, Oleson KW, Roupsard O, Veenendaal E, Viovy N, Williams C, Woodward FI, Papale D (2010) Terrestrial gross carbon dioxide uptake: global distribution and covariation with climate. Science 329:834–838. https://doi.org/10.1126/science.1184984

Brunner R, Ali F, Calvo Amberl C, Aquino P, Bautista K, Bendandi B, Karpati A, Lai K, Barbour McKellar J, Roos M, Rudder N, Soergel E, Valencia N (2010) Back to its roots: REDD+ via the Copenhagen Accord. Reconsidering Dev 1(1):1–10

Butler R, Laurance W (2009) Is oil palm the next emerging threat to the Amazon? Trop Conserv Sci 2:1–10

FAO (2009) How to feed the world in 2050. In: Proceedings of the Expert Meeting FAO Headquarters. FAO, Rome

FAO (2015) Global forest resources assessment. How are the world’s forests changing? 2nd edn. FAO, Rome 2016

Frankie GW, Mata A, Vinson SB (2004) Biodiversity conservation in Costa Rica: learning the lessons in a seasonal dry forest. University of California Press, Berkeley, 352pp

Gibbs HK, Ruesch AS, Achard F, Clayton MK, Holmgren P, Ramankutty N, Foley JA (2010) Tropical forests were the primary sources of new agricultural land in the 1980s and 1990s. PNAS 107(38):16732–16737

Hansen MC, Stehman SV, Potapov PV, Loveland TR, Townshend JRG, DeFries RS, Pittman KW, Arunarwati B, Stolle F, Steininger MK, Carroll M, DiMiceli C (2008) Humid tropical forest clearing from 2000 to 2005 quantified by using multitemporal and multiresolution remotely sensed data. PNAS 105(27):9439–9444

IPCC, 2014: Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (eds.)]. IPCC, Geneva, Switzerland, 151 pp.

IPCC (2018) Special report on the impacts of global warming of 1.5°C above pre-industrial levels. Summary for policy makers. IPCC, Geneva, 33pp

Millennium Ecosystem Assessment, 2005. Ecosystems and Human Well-being: Synthesis. Island Press, Washington, DC.

Mollicone D, Achard F, Federici S, Eva HD, Grassi G, Belward A, Raes F, Seufert G, Stibig HJ, Matteucci G, Schulze ED (2007) An incentive mechanism for reducing emissions from conversion of intact and non-intact forests. Clim Change 83:477–493

Morris PJ (2010) Anthropogenic impacts on tropical forest biodiversity: a network structure and ecosystem functioning perspective. Philos Trans R Soc B 365:3709–3718. https://doi.org/10.1098/rstb.2010.0273

Morton DC, DeFries RS, Shimabukuro YE, Anderson LO, Arai E, Espirito-Santo FB, Freitas R, Morisette J (2006) Cropland expansion changes deforestation dynamics in the southern Brazilian Amazon. Proc Natl Acad Sci USA 103(39):14637–14641

Nepstad DC, Stickler CL, Soares-Filho B, Merry F (2008) Interactions among Amazon land use, forests and climate: prospects for a near-term forest tipping point. Philos Trans R Soc B 363:1737–1746

Pan Y, Birdsey RA, Fang J, Houghton R, Kauppi PE, Kurz WA, Phillips OL, Shvidenko A, Lewis SL, Canadell JG, Ciais P, Jackson RB, Pacala SW, McGuire AD, Piao SA, Rautiainen S, Hayes D (2011) A large and persistent carbon sink in the world’s forests. Science 333:988–993. https://doi.org/10.1146/annurev-ecolsys-110512-135914

Pan Y, Birdsey RA, Phillips OL, Jackson RB (2013) The structure, distribution, and biomass of the world’s forests. Annu Rev Ecol Evol Syst 44:593–622. https://doi.org/10.1146/annurev-ecolsys-110512-135914

Parker C, Mitchel A, Trivedi M, Mardas N, Sosis K (2009) The little REDD+ book. Global Canopy Programme, Oxford

ReAchard F, Federici S, Eva HD, Grassi G, Belward A, Raes F, Seufert G, Stibig HJ, Matteucci G, Schulze ED (2007) An incentive mechanism for reducing emissions from conversion of intact and non-intact forests. Clim Change 83:477–493

sCBD (2010) Strategic action plan for biodiversity 2011–2020 and the Aichi targets. Secretariat of the Convention on Biological Diversity, Montreal

Singh L, Singh JS (1991) Storage and flux of nutrients in a dry tropical forest in India. Ann Bot 68:275–284. https://doi.org/10.1093/oxfordjournals.aob.a088253

Whitmore TC (1990) An introduction to tropical rain forests. Clarendon Press, Oxford, pp xi + 226 pp

Influence of Changing Patterns of Precipitation and Temperature on Tropical Soil Ecosystem

2

Talat Afreen, Hema Singh, and J. S. Singh

Abstract

Tropical soils are nutrient limited and are markedly influenced by seasonality. The seasonal precipitation affects soil wet-dry cycle and has significant consequences for ecosystem functioning and processes. It has been predicted that in tropical countries, a substantial change will occur in the annual mean temperature and precipitation regimes in the future. However, we have little knowledge as to how the soil properties and processes will change with the change in the climatic conditions in tropics. In this chapter, we have tried to collate studies related to the effect of changing climatic condition (temperature and precipitation) on the soil in the tropics. The change in precipitation, than temperature, has more impact on the ecosystem processes in the tropics. Within the tropics the moist and dry ecosystems respond differently to the change in precipitation. Perhaps experimental studies using rain-out shelters may provide more precise information on the effect of the changing pattern of precipitation and drought on the health and vigour of tropical soils. There is a conspicuous lack of literature on the effect of temperature on tropical soils; for getting a complete picture and more precise information, we need more focused studies related to climate change in the tropical region.

Keywords

Dry ecosystem · Moist ecosystem · Precipitation · Seasonality · Soil properties · Tropics

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2.1 Introduction

We all have witnessed the changes in climatic conditions one way or the other. The mean global surface temperature has increased by 0.6 °C (Gitay et al. 2002) and is expected to increase up to 0.2 °C per decade (IPCC 2007). The increase in temperature could intensify the hydrological cycle and alter atmospheric circulation causing a change in precipitation pattern and atmospheric moisture content (Gitay et al. 2002). Climate change effects include decreases or increases in precipitation, increased interannual variability and increased frequency of intense wet and dry years (Easterling et al. 2000; Jentsch and Beierkuhnlein 2008; Singh et al. 2013; Smith 2011). The climate is now changing with greater pace than ever (Pachauri et al. 2014; Greve et al. 2014; Chadwick et al. 2016).

The rainfall pattern over tropical countries is quite erratic (Malhi et al. 2008; Feng et al. 2013) with marked uncertainty in the arrival, duration and intensity of seasonal precipitation. Many climate models have predicted a decrease in precipitation with an increase of dry periods (e.g. Maloney et al. 2014; Chadwick et al. 2016; Duffy et al. 2015). This is bound to affect the plant production and hence the soil. It has been reported that the Indian summer monsoon rainfall would increase in the future, as circulation of monsoonal will become weak (Flato et al. 2013). The altered precipitation affects soil water availability in terrestrial ecosystems including grassland and forest. Soil water availability controls the processes operating beneath the ground as well as above the ground and has long-term consequences on ecological functioning of the ecosystem (Austin et al. 2004). For example, studies indicate that with annual rainfall decrease or with the length and severity of the dry season, the forest deciduousness increase (Reich 1995; Borchert 1998; Condit et al. 2000; Bohlman 2010; Gond et al. 2013). The deciduousness affects rate of primary production as well as several aspects of nutrient cycling. Nutrient-poor soils are expected to favour species with a conservative strategy, i.e. with a long leaf life span and thus an evergreen leaf habit, while relatively nutrient-rich soils should favour acquisitive, fast-growing species with short leaf life span and a deciduous leaf habit to avoid the unfavourable growth period (Aerts 1995; Givnish 2002). Thus, in climates with seasonal drought, we should expect a trade-off between a deciduous habit enabling plants to cope with drought and an evergreen habit enabling them to cope with nutrient shortages.

The change in the climatic conditions could have significant consequences for the nutrient-depleted soils of the tropical region. But surprisingly, there is a lack of studies on the effect of changing climate on the soil properties and processes. The principal objective of this communication is to review the information on the effect of climate change, especially the change in temperature and precipitation on tropical soil ecosystem. This paper would help us to understand why the change in precipitation, than temperature, has more impact on the ecosystem processes in the tropics. It will be pertinent to study the accumulation, transformation and decomposition of soil organic matter for understanding the role of carbon pool of the soil in global carbon cycle under the changing climate. Soil ecosystems in the tropics are poorly studied despite the fact that one-third of the soils of the world are in the tropics and these support more than three-quarters of the world population (Hartemink 2002).

2.2 Soil and Its Importance

Soil provides the substratum for plant growth and is associated with the atmosphere through nutrient and hydrological cycling. The soil is considered as one of the most valuable natural capitals as it provides many of the vital services (Dominati et al. 2010) without whom imagination of life on earth is impossible. The soil has four essential components: organic, mineral, liquid and gases. Important soil properties are physical (soil temperature, soil moisture, texture, bulk density), chemical (nitrogen, carbon, phosphorus) and biological (microbial biomass, soil respiration, mineralization). Soil is intimately connected with plants, and this relationship generates an emergent property, viz. nutrient cycling which is the backbone of soil fertility and productivity. The soil also acts as a reservoir of water and nutrients, regulates climate through carbon sequestration and by regulating greenhouse gas (N_2O, CH_4) emissions, provides habitat for numerous species and physical support to plants and animals, etc. (Dominati et al. 2010). Thus, soil is involved with all the four important ecosystem services, viz. provisioning, regulating, cultural and supporting.

Soil organic matter, which is an important indicator of fertility and quality of the soil, is mainly composed of carbon and nitrogen (Brady and Weil 2008). According to several authors, changing climatic conditions are among the primary drivers that affect soil properties and processes (Dominati et al. 2010; Brevik 2013), and therefore, a major concern is to identify how climate change affects the concentration of carbon and nitrogen in the organic matter (Brevik 2013) which will in turn affect nutrient cycling.

The impact of climatic conditions would be different in different types of soil. Tropical soil, for example, stores more organic carbon (7.1 kgC m^{-2}) with a residence time of 10 years or less than temperate soil having organic carbon 5.2 kgC m^{-2} with a residence time of 10, 100 and 1000 years (Trumbore 1993). Hence tropical soil emits more soil CO_2 annually which is in the range of 1.9–5.5 kgC m^{-2} $year^{-1}$ than temperate land (0.22–0.45 kgC m^{-2} $year^{-1}$) (Trumbore 1993). This would indicate relatively low fertility of the tropical soil. However, relatively little information is available about the soil resource of the tropics compared to that of the temperate region (Hartemink 2002). Evidently, more research is needed to explore the effect of changing climatic condition such as an increase in temperature and variability in precipitation regime on tropical soils.

2.3 Effect of Temperature and Precipitation on Soil Processes

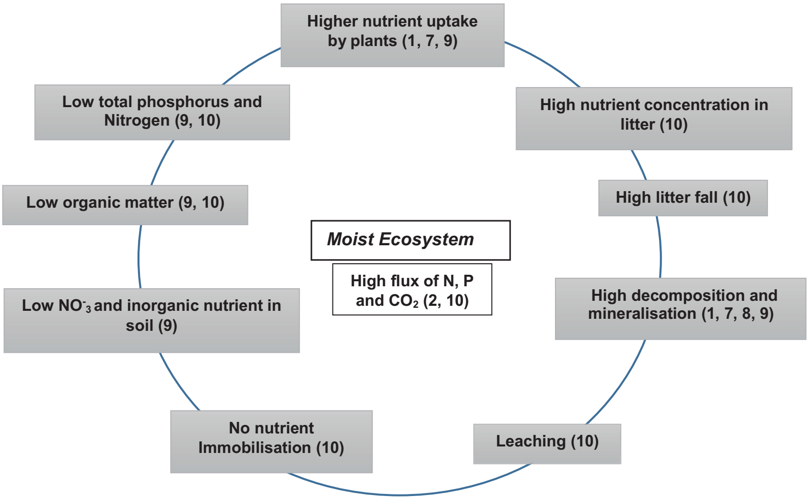
2.3.1 Nutrient Cycling

Nutrient cycling refers to the use and reuse of nutrients and involves movement or flow of nutrients from the physical environment to the biological system, i.e. living organisms and then back into the physical environment. The efficiency of the nutrient cycling of an ecosystem defines its productivity and nutrient loss through leaching and fluxes. Nutrient cycling involves important processes such as decomposition, mineralization and immobilization, which are important for regenerating ecosystem fertility by releasing nutrients for plants, all of which are affected by precipitation, temperature and plants. Moist and dry tropical ecosystems differ in several aspects of nutrient cycling (Fig. 2.1).

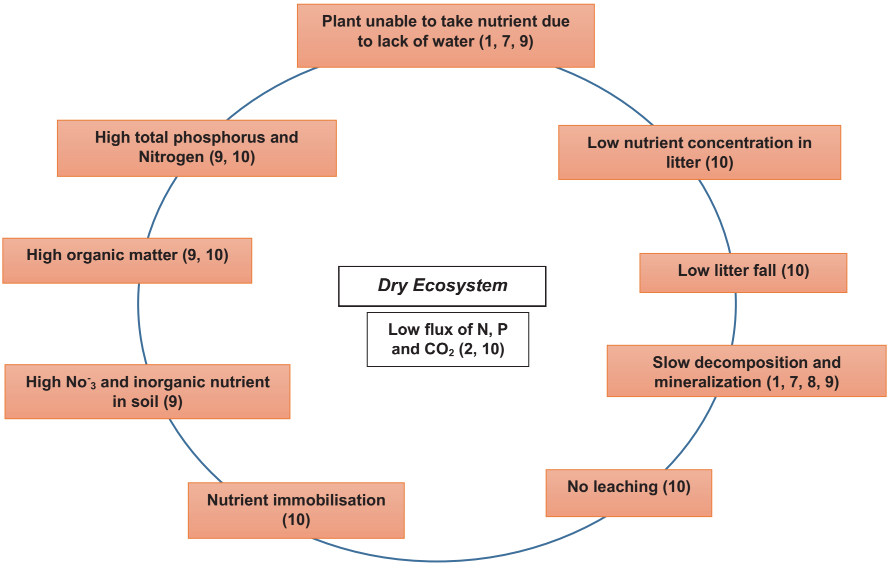
Such features as low range of mean annual precipitation (MAP), annual ratio of precipitation to potential evapotranspiration of less than 1.0, and 4–7 dry months in a year (Dirzo et al. 2011) control the biogeochemical cycles (Campo et al. 2001; Gei and Powers 2014; Verduzco et al. 2015) in the tropical dry forests. However, the potential effect of rainfall variability on biogeochemical processes in these water-stressed ecosystems has not been studied as extensively as for tropical wet forest. This lack of information limits our understanding of the potential effects of predicted increases in the frequency and duration of droughts and long-term reduction in mean annual precipitation (Meir and Pennington 2011).

Leaf nutrient resorption from senescing leaves and its efficiency is an important aspect of nutrient cycling. Foliar resorption can provide a substantial percentage of the nitrogen and phosphorus used annually by forests (Ryan and Bormann 1982). The study of Lal et al. (2001) indicated that resorbed nutrients, particularly N, P and K, could be a significant source of plant nutrient supply and are likely to meet a significant proportion of the nutrient demand of the developing leaves in the majority of species studied. As an example, the nutrients retranslocated from senescing leaves support 50–100% of leaf area development and 46–80% of leaf mass development in tropical deciduous species. This internal cycling, by supporting pre-monsoon leaf initiation and expansion, enables the trees to take full advantage of the rainy season when water and soil nutrients become abundant and support a high forest productivity. Resorption proficiency is a parameter describing the minimum level to which a nutrient is reduced during senescence and has also been used to quantify nutrient resorption (Killingbeck 1996). Higher proficiencies correspond to lower final nutrient concentrations in senesced leaves. Resorption proficiency and resorption efficiency continue to receive widespread attention in attempts to establish their relationship with soil fertility, plant nutritional status, plant functional groups and climate (Lal et al. 2001; Wright and Westoby 2003; Kobe et al. 2005; Rentería et al. 2005; Drenovsky and Richards 2006; Ratnam et al. 2008; Yuan and Chen 2009a, b).

The resorption efficiency of P, but not N, increased in years with low rainfall, suggesting that the costs of nutrient resorption relative to acquisition from soil differ



Negative effect of Rainfall, Inefficient nutrient cycling (3, 4, 5, 6, 8)



Positive effect of Rainfall, Efficient nutrient cycling (3, 4, 5, 6, 8)

Fig. 2.1 Nutrient cycling in tropical moist and dry ecosystems. N, P NO^-_3 and CO_2 represent the nitrogen phosphorus, nitrate and carbon dioxide, respectively. Numbers in each box represent the source: 1, Campo et al. (1998); 2, Harper et al. (2005); 3, Heisler-White et al. (2008); 4, Knapp et al. (2008); 5, Heisler-White et al. (2009); 6, Thomey et al. (2011); 7, Bejarano et al. (2014); 8, Sala et al. (2015); 9, Campo (2016); 10, Campo and Merino (2016) between N and P and that P conservation increases when rainfall decreases (Rentería and Jaramillo 2011) indicating that water availability controls P dynamics in the Chamela Dry Tropical Forest. Earlier, Rentería et al. (2005), working with six dry deciduous tree species, reported that the levels of green and senesced leaf P, but not N, change in response to topographic-related changes in nutrient and water availability due to changes in annual rainfall. These authors suggested that water more than soil nutrient availability controls nutrient resorption.

The study of Singh and Singh (1991a, b, 1993) revealed that the tropical deciduous forest is characterised by three levels of superimposed mineral cycling, viz. internal (leaf to stem and stem to leaf, i.e. reuse of resorbed nutrients), short-term through short-lived components (i.e. through leaf litterfall and fine roots to soil and then back to plant after release by decomposition) and long-term through long-lived components (through the fall of branch, bole and course root litter and their decomposition).

Campo (2016) found that nutrient use efficiency increased with reduced mean annual precipitation (MAP); N and P are cycled more efficiently with greater water deficit (N use efficiency increased from 56 at the subhumid site to 71 at the semiarid site, and P use efficiency increased from 720 at the subhumid site to 1205 at the semiarid site). Large increases in P-use efficiency (an increase of 40%) than in N-use efficiency (an increase of 21%) with decrease in MAP show that the extent of the dry season and/or intensity of drought may have created P limitation in these ecosystems (Campo and Vázquez-Yanes 2004). In the TDF of the Yucatan Peninsula where soil pH is more than neutral, P may be chemically bound to calcium (Gamboa et al. 2010; Cuevas et al. 2013), and this condition would be pronounced in drier sites, where shallow soils and slow P release create bad P conditions.

2.3.2 Decomposition and Mineralization

Decomposition and mineralization are two essential processes by which organic substance such as leaf litter, plant debris, dead organisms, etc. are broken down into simpler organic and inorganic materials. Decomposition is mainly a biological process, and its speed is governed by three major factors such as physical environment like soil temperature and moisture, soil organisms, and quality of soil organic matter (Brussaard 1994).

Temperature has been argued to influence the decomposition rate more in temperate climate, and precipitation has a greater impact in tropical climate where it is considered a primary driver for decomposition (Powers et al. 2009; Berg et al. 1993; Gholz et al. 2000; Trofymow et al. 2002). Soil temperature and soil moisture are predominantly governed by climatic conditions and seasonality. Decomposition rate is higher in tropical than in temperate regions (Powers et al. 2009). In determining decomposition rate, rainfall becomes crucial as it influences soil temperature and soil moisture (Salamanca et al. 2003).

Studies have reported that timing and magnitude of precipitation events markedly affect the decomposition processes (Anaya et al. 2012), and annual decay rate consistently increases with the increase in precipitation in tropical ecosystems (Campo and Merino 2016). In a pantropical study, Powers et al. (2009) found that the average decomposition rate and the ratio of above- and belowground decay increase linearly with the annual precipitation. A study on decay dynamics reported the inhibitory effect of excessive addition of water on the decomposition rate in mesic to wet climatic conditions (Schuur 2001). It might probably be due to the anaerobic condition created by water abundance. Precipitation affects decomposition rate both directly and indirectly. It affects directly by affecting microbial population and mesofauna and their activities and by leaching of the soluble compounds and indirectly by changing litter chemistry and plant traits (Fragoso and Lavelle 1992; Cornejo et al. 1994; Austin and Vitousek 2000).

Although several studies on the influence of precipitation regime on ecosystem C cycling in tropical forests are available (e.g. Schuur 2003; Powers et al. 2009; Posada and Schuur 2011; Malhi et al. 2015), a full understanding of the impact remains incomplete (Powers et al. 2011; Marín-Spiotta and Sharma 2013). In TDFs most ecosystem processes are controlled by water availability (Campo et al. 1998, 2001), for example, slow decomposition of organic matter occurs in drought period causing high carbon storage in the forest and deposition of free light fraction of organic matter (Cuevas et al. 2013; Roa-Fuentes et al. 2013).

In a study by Pandey et al. (2014), mean weight loss of litter through decomposition showed positive relationship with soil surface temperature and soil moisture content. This study showed higher weight loss during rainy season as warm and humid conditions in rainy season favour decomposition and lower weight loss during winter season as cold and dry conditions negatively affect litter weight loss in tropical forest (Arunachalam et al. 1998; Maithani et al. 1996).

N and P mineralization in tropical soil is closely related to the rainfall pattern (Allen et al. 2017). The study by Jha et al. (1996a, b) in a tropical dry forest showed that the rate of N mineralization increased in the rainy season as soil moisture content increased. Jha et al. (1996a) studied the N mineralization rates and the size of the viable community of nitrifying bacteria in a dry forest site and an adjoining cropland site and found a strong positive relationship between N mineralization and soil moisture, which indicated that the mineralization process was moisture-limited. In the study of Jha et al. (1996a), the most probable number (MPN) counts of ammonia- and nitrite-oxidizing bacteria were largest in rainy season and smallest in the dry season and were significantly and positively related to soil moisture. The size of the viable community of nitrifiers was positively related with nitrification as well as N mineralization rates. The activity of nitrifiers is reported to be inhibited more than that of ammonifiers at soil moisture potentials below −1.5 MPa (Dommergues 1966). While substrate limitation was the major inhibiting factor for the activity of nitrifying bacteria at soil water potentials greater than −0.6 MPa, the adverse physiological effects associated with cell dehydration were more inhibiting at water potentials of less than −0.6 MPa (Stark and Firestone 1995). Thus soil moisture affects the activity of nitrifying bacteria through both dehydration and substrate limitation. The sensitivity of nitrifiers to soil moisture is related to their high energy requirement, diverting energy sources which might otherwise be used to synthesize compatible solutes such as amine or polyols, which could help the organisms to withstand dry conditions (Sprent 1987).

2.3.3 Soil CO_2 Flux

Soil respiration or soil CO_2 flux is the production of carbon dioxide by the respiration of soil microorganisms and plant roots. Soil CO_2 flux is the largest carbon flux to the atmosphere by the earth (Vicca et al. 2014). Soil respiration includes autotrophic respiration (respiration by plant root) as well as heterotrophic respiration (respiration by microbes and soil fauna); hence it indicates the belowground productivity and metabolic activity (Cleveland et al. 2010). Soil CO_2 flux is regulated by various factors including root biomass and its activity. Soil CO_2flux increases with an increase in root biomass (Schlesinger and Andrews 2000). Root biomass and its activity contribute 20–90% to the total CO_2 flux globally from the soil (Boone et al. 1998). Apart from root biomass, soil CO_2 flux is also strongly regulated by substrate availability to the microbes, soil temperature and soil moisture (Linn and Doran 1984; Craine et al. 1999; Zak et al. 1999).

Altered precipitation pattern would lead to variation in soil moisture availability and hence will affect both plant and microbial processes and alter the soil CO_2 flux (Harper et al. 2005). General predictions indicate that increased water stress would decrease soil CO_2 flux (Harper et al. 2005). Studies conducted across the globe indicated that soil CO_2 flux is more susceptible in the twenty-first century, due to a decrease in mean annual precipitation in tropics in the near future (Neelin et al. 2006; Flato et al. 2013). The change in climatic conditions alters hydrological cycle and apparently results in an enormous and nonlinear effect on the soil CO_2 flux (Vicca et al. 2014).

As indicated earlier, soil organic matter decomposition rate also varies with the change in temperature and precipitation (Zhang et al. 2015). Thus soil CO_2 flux would change as substrate availability fluctuates with these changes. A hump-shaped curve describes the relationship between soil CO_2 flux and soil moisture as well as between soil organic carbon and soil moisture (Orchard and Cook 1983; Wood and Silver 2012; Vicca et al. 2014), indicating that carbon mineralization decreases in both dry and inundated condition due to lack of water in the former and lack of oxygen in the latter (Allen et al. 2017). Studies have indicated that the lack of rain in the dry period and drought conditions decrease soil CO_2 flux in dry tropical forests (Adachi et al. 2009; Wood and Silver 2012). In a rainfall manipulation meta-analysis experiment, Wu et al. (2011) proved that decreasing precipitation reduced soil CO_2 flux by 12% and increased precipitation increased the soil CO_2 flux by 30%.

Waring and Powers (2016) observed that soil CO_2 fluxes are linked to seasonal and short-term variation in water availability, as also indicated by other studies (Birch 1958; Cook and Orchard 2008; Moyano et al. 2012). In the field manipulation, rewetting of dry soils produced an immediate pulse of CO_2, accompanied by rapid immobilization of N into the microbial biomass. Strong CO_2 pulses were also observed in previously moistened soil, suggesting that observed respiratory responses were biological in origin and did not simply represent flushing of pent-up CO_2 out of soil pore space. Instead, CO_2 pulses likely represent rapid microbial metabolism of organic substrate that was previously unavailable due to occlusion in soil aggregates or limitations on diffusion (Manzoni et al. 2014).

2.3.4 Microbial Biomass

Microbial biomass measures the size, diversity and activity of microbial population in soil and is related to soil organic matter decomposition and nutrient mineralization. It is a biological soil property and is an important indicator of soil fertility (Dominati et al. 2010). In the tropical soil, microbial population constitutes a considerable number (~10^{11}–10^{12} per g soil) (Dubinsky et al. 2010), preserving thus a huge reservoir of functional and physiological plasticity that allows them to adapt to different environmental conditions (Bouskill et al. 2013).

Soil moisture and organic carbon are the two major governing factors for the microbial biomass. Low precipitation or drought generates a significant metabolic response to microbes (Bouskill et al. 2016), hence affecting the activities and composition of the microbial communities. In a study by Bouskill et al. (2013), microbial community structure got reduced under drought condition, showing the change in precipitation regime.

2.4 Effect of Temperature and Precipitation on Soil Properties

2.4.1 Soil Carbon

Carbon is one of the vital non-mineral nutrients utilised by the ecosystem. Soils are one of the largest reservoirs of carbon and hence play a major role in governing the climate regime (Post et al. 1982; Swift 2001) by acting as a source or sink of CO_2, one of the major greenhouse gases. Soil carbon pool may act as a source or sink of atmospheric carbon depending upon the land use and climatic condition (Houghton et al. 1987; Don et al. 2011; Zatta et al. 2014).

We have discussed above that precipitation has a greater influence on ecosystem processes in the tropics than in temperate regions. Precipitation affects soil organic carbon in several ways, for instance by altering available soil nutrient pool, microbial activity and fine root and litter chemistry (Schimel et al. 1994; Hutsch et al. 2002; Sardans et al. 2008; Posada and Schuur 2011). Zhang et al. (2015) showed that soil organic matter decomposition rate increases significantly with increasing mean annual temperature and precipitation. In a review, Wei et al. (2014a, b) found that the increase in mean annual precipitation increases the soil organic carbon turnover rate globally. In the tropics where there is a seasonal climate, soil carbon pool including the microbial biomass carbon was higher in the drier season than in wetter season (Singh et al. 1989). Campo and Merino (2016) found negative relationship between mean annual precipitation and soil carbon sequestration in the tropical dry forest in Mexico.

2.4.2 Soil Nitrogen

The tropical soil is nutrient-limited, and among all the essential nutrients, nitrogen is the most crucial. The rate at which nitrogen is made available to the plant defines the functioning of the ecosystem. Soil nitrogen dynamics is governed by the immobilization and release of nitrogen (Roy and Singh 1994, 1995). Precipitation affects nutrient availability through soil wet-dry cycle (Austin et al. 2004). Soil carbon, nitrogen concentration and organic matter decrease with the increasing rainfall (Cuevas et al. 2013). Campo and Merino (2016) found increase in the concentration of soil inorganic nitrogen in the driest site on the precipitation gradient. Singh et al. (1989) also reported nitrogen accumulation in the dry season. High concentration of nutrients on the driest site or dry season may be due to reduced leaching and release of nutrients immobilized in soil (Singh et al. 1989; Campo et al. 1998).

Nitrogen losses occur due to nitrate leaching and denitrification, particularly in the form of gases, and are strongly related to the soil moisture availability (Austin et al. 2004). But the extent of the effect of precipitation on organic matter (nitrogen) and inorganic N pool relies heavily on the soil organisms and resource availability (Austin et al. 2004). Overall the biological processes appear to be more vigorous in moist soil resulting in a rapid cycling of the organic matter (Cuevas et al. 2013).

2.5 Conclusion

Changes in the climate over the tropical countries especially change in temperature and precipitation regimes, as suggested by several models, would have long-term consequences on the ecological functioning of the ecosystems. This review highlights the possible change in soil properties and processes in the nutrient-limited tropical soils due to changing climatic conditions. Decrease in rainfall would increase the deciduousness of the tropical vegetation that could lead to significant changes in the nutrient cycling which is the backbone of soil fertility and productivity of an ecosystem. Decrease in mean annual precipitation will possibly increase the nutrient use efficiency of the plants; decrease soil carbon, nitrogen and phosphorus concentration; decrease decomposition and mineralization rate; decrease the fluxes of CO_2, N and P; and decrease microbial population and its activity though microbial biomass could increase due to immobilization of nutrients and carbon. The above signifies the sensitivity of tropical ecosystems towards changing precipitation regime. To draw a final conclusion on the effects the changing climatic conditions on the tropical ecosystem, more studies are needed.

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References

Adachi M, Ishida A, Bunyavejchewin S, Okuda T, Koizumi H (2009) Spatial and temporal variation in soil respiration in a seasonally dry tropical forest, Thailand. J Trop Ecol 25(5):531–539

Aerts R (1995) The advantages of being evergreen. Trends Ecol Evol 10(10):402–407

Allen K, Dupuy JM, Gei MG, Hulshof C, Medvigy D, Pizano C, Salgado-Negret B, Smith CM, Trierweiler A, Van Bloem SJ, Waring BG (2017) Will seasonally dry tropical forests be sensitive or resistant to future changes in rainfall regimes? Environ Res Lett 12(2):023001

Anaya C, Jaramillo V, Martinez-Yrizar A, Garcia-Oliva F (2012) Large rainfall pulses control litter decomposition in a tropical dry forest: evidence from an 8-year study. Ecosystems 15:652–663

Arunachalam A, Maithani K, Pandey HN, Tripathi RS (1998) Leaf litter decomposition and nutrient mineralization patterns in regrowing stands of a humid subtropical forest after tree cutting. For Ecol Manag 109(1):151–161

Austin AT, Vitousek PM (2000) Precipitation, decomposition and litter decomposability of Metrosiderospolymorpha in native forests on Hawaii. J Ecol 88(1):129–138

Austin AT, Yahdjian L, Stark JM, Belnap J, Porporato A, Norton U, Ravetta DA, Schaeffer SM (2004) Water pulses and biogeochemical cycles in arid and semiarid ecosystems. Oecologia 141(2):221–235

Bejarano M, Etchevers JD, Ruíz-Suárez G, Campo J (2014) The effects of increased N input on soil C and N dynamics in seasonally dry tropical forests: an experimental approach. Appl Soil Ecol 73:105–115

Berg B, Berg MP, Bottner P, Box E, Breymeyer A, De Anta RC, Couteaux M, Escudero A, Gallardo A, Kratz W, Madeira M (1993) Litter mass loss rates in pine forests of Europe and Eastern United States: some relationships with climate and litter quality. Biogeochemistry 20(3):127–159

Birch HF (1958) The effect of soil drying on humus decomposition and nitrogen availability. Plant Soil 10(1):9–31

Bohlman SA (2010) Landscape patterns and environmental controls of deciduousness in forests of central Panama. Glob Ecol Biogeogr 19(3):376–385

Boone RD, Nadelhoffer KJ, Canary JD, Kaye JP (1998) Roots exert a strong influence on the temperature sensitivity of soil respiration. Nature 396(6711):570–572

Borchert R (1998) Responses of tropical trees to rainfall seasonality and its long-term changes. Clim Chang 39(2):381–393

Bouskill NJ, Lim HC, Borglin S, Salve R, Wood TE, Silver WL, Brodie EL (2013) Pre-exposure to drought increases the resistance of tropical forest soil bacterial communities to extended drought. ISME J 7(2):384–394

Bouskill NJ, Wood TE, Baran R, Hao Z, Ye Z, Bowen BP, Lim HC, Nico PS, Holman HY, Gilbert B, Silver WL (2016) Belowground response to drought in a tropical forest soil. II. Change in microbial function impacts carbon composition. Front Microbiol 7(525):1–11

Brady NC, Weil RR (2008) The nature and properties of soils, 14th edn. Pearson Prentice Hall, Upper Saddle River

Brevik EC (2013) April. Climate change, soils, and human health. In: EGU General Assembly Conference Abstracts, vol 15, pp 7–12

Brussaard L (1994) Interrelationships between biological activities, soil properties and soil management. In: Greenland DJ, Szabolcs I (eds) Soil resilience and sustainable land use. CAB International, Wallingford, pp 309–329

Campo J (2016) Shift from ecosystem P to N limitation at precipitation gradient in tropical dry forests at Yucatan, Mexico. Environ Res Lett 11(9):095006

Campo J, Merino A (2016) Variations in soil carbon sequestration and their determinants along a precipitation gradient in seasonally dry tropical forest ecosystems. Glob Chang Biol 22(5):1942–1956

Campo J, Vazquez-Yanes C (2004) Effects of nutrient limitation on aboveground carbon dynamics during tropical dry forest regeneration in Yucatan, Mexico. Ecosystems 7(3):311–319

Campo J, Jaramillo VJ, Maass JM (1998) Pulses of soil phosphorus availability in a Mexican tropical dry forest: effects of seasonality and level of wetting. Oecologia 115(1):167–172

Campo J, Maass M, Jaramillo VJ, Martínez-Yrízar A, Sarukhan J (2001) Phosphorus cycling in a Mexican tropical dry forest ecosystem. Biogeochemistry 53(2):161–179

Chadwick R, Good P, Martin G, Rowell DP (2016) Large rainfall changes consistently projected over substantial areas of tropical land. Nat Clim Chang 6(2):177

Change IPCC (2007) Climate change 2007: The physical science basis. Agenda 6(07):333

Cleveland CC, Wieder WR, Reed SC, Townsend AR (2010) Experimental drought in a tropical rain forest increases soil carbon dioxide losses to the atmosphere. Ecology 91(8):2313–2323

Condit R, Watts K, Bohlman SA, Perez R, Foster RB, Hubbell SP (2000) Quantifying the deciduousness of tropical forest canopies under varying climates. J Veg Sci 11(5):649–658

Cook FJ, Orchard VA (2008) Relationships between soil respiration and soil moisture. Soil Biol Biochem 40(5):1013–1018

Cornejo FH, Varela A, Wright SJ (1994) Tropical forest litter decomposition under seasonal drought: nutrient release, fungi and bacteria. Oikos 70(2):183–190

Craine JM, Wedin DA, Chapin FS (1999) Predominance of ecophysiological controls on soil CO_2 flux in a Minnesota grassland. Plant Soil 207(1):77–86

Cuevas RM, Hidalgo C, Payan F, Etchevers JD, Campo J (2013) Precipitation influences on active fractions of soil organic matter in seasonally dry tropical forests of the Yucatan: regional and seasonal patterns. Eur J For Res 132:667–677

Dirzo R, Young HS, Mooney HA (eds) (2011) Seasonally dry tropical forests: ecology and conservation. Island Press, Washington, DC

Dominati E, Patterson M, Mackay A (2010) A framework for classifying and quantifying the natural capital and ecosystem services of soils. Ecol Econ 69(9):1858–1868

Dommergues YR (1966) Biologie du sol. Presses Universitaire de France, Paris

Don A, Schumacher J, Freibauer A (2011) Impact of tropical land-use change on soil organic carbon stocks–a meta-analysis. Glob Chang Biol 17(4):1658–1670

Drenovsky RE, Richards JH (2006) Low leaf N and P resorption contributes to nutrient limitation in two desert shrubs. Plant Ecol 183(2):305–314

Dubinsky EA, Silver WL, Firestone MK (2010) Tropical forest soil microbial communities couple iron and carbon biogeochemistry. Ecology 91(9):2604–2612

Duffy PB, Brando P, Asner GP, Field CB (2015) Projections of future meteorological drought and wet periods in the Amazon. Proc Natl Acad Sci 112(43):13172–13177

Easterling DR, Meehl GA, Parmesan C, Changnon SA, Karl TR, Mearns LO (2000) Climate extremes: observations, modeling, and impacts. Science 289(5487):2068–2074

Feng X, Porporato A, Rodriguez-Iturbe I (2013) Changes in rainfall seasonality in the tropics. Nat Clim Chang 3(9):811–815

Flato G, Marotzke J, Abiodun B, Braconnot P, Chou SC, Collins WJ, Cox P, Driouech F, Emori S, Eyring V, Forest C (2013) Evaluation of climate models. In: Climate change. The physical science basis. Contribution of working group I to the fifth assessment report of the intergovernmental panel on climate change, vol 5. IPCC, Cambridge, pp 741–866

Fragoso C, Lavelle P (1992) Earthworm communities of tropical rain forests. Soil Biol Biochem 24(12):1397–1408

Gamboa AM, Hidalgo C, De Leon F, Etchevers JD, Gallardo JF, Campo J (2010) Nutrient addition differentially affects soil carbon sequestration in secondary tropical dry forests: early-versus late-succession stages. Restor Ecol 18(2):252–260

Gei MG, Powers JS (2014) Nutrient cycling in tropical dry forests. In: Tropical dry forests in the Americas: ecology, conservation, and management. CRC Press, Boca Raton, pp 141–155

Gholz HL, Wedin DA, Smitherman SM, Harmon ME, Parton WJ (2000) Long-term dynamics of pine and hardwood litter in contrasting environments: toward a global model of decomposition. Glob Chang Biol 6(7):751–765

Gitay H, Suarez A, Watson RT, Dokken DJ (2002) Climate change and biodiversity. In: IPCC (Inter- governmental Panel on Climate Change) technical paper V, 86. http://www.ipcc.ch

Givnish TJ (2002) Adaptive significance of evergreen vs. deciduous leaves: solving the triple paradox. Silva Fennica 36(3):703–743

Gond V, Fayolle A, Pennec A, Cornu G, Mayaux P, Camberlin P, Doumenge C, Fauvet N, Gourlet-Fleury S (2013) Vegetation structure and greenness in Central Africa from Modis multi-temporal data. Philos Trans R Soc B 368(1625):2012–0309

Greve P, Orlowsky B, Mueller B, Sheffield J, Reichstein M, Seneviratne SI (2014) Global assessment of trends in wetting and drying over land. Nat Geosci 7(10):716–721

Harper CW, Blair JM, Fay PA, Knapp AK, Carlisle JD (2005) Increased rainfall variability and reduced rainfall amount decreases soil CO2 flux in a grassland ecosystem. Glob Chang Biol 11(2):322–334

Hartemink AE (2002) Soil science in tropical and temperate regions some differences and similarities. Adv Agron 77:269–292

Heisler-White JL, Knapp AK, Kelly EF (2008) Increasing precipitation event size increases aboveground net primary productivity in a semi-arid grassland. Oecologia 158(1):129–140

Heisler-White JL, Blair JM, Kelly EF, Harmoney K, Knapp AK (2009) Contingent productivity responses to more extreme rainfall regimes across a grassland biome. Glob Chang Biol 15(12):2894–2904

Houghton RA, Boone RD, Fruci JR, Hobbie JE, Melillo JM, Palm CA, Peterson BJ, Shaver GR, Woodwell GM, Moore B, Skole DL (1987) The flux of carbon from terrestrial ecosystems to the atmosphere in 1980 due to changes in land use: geographic distribution of the global flux. Tellus Ser B Chem Phys Meteorol 39(1–2):122–139

Hütsch BW, Augustin J, Merbach W (2002) Plant rhizodeposition—an important source for carbon turnover in soils. J Plant Nutr Soil Sci 165(4):397–407

Jentsch A, Beierkuhnlein C (2008) Research frontiers in climate change: effects of extreme meteorological events on ecosystems. Compt Rendus Geosci 340(9):621–628

Jha PB, Kashyap AK, Singh JS (1996a) Effect of fertilizer and organic matter inputs on nitrifier populations and N-mineralization rates in a dry tropical region, India. Appl Soil Ecol 4(3):231–241

Jha PB, Singh JS, Kashyap AK (1996b) Dynamics of viable nitrifier community and nutrient availability in dry tropical forest habitat as affected by cultivation and soil texture. Plant Soil 180(2):277–285

Killingbeck KT (1996) Nutrients in senesced leaves: keys to the search for potential resorption and resorption proficiency. Ecology 77(6):1716–1727

Knapp AK, Beier C, Briske DD, Classen AT, Luo Y, Reichstein M, Smith MD, Smith SD, Bell JE, Fay PA, Heisler JL (2008) Consequences of more extreme precipitation regimes for terrestrial ecosystems. AIBS Bull 58(9):811–821

Kobe RK, Lepczyk CA, Iyer M (2005) Resorption efficiency decreases with increasing green leaf nutrients in a global data set. Ecology 86(10):2780–2792

Lal CB, Annapurna C, Raghubanshi AS, Singh JS (2001) Foliar demand and resource economy of nutrients in dry tropical forest species. J Veg Sci 12(1):5–14

Linn DM, Doran JW (1984) Effect of water-filled pore space on carbon dioxide and nitrous oxide production in tilled and nontilled soils. Soil Sci Soc Am J 48(6):1267–1272

Maithani K, Tripathi RS, Arunachalam A, Pandey HN (1996) Seasonal dynamics of microbial biomass C, N and P during regrowth of a disturbed subtropical humid forest in north-east India. Appl Soil Ecol 4(1):31–37

Malhi Y, Roberts JT, Betts RA, Killeen TJ, Li W, Nobre CA (2008) Climate change, deforestation, and the fate of the Amazon. Science 319(5860):169–172

Malhi Y, Doughty CE, Goldsmith GR, Metcalfe DB, Girardin CA, Marthews TR, Aguila-Pasquel J, Aragao LE, Araujo-Murakami A, Brando P, Costa AC (2015) The linkages between photosynthesis, productivity, growth and biomass in lowland Amazonian forests. Glob Chang Biol 21(6):2283–2295

Maloney ED, Camargo SJ, Chang E, Colle B, Fu R, Geil KL, Hu Q, Jiang X, Johnson N, Karnauskas KB, Kinter J (2014) North American climate in CMIP5 experiments: part III: assessment of twenty-first-century projections. J Clim 27(6):2230–2270

Manzoni S, Schaeffer SM, Katul G, Porporato A, Schimel JP (2014) A theoretical analysis of microbial eco-physiological and diffusion limitations to carbon cycling in drying soils. Soil Biol Biochem 73:69–83

Marin-Spiotta E, Sharma S (2013) Carbon storage in successional and plantation forest soils: a tropical analysis. Glob Ecol Biogeogr 22(1):105–117

Meir P, Pennington RT (2011) Climatic change and seasonally dry tropical forests. In: Seasonally dry tropical forests. Center for Resource Economics, Island Press, Washington, DC, pp 279–299

Moyano FE, Vasilyeva NA, Bouckaert L, Cook F, Craine JM, Don A, Epron D, Formanek P, Franzluebbers A, Ilstedt U, Katterer T (2012) The moisture response of soil heterotrophic respiration: interaction with soil properties. Biogeosciences 9(3):1173–1182

Neelin JD, Munnich M, Su H, Meyerson JE, Holloway CE (2006) Tropical drying trends in global warming models and observations. Proc Natl Acad Sci 103(16):6110–6115

Orchard VA, Cook FJ (1983) Relationship between soil respiration and soil moisture. Soil Biol Biochem 15(4):447–453

Pachauri RK, Allen MR, Barros VR, Broome J, Cramer W, Christ R, Church JA, Clarke L, Dahe Q, Dasgupta P, Dubash NK (2014) Climate change 2014: synthesis report. Contribution of working groups I, II and III to the fifth assessment report of the intergovernmental panel on climate change. IPCC, p 151

Pandey SK, Singh H, Singh JS (2014) Effect of environmental conditions on decomposition in eight woody species of a dry tropical forest. Plant Biosyst Int J Deal Asp Plant Biol 148(3):410–418

Posada JM, Schuur EA (2011) Relationships among precipitation regime, nutrient availability, and carbon turnover in tropical rain forests. Oecologia 165(3):783–795

Post WM, Emanuel WR, Zinke PJ, Stangenberger AG (1982) Soil carbon pools and world life zones. Nature 298(5870):156–159

Powers JS, Montgomery RA, Adair EC, Brearley FQ, DeWalt SJ, Castanho CT, Chave J, Deinert E, Ganzhorn JU, Gilbert ME, Gonzalez-Iturbe JA (2009) Decomposition in tropical forests: a pan-tropical study of the effects of litter type, litter placement and mesofaunal exclusion across a precipitation gradient. J Ecol 97(4):801–811

Powers JS, Corre MD, Twine TE, Veldkamp E (2011) Geographic bias of field observations of soil carbon stocks with tropical land-use changes precludes spatial extrapolation. Proc Natl Acad Sci 108(15):6318–6322

Ratnam J, Sankaran M, Hanan NP, Grant RC, Zambatis N (2008) Nutrient resorption patterns of plant functional groups in a tropical savanna: variation and functional significance. Oecologia 157(1):141–151

Reich PB (1995) Phenology of tropical forests: patterns, causes, and consequences. Can J Bot 73(2):164–174

Renteria LY, Jaramillo VJ (2011) Rainfall drives leaf traits and leaf nutrient resorption in a tropical dry forest in Mexico. Oecologia 165(1):201–211

Renteria LY, Jaramillo VJ, Martínez-Yrizar A, Perez-Jimenez A (2005) Nitrogen and phosphorus resorption in trees of a Mexican tropical dry forest. Trees 19(4):431–441

Roa-Fuentes LL, Hidalgo C, Etchevers JD, Campo J (2013) The effects of precipitation regime on soil carbon pools on the Yucatan Peninsula. J Trop Ecol 29(5):463–466

Roy S, Singh JS (1994) Consequences of habitat heterogeneity for availability of nutrients in a dry tropical forest. J Ecol 82:503–509

Roy S, Singh JS (1995) Seasonal and spatial dynamics of plant-available N and P pools and N-mineralization in relation to fine roots in a dry tropical forest habitat. Soil Biol Biochem 27(1):33–40

Ryan DF, Bormann FH (1982) Nutrient resorption in northern hardwood forests. Bioscience 32(1):29–32

Sala OE, Gherardi LA, Peters DP (2015) Enhanced precipitation variability effects on water losses and ecosystem functioning: differential response of arid and Mesic regions. Clim Chang 131(2):213–227

Salamanca EF, Kaneko N, Katagiri S (2003) Rainfall manipulation effects on litter decomposition and the microbial biomass of the forest floor. Appl Soil Ecol 22(3):271–281

Sardans J, Penuuelas J, Estiarte M, Prieto P (2008) Warming and drought alter C and N concentration, allocation and accumulation in a Mediterranean shrubland. Glob Chang Biol 14(10):2304–2316

Schimel DS, Braswell BH, Holland EA, McKeown R, Ojima DS, Painter TH, Parton WJ, Townsend AR (1994) Climatic, edaphic, and biotic controls over storage and turnover of carbon in soils. Glob Biogeochem Cycles 8(3):279–293

Schlesinger WH, Andrews JA (2000) Soil respiration and the global carbon cycle. Biogeochemistry 48(1):7–20

Schuur EA (2001) The effect of water on decomposition dynamics in mesic to wet Hawaiian montane forests. Ecosystems 4(3):259–273

Schuur EA (2003) Productivity and global climate revisited: the sensitivity of tropical forest growth to precipitation. Ecology 84(5):1165–1170

Singh L, Singh JS (1991a) Species structure, dry matter dynamics and carbon flux of a dry tropical forest in India. Ann Bot 68(3):263–273

Singh L, Singh JS (1991b) Storage and flux of nutrients in a dry tropical forest in India. Ann Bot 68(3):275–284

Singh L, Singh JS (1993) Importance of short-lived components of a dry tropical forest for biomass production and nutrient cycling. J Veg Sci 4(5):681–686

Singh JS, Raghubanshi AS, Singh RS, Srivastava SC (1989) Microbial biomass acts as a source of plant nutrients in dry tropical forest and savanna. Nature 338(6215):499–500

Singh D, Tsiang M, Rajaratnam B, Diffenbaugh NS (2013) Precipitation extremes over the continental United States in a transient, high-resolution, ensemble climate model experiment. J Geophys Res-Atmos 118(13):7063–7086

Smith MD (2011) An ecological perspective on extreme climatic events: a synthetic definition and framework to guide future research. J Ecol 99(3):656–663

Sprent JI (1987) The ecology of the nitrogen cycle. Cambridge University Press, Cambridge

Stark JM, Firestone MK (1995) Mechanisms for soil moisture effects on activity of nitrifying bacteria. Appl Environ Microbiol 61(1):218–221

Swift RS (2001) Sequestration of carbon by soil. Soil Sci 166(11):858–871

Thomey ML, Collins SL, Vargas R, Johnson JE, Brown RF, Natvig DO, Friggens MT (2011) Effect of precipitation variability on net primary production and soil respiration in a Chihuahuan Desert grassland. Glob Chang Biol 17(4):1505–1515

Trofymow JA, Moore TR, Titus B, Prescott C, Morrison I, Siltanen M, Smith S, Fyles J, Wein R, Camire C, Duschene L (2002) Rates of litter decomposition over 6 years in Canadian forests: influence of litter quality and climate. Can J For Res 32(5):789–804

Trumbore SE (1993) Comparison of carbon dynamics in tropical and temperate soils using radiocarbon measurements. Glob Biogeochem Cycles 7(2):275–290

Verduzco VS, Garatuza-Payan J, Yepez EA, Watts CJ, Rodriguez JC, Robles-Morua A, Vivoni ER (2015) Variations of net ecosystem production due to seasonal precipitation differences in a tropical dry forest of northwest Mexico. J Geophys Res Biogeosci 120(10):2081–2094

Vicca S, Bahn M, Estiarte M, Van Loon EE, Vargas R, Alberti G, Ambus P, Arain MA, Beier C, Bentley LP, Borken W (2014) Can current moisture responses predict soil CO2 efflux under altered precipitation regimes? a synthesis of manipulation experiments. Biogeosciences 11(11):2991–3013

Waring BG, Powers JS (2016) Unraveling the mechanisms underlying pulse dynamics of soil respiration in tropical dry forests. Environ Res Lett 11(10):105005

Wei X, Huang L, Xiang Y, Shao M, Zhang X, Gale W (2014a) The dynamics of soil OC and N after conversion of forest to cropland. Agric For Meteorol 194:188–196

Wei X, Shao M, Gale W, Li L (2014b) Global pattern of soil carbon losses due to the conversion of forests to agricultural land. Sci Rep 4:4062

Wood TE, Silver WL (2012) Strong spatial variability in trace gasdynamics following experimental drought in a humid tropical forest. Glob Biogeochem Cycles 26(3):GB3005

Wright IJ, Westoby M (2003) Nutrient concentration, resorption and lifespan: leaf traits of Australian sclerophyll species. Funct Ecol 17(1):10–19

Wu Z, Dijkstra P, Koch GW, Penuelas J, Hungate BA (2011) Responses of terrestrial ecosystems to temperature and precipitation change: a meta-analysis of experimental manipulation. Glob Chang Biol 17(2):927–942

Yuan ZY, Chen HY (2009a) Global-scale patterns of nutrient resorption associated with latitude, temperature and precipitation. Glob Ecol Biogeogr 18(1):11–18

Yuan Z, Chen HY (2009b) Global trends in senesced-leaf nitrogen and phosphorus. Glob Ecol Biogeogr 18(5):532–542

Zak DR, Holmes WE, MacDonald NW, Pregitzer KS (1999) Soil temperature, matric potential, and the kinetics of microbial respiration and nitrogen mineralization. Soil Sci Soc Am J 63(3):575–584

Zatta A, Clifton-Brown J, Robson P, Hastings A, Monti A (2014) Land use change from C3 grassland to C4 Miscanthus: effects on soil carbon content and estimated mitigation benefit after six years. GCB Bioenergy 6(4):360–370

Zhang K, Dang H, Zhang Q, Cheng X (2015) Soil carbon dynamics following land-use change varied with temperature and precipitation gradients: evidence from stable isotopes. Glob Chang Biol 21(7):2762–2772

Nature and Extent of Forest Degradation in Central Himalayas

3

Surendra P. Singh, Anvita Pandey, and Vishal Singh

Abstract

Forest degradation is widespread in developing countries, as poor people depend on forest biomass collection on a "day-to-day" basis, giving little respite to forest ecosystems to recover. In the Himalayas, collection of firewood, tree leaf fodder, and leaf litter from forest floor is one of the main causes of forest degradation, particularly in oak (*Quercus leucotrichophora*) and pine (*Pinus roxburghii*) elevation belt (1000–2200 m elevation) of the Western and Central Himalayas. In this area, whole tree cutting is uncommon, but most trees are lopped until they become severely denuded, with little crown left. Compared to healthy forests, generally such degraded forests have 40–50% less biomass but about 80% less net primary productivity and 86.4% less leaf area. As degradation progresses, the biomass extracted from the forest declines, but in proportional terms it increases, resulting in a rapid disintegration of ecosystem structure and functions. Some of the observed effects are (1) reduction of soil carbon and nitrogen by 40–50%; (2) desiccation of oak acorns lying on forest floor well before the arrival of monsoon; (3) about 40% reduction in leaf litter decomposition; (4) 35% reduction in ectomycorrhizal association with roots, as indicated by fungal sporocarp density; and (5) 25% reduction in soil water holding capacity and 25% increase in soil bulk density (these values are in comparison to a relatively undisturbed forest). Recovery of a degraded forest (chronic disturbance) has not yet been investigated

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but is likely to be slower than that of a clear-cut (acute disturbance) forest. However, some interventions can result in a faster recovery.

Keywords
Forest degradation · Forest biomass · Leaf litter and net primary productivity · Himalayas · Oak and pine · Recovery

3.1 Introduction

Land-use change activities contribute to 1.6 billion tons of carbon released annually into the atmosphere, of which a major part results from deforestation and forest degradation (Denman 2007). In 1990 the area of world forest was 4128 million ha, but it declined to 3999 million ha in 2015, and as per an assessment conducted by FAO 2015 for a period between 2010 and 2015, the natural forest area (which accounts for 93% of the forest area) has been declining at the rate of 6.5 million ha per year. While forest area continued to decline, the human populations increased at a rapid pace, resulting in a decreased per capita forest area. From the climate change angle, the carbon stocks in the global area forest have declined by 11 gigatons (Gt) in the last 25 years (FAO 2015). As a result of international negotiations being held each year in the Conference of the Parties (CoP), countries are encouraged to provide reliable estimates of deforestation and forest degradation. In addition to this, countries have their own targets to check emissions through forest conversion and improve the forest cover. In spite of the relevance of quantitative national-level data, for most countries they are unreliable, and the drivers of deforestation and forest degradation are not well understood (Hosonuma et al. 2012).

Deforestation occurs as discrete patches of forest are removed at particular points in time and space. It is widely believed that the extent of deforestation and forest degradation in developing countries is large and growing over time. Studies conducted in the mid-Himalayan region contend that forest degradation, not deforestation, is the key problem (Baland and Mookherjee 2014; Singh et al. 2014). Chronic disturbance may not lead to deforestation depending upon the practices of the local communities, for example, local people may apply rotational use of a degraded forest to conserve it. The nature of degradation in the Himalayas is different when compared to degradation in other regions of the world (Singh 1998). Starting from the theory of Himalayan degradation in the 1980s, which predicted the complete loss of Himalayan forests and devastations in downstream areas, adverse environmental repercussions of forest degradation have been a major issue in scientific research and public debate (Ives 1989; Singh et al. 1984, 2014; Singh and Singh 1987; Singh 1998; Thadani 1999; Singh and Thadani 2015).

While deforestation generally is associated with commerce, forest degradation has more of a poverty angle; it is rare in rich countries (Skutsch et al. 2008). Poor local people living in and around forests depend on forest biomass to sustain their subsistence living. In the Indian Himalayan state of Uttarakhand, each energy unit produced agronomically entails the consumption of ten energy units from the forest

(Singh and Singh 1992). Degradation is less caused by logging than by extraction of various other forest products (firewood, charcoal, fodder, non-timber forest products like seeds, resin, and roots) or by patchy clearance and regrowth associated with shifting agriculture by indigenous communities (Tewari et al. 2008). Forest degradation leads to reduced carbon levels for many decades and cannot be easily detected by satellite images (DeFries et al. 2007).

In order to meet the "day-to-day" demands of fuel wood, and fodder, people in the Central and Western Himalayan regions remove small and invisible fractions of biomass at a given time, generally as headloads of fire wood or of fodder and leaf litter. Grazing is yet another common practice followed in this region and each household has a number of livestock. This form of disturbance is chronic in nature (Singh 1998), unlike deforestation which comprises acute disturbance. The problem with the chronic form of disturbance is that ecosystems often do not get adequate time to recover because human presence and interference never stop. This can cause adverse changes in the forest even if rates of biomass removal are within the carrying capacity of the forest net primary production (Singh 1998). For example, browsing of tree seedlings results in little removal of biomass, but it may choke the regeneration eventually leading to loss of the forest. Generally, whole tree cutting is uncommon; instead, people cut branches and twigs of trees. This form of disturbance occurs in much of the Himalayan east-to-west arc, and includes Indian states of Kashmir, Himachal Pradesh, Uttarakhand, and much of Nepal.

This study focused on the elevation zone of pure and mixed stands of chir pine (*Pinus roxburghii*) and banj oak (*Quercus leucotrichophora*), where the majority of the mountain people live, i.e., between 1000 and 2200 m elevation. This form of chronic disturbance, in which whole tree cutting is avoided but individual trees are severely denuded and tree crown is drastically reduced, is not well-recognized globally. The scale of forest degradation in the Himalayas is possibly decreasing, but trends are still uncertain. In this article, we (1) characterize the nature of the forest degradation in the Himalayas, (2) analyze the impact of chronic disturbance at stand and individual tree level, (3) and discuss some of the possible interventions to address it. We consider forest degradation at regional, stand, and individual tree levels. Generally, a degradation study focuses on the loss of crown cover and stocking density, but here we have also focused on analyzing changes which affect processes, like litter decomposition and regeneration, and structural features like leaf area and soil organic matter. We also shed light on recovery of a degraded oak forest, which is likely to be different from that of a clear-cut forest (acute disturbance).

3.2 Extent of Forest Degradation in the Himalayas

As per the assessment of the India State of Forest Report 2015, the forest cover of the country is 70.17 million ha which constitutes 21.34% of the geographical area of the country. In terms of density classes, area covered by very dense forest (cover density of 70% and more) is 12.2% and that by moderately dense forest (cover density of 40% and more but less than 70%) is 44.9% and open forest (cover density of

10% and more but less than 40%) is 42.8%. The forest cover in the Himalayan states is 22.65 million ha which constitutes 38.83% of their geographical area. Of the Himalayan forest cover, 17.46% is covered by very dense forest that with moderately disturbed forest is 44.82% and open forest is 37.72%. Some of the forests under 40–70% cover category could be almost undisturbed as in forests such as chir pine (*Pinus roxburghii*); the upper limit of a healthy stand may be less than 70% in several situations. If only the area that can support forest is considered (generally area up to 4000 m elevation) then potential forest cover in the Himalayas could be close to 60%, but a sizeable portion is degraded (open-type).

In a decade-long study in the Indian states of Uttarakhand, Himachal Pradesh, and Nepal, based on ground-level measurements, Baland (2009) found that more than 50% of forests have less than 40% canopy cover because of degradation. In them, trees were lopped about two-thirds of tree heights. Such denuded trees become unhealthy and vulnerable to climatic extremes, like drought, storms, and frost, and incapable of sequestering carbon and intercepting rain. This long-term project showed that 70% of the plots had a sign of high livestock trail density, lopping of trees was more than 30% of height in 80% plots, and stumps burned by fire were in 43 plots (Baland 2009).

Studies that assess degradation using a combination of optical remote sensing techniques, including high-resolution images (Souza 2003; Asner 2009), have focused only on one type of degradation, selective logging, which is not representative of degradation as it occurs in the Himalayas. Furthermore, these techniques can detect where degradation occurs, but cannot measure the loss of biomass below canopy or at a scale below the resolution of the images. Prabhakar et al. (2006) argue that the widely cited results of FSI considerably underestimate the extent of degradation. A comparison made by Prabhakar et al. (2017) indicates that 78% of forest (including scrub) was degraded, whereas the FSI data indicate that only 19% was under the degraded category. The difference was largely because FSI treated highly degraded forest as non-forest area. Baland et al. (2010) conducted detailed ground-level forest surveys in Himachal Pradesh and Uttarakhand and showed that 61% of forest areas sampled exhibited crown cover below the ecologically sustainable threshold of 40% as used by FSI. Baland et al. (2010) suggest that majority of the forest in the 83 studied villages of Uttarakhand fall under the degraded category with an average percent canopy cover of 37.7 ± 11.1 and percentage lopping of 65.3 ± 13.1 trees. Over 90% of firewood and 62% of leaf fodder were extracted from the local forests. In community-based interviews, 91% of the responses were that the village forest stock is depleting and 30% stated "drastic" forest degradation had occurred over the past 25 years.

3.2.1 Forest Biomass, Carbon Sequestration, and Other Ecosystem Functions

Forest degradation, as indicated by more loss of forest biomass than forest area, has been quite common in Uttarakhand (Rathore et al. 1997). In 16 years' time in the

Baliya catchment (29°17′–29–25′ N lat., 79° 25′–79° 35′ long) of Nainital, the forest area (2802 ha in 1973) declined by 35%, and the aboveground forest biomass by 40%, at the rate of 1800 t year^{-1}. This amount of forest biomass loss is equal to clear-cutting of 90 ha of a typical chir pine (*Pinus roxburghii*) forest. Forests were highly fragmented; above 50% of forest fragments were larger than 60 ha area. While light-demanding chir pine increased, several shade-adapted undercanopy species including *Rhododendron arboreum* were missing.

A study (Raikwal 2009) conducted in Kumaun Himalayas indicates that degradation in an oak forest proceeds, typically total tree density decreases, but the chir pine density increases proportionally, provided its stands are in proximity (Table 3.1). Oak produces seedlings (individuals up to 30 cm height), but not many seem to reach sapling stage (>30 cm height up to 2 m and <7.85 cm girth at breast height (GBH) 1.37 m). At Mukteshwar site, Kumaun Himalayas (S1), only chir pine saplings were present, indicating that disturbance will eventually result in the conversion of oak forest into chir pine forest. In most disturbed sites, chir pine already accounted for about half of trees. If chir pine were not there, the disturbed sites would have only 20% of trees of an undisturbed forest stand. The presence of a large number of denuded oak trees due to lopping of branches and leaf-bearing twigs is a common sight in agriculture-dominated areas. Compared to undisturbed oak forest stand (D_0), the most disturbed forest stand (D_2) had 52.3% less tree density and 86.4% less leaf area index (Table 3.2). The impact of drastic reduction in leaf area is reflected in the difference between stand biomass and carbon sequestration rate. While tree biomass values in D_2 were 44–60% of D_0, the carbon sequestration rates based on repeated girth measurements in D_2 were only 20.4–21.6% of that in D_0. These high drop-offs in carbon sequestration are comparable with that of leaf area index. Though the impairment of hydrologic regulation was not studied, the drop in soil moisture due to forest degradation gives evidence of drastically reduced canopy interception and increased runoff.

When forest degradation proceeds in a forest stand, a stage comes when even a small amount of biomass removal is enough to disintegrate ecosystem processes. People then increase the forest area from where they collect biomass but without stopping collection from already degraded sites. Soil compaction and consequent reduced water infiltration and increased evaporation are likely to make soil dry.

However, eventually a time may come when biomass collection is stopped as it becomes extremely unrewarding. The effect of such a degraded condition can also be seen in a lower leaf litter decomposition rate. The annual oak leaf litter weight loss in a degraded forest is merely 45.6%, compared to 77.45% in an undisturbed forest stand (Table 3.2) (Singh et al. 2014). That forest degradation leads to slower rates of litter decomposition, and hence nutrient turnover rates emphasize its adverse impact on ecosystem processes.

Litter collection by local villagers was the highest in D_0 forest stands, as one can collect more litter from a place in a given time when it is abundant. It decreased with degradation, but litter removal as the fraction of litter fall increased with forest degradation (Table 3.2). In the least degraded (D_0) and moderately degraded forest stands, the annual litter collection was about 28% of annual litter fall, but in the

Table 3.1 Changes in species composition and regeneration in oak forest with increasing degradation. This study is located in Mukteshwar at 1400 m–2200 m altitudes

Tree vegetational parameters											
D0				D1				D2			
Species	Density (tree/ha)	MBA (cm^2/tree)	IVI	Species	Density	MB (cm^2/tree)	IVI	Species	Density (tree/ha)	MBA (cm^2/tree)	IVI
Q. leucotrichophora	450	1090.14	166.23	*Q. leucotrichophora*	450	1103.4	206.68	*Q. leucotrichophora*	167	385.86	146.2
P. roxburghii	17	17.52	11.06	*P. roxburghii*	50	221.43	29.4	*P. roxburghii*	183	374.15	153.8
R. arboreum	17	176.28	20.53	*R. arboreum*	17	430.58	25.08				
M. esculenta	67	68	17.31	*M. esculenta*	33	257.2	27.1				
Q. floribunda	133	840.47	53								
L. ovalifolia	33	80.88	13.2								
P. pashia	17	56.01	11.15								
Total	**734**	**2329.3**	**300**	**Total**	**550**	**2012.6**	**300**	**Total**	**350**	**760.01**	**300**
Sapling											
Q. leucotrichophora	33	3.9	83.81	*Q. leucotrichophora*	67	3.21	82.7	*P. roxburghii*	67	6.88	300
R. arboreum	67	4.94	139.1	*R. arboreum*	100	4.66	153.83				
Q. floribunda	33	6.59	77.09	*M. esculenta*	33	5.75	63.48				
Total	**133**	**15.43**	**300**		**200**	**13.62**	**300**		**67**	**6.88**	**300**

Seedling layer											
Q. leucotrichophora	33	0.011	105.85	*Q. leucotrichophora*	33	0	27.861	*Q. leucotrichophora*	433	0.001	122.886
Q. floribunda	267	0.002	194.15	*M. esculenta*	33	0	27.861	*P. roxburghii*	367	0.002	117.827
				R. arboreum	67	0.008	90.942	*Q. floribunda*	33	0.01	59.827
				F. nemoralis	100	0.002	72.894				
				L. ovalifolia	33	0	28.185				
				C. deodara	33	0.008	52.258				
Total	**300**	**0.13**	**300**	**Total**	**299**	**0.019**	**300**	**Total**	**833**	**0.013**	**300**

Source: Raikwal (2009)

D_0, largely undisturbed forest; D_1, moderately disturbed forest; D_2, degraded forest

Table 3.2 Leaf litter characteristics and leaf area index in relation to forest degradation

	Litter removal ($t\ ha^{-1}$)		Litter fall ($t\ ha^{-1}$)		Litter decomposition ($t\ ha^{-1}$)		LAI
Disturbance Category	S1	S2	S1	S2	S1	S2	S1
D_0	–	1.73	5.94	6.17	73.9	77.4 ± 0.0	2.94 ± 0.1
D_1	–	1.21	–	4.3	–	61.1 ± 0.0	2.11 ± 0.1
D_2	–	1.05	–	1.8	48.42	45.5 ± 0.0	0.4 ± 0.0

Source: Raikwal (2009) and Singh (2009)
S1, Mukteshwar (Site 1); S2, Lamgarha (Site 2); D_0, largely undisturbed forest D_1, moderately disturbed forest; D_2, degraded forest

Table 3.3 Impact of degradation on biomass stocks and carbon sequestration rate

Disturbance category	Biomass ($t\ ha^{-1}$) S1	Carbon sequestration ($t\ ha^{-1}\ year^{-1}$) S1	Biomass ($t\ ha^{-1}$) S2	Carbon sequestration ($t\ ha^{-1}\ year^{-1}$) S2
D_0	631	4.02	518.24	2.41
D_1	564.1	2.35	404.08	1.48
D_2	280.9	0.82	311.66	0.52

Source: Raikwal (2009) and Singh (2009)
S1, Mukteshwar; S2, Lamgarha; D_0, largely undisturbed forest; D_1, moderately disturbed forest; D_2, degraded forest

severely degraded stand, it jumped to about 58% (Tables 3.2 and 3.3). Thus, the rate of degradation accelerates as a forest stand reaches a degradation point.

3.2.2 Fine Roots and Mycorrhizae

Because of forest degradation, the root mass declined far more than the forest biomass. In banj oak D_2, the forest biomass was 42.7% of undisturbed forest (D_0), whereas the fine root mass D_2 was only 19.4% of $D_{0.}$ The mycorrhizal association measured as sporocarp number in D_2 was about 40% of D_0 as well as D_1 (Raikwal 2009). Sporocarps of mycorrhizae were quite low also in chir pine forest, compared to that of oak forest (70% of D_0). Reduced litter cover and organic matter, higher light intensity, and other factors associated with forest degradation are likely to adversely affect mycorrhizal growth. Among the sporocarp bearing fungi, *Agaricus arvensis* and *Russula vesca* were the most abundant in undisturbed stands. Fungal sporocarp density remained unaffected by initial forest degradation but was drastically reduced in severe degradation, the latter accounting for only 4% of sporocarp density of 0.25 individuals in comparison to D_0 (Table 3.4).

3.2.3 Grazing Trails

One of the major consequences of grazing, apart from biomass removal, is the formation of grazing trails, which in a way represent the cumulative degradation over

Table 3.4 Impact of forest degradation on fine root density, fine root mass, and fungal sporocarp density

Disturbance category	Fine root density (0–20 cm) depth 1 × 1 m^2	Fine root mass (kg ha^{-1}) top layer	Fungal sporocarp density (5 × 5 m^2)
D_0	1077.8 ± 63.7	331	6.20
D_1	761.1 ± 78.5	242	6.26
D_2	147.2 ± 13.4	64.1	0.25

Source: Raikwal (2009)

D_0, largely undisturbed forest; D_1, moderately disturbed forest; D_2, degraded forest

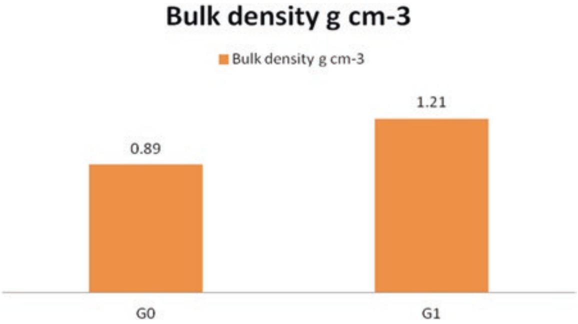


Fig. 3.1 Effect of grazing trail on soil bulk density, outside grazing trail (G0) and within grazing trail (G1). (Source: Joshi et al. 1991)

time. In extreme situations the density of grazing trails increased, and they were almost devoid of soil carbon. In an area where degradation has slowed down, grazing trails are disappearing, so they can be used as an indicator of recovery. A comparison between trails and the remaining area of a degraded forest site indicated that the animal trampling greatly reduced soil content and increased sand (by 11–14%) and bulk density (by 20%), decreasing water holding capacity (by 18–32%) (Figs. 3.1 and 3.2).

3.2.4 Impact of Degradation on Seed and Seed Germination

There is evidence to suggest that the percentage of healthy oak seeds (not infested with weevils) declines with forest degradation and seeds get more desiccated. Recent climatic warming seems to combine with the degradation to reduce the seed moisture content below the critical level (25–30% of fresh seed weight, Upreti et al. 1985) required for seed germination. It may be recalled that banj oak seeds are dispersed during winters and remain on the ground for several months and germinate when conditions become warm and moist due to rain. In contrast, pine seeds are orthodox, and they are dispersed just before the monsoon, so the probability of desiccation is minimal. This is one of the reasons why in a degraded oak forest, oaks fail to regenerate and chir pine takes over. Banj oak seeds are recalcitrant and become non-viable after the loss of seed moisture to a certain point, generally below 25–30%.

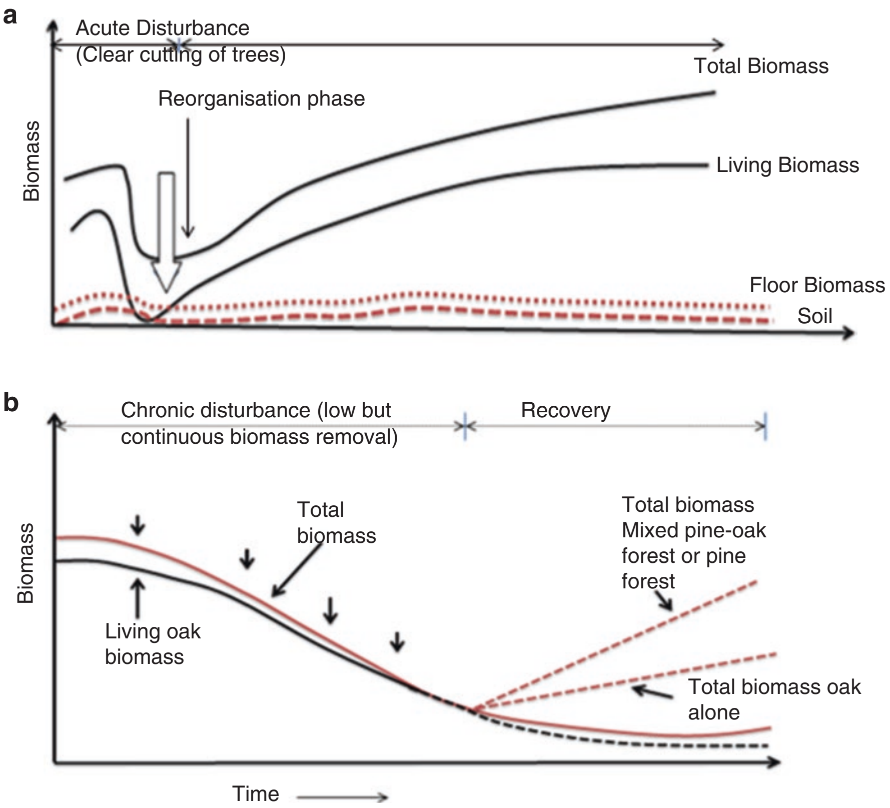


Fig. 3.2 A representation of recovery of banj oak forest (*Quercus leucotrichophora*) following (**a**) an acute disturbance (based on Cousens et al. 1980) and (**b**) chronic disturbance. Total biomass includes living biomass and dead organic matter in forest floor, dead wood, and mineral soil. In both it is assumed that during recovery phase, there is no anthropogenic disturbance. The chronic disturbance includes "day-to-day" lopping of oak tree branches and leaves, and litter collection for composting, to be transferring to crop fields. Small surface fires, grazing and NTFP's (non-timber forest products) collections are also part of chronic disturbance. Oak stand can lead to arrival of chir pine, and eventually conversion of banj, into a mixed or pine forest. Or in an acute disturbance, total biomass declines during the first 10 or 20 years in spite of the increase in biomass; it is referred to as reorganization phase. (Cousens et al. 1980)

3.3 Structural and Physiological Changes in Individual Trees Due to the Chronic Disturbances of Stands

Studies on individual tree level impacts of chronic disturbance are limited. Zobel and Singh (1997) made an attempt to study tree level changes due to forest degradation in banj oak and chir pine forest of Nainital. Compared to a largely intact oak stand, the most degraded oak stand had more than four times higher livestock trail density and about 64% of crown cover, 55% crown enclosures, and 53% forest floor litter mass. While trees in the intact oak forest were hardly lopped, in the most

Table 3.5 Structural and physiological changes in the tress of banj oak (*Quercus leucotrichophora*) and chir pine (*Pinus roxburghii*) due to chronic disturbance of the stands. Sites were located in Nainital, India (29° 23′ N Lat., 79° 27′ E Long)

Tree character	Impact of disturbance
Twig diameter	Increased with disturbance, thickest twigs being in MD; more thickening due to disturbance in oak than pine
Specific leaf area ($cm^2\ g^{-1}$)	Lowest in MDs
Difference in predawn-midday tree water potentials difference	Increased with disturbance largely because of more negative midday water potential
N concentration in xylem sap	Decreased with disturbance at the time of leaf expansion in May

Derived from Zobel and Singh (1995)

LD least disturbed, *M1D* moderately disturbed, *MD2* most disturbed (Zobel and Singh 1995)

degraded stand, 73% of them were lopped. Differences between the least and most disturbed pine stands for above characters were less apparent because of the lesser dependence of local communities on pine trees in comparison to oak trees for meeting "day-to-day" needs. Success in identifying useful indicators of stress, based on tree individuals, was less than expected. The banj oak trees of the most disturbed stands had lower specific leaf area, thicker twigs, higher difference between predawn and midday tree water potential, and lower xylem sap nitrogen concentration (measured in May, Table 3.5). Greater twig thickness and specific leaf mass seem to be correlated with small crown volume and open condition. Zobel and Singh (1995) referred to the observation that the concentration of starch produced decreased with disturbance in certain months. It seems that stand level characters are easier to measure than tree level characters as stress indicators.

3.3.1 Parasite Infestation in Degraded Oak Forests

Oak forests (*Q. leucotrichophora* and *Q. floribunda*) are affected by parasitic (hemiparasitic) effects of mistletoe (*Taxillus vestitus*). Lopping of trees promotes the growth of the mistletoe population (Gumber et al. 2017), by increasing light availability to the parasite and providing cut-end surfaces for seed germination (Garkoti et al. 2002). The mistletoe causes stress in the oak hosts by extracting water and passively taking up sap nitrogen. The parasite is able to maintain higher leaf water conductance than the host by keeping lower water potential through osmotic adjustment and by keeping stomata open with a lower plant water potential than the host. Furthermore, the presence of the parasite reduces the leaf water conductance of the host (Garkoti et al. 2002). For maintaining a high rate of photosynthesis, the parasite continues to extract water and minerals from the host even during droughts, thus further degrading the forest. Because of the increased stress of pre-monsoon drought due to global warming, the impact of parasites on oak forests may aggravate.

3.4 Impact of Ban on Tree Cutting on Forest Degradation and Species Composition

The 1927 Forest Act during the British Raj codified the reservation of forests, marked forest boundaries, and prohibited the village people from cultivating forested land, lopping trees, or grazing livestock in the forest (Tucker 1986; Tucker and King 1984; Rawat and Singh 1989). Establishment of forest reserves by the British led to a widespread protest by the people and several incidents of incendiaries. The protest resulted in the creation in 1930 of Van Panchayats (community-managed forests) under which a certain forest area was allotted to a particular village to meet the people's needs. However, the area in most cases was not enough, so these forests also suffered from degradation. Commercial tree cutting through forest contractors established under the British Forestry continued after Independence. Major timber species like *Pinus roxburghii*, *Shorea robusta*, and *Cedrus deodara* were put to rotational harvesting based on the rate of growth of species and maturity period. Local people began to resist the system in which contractors often over-harvested trees. Eventually, it resulted in what is widely called "Chipko Andolan" (women protecting trees by hugging them) and then in a ban on tree cutting above 1000 m elevation. The plea was that forest cover is required to protect mountain slopes and watersheds. It was expected that the ban on tree cutting would conserve forests and local people would be able to meet their biomass needs sustainably.

The ban on tree cutting in 1980 did improve forest condition at the regional scale then. Ironically, the ban favored chir pine at the expense of banj oak and other broadleaf tree species, because chronic forms of disturbance continued even after the ban on tree cutting. The ban on "whole tree cutting" may have increased the legitimacy of lopping of branches, resulting in an accelerated denudation of trees. There are several instances of the intrusion of the chir pine into oak forests, due to increased light after the reduction of oak canopy cover as a result of canopy thinning. Subsequent to the ban on tree cutting, silvicultural practices, such as thinning and pruning in chir pine, stopped, resulting in the densification of stands. Thus, the open-type chir pine forest stands are being converted into dense stands consisting of weak stems with thin bark. How the densification of pine stands is affecting ecosystem structure and processes and fire resistance of trees is hardly known. Thick bark is one of the adaptations of chir pine to surface fires (Singh and Singh 1992); thus, stems with thin bark are likely to be vulnerable to fire.

3.5 Recovery of Degraded Oak Forest Ecosystem

When a forest is clear-cut (acute disturbance), live biomass is drastically reduced, while soil and litter pools remain by and large unaffected. Subsequent to clear-cutting of trees, a reorganization phase of 10–20 years sets in, in which overall forest carbon declines in spite of the accumulation of living biomass, because of the rapid decomposition of organic matter of soil and forest floor (Bormann and Likens 1979). Depending upon the practice of biomass removal after tree cutting, the amount of dead wood and roots left in a forest varies. During the reorganization phase, it declines rapidly because of oxidation and consumption by heterotrophs and transfer to forest floor and soil (Cousens et al. 1980).

Similarly, forest floor gets quickly depleted after tree cutting. All these organic matter components are largely incorporated in the soil. Organic matter up to 45 cm soil depth was estimated at 173 t ha^{-1} in the Hubbard Brook forest study (Borman and Likens 1979), and in some of the Himalayan forests, it was found to range from about 162 to 241 t ha^{-1} (Singh and Singh 1992).

In contrast, in chronic forest disturbance, along with the decrease of live biomass, often over several years, the amounts of soil and litter carbon also decline. As described earlier, soil gets compacted and eroded, restricting recovery processes. The live biomass structure of a degraded forest is considerably altered, with proportionately more stems and coarse roots, and less leaves and fine roots, compared to a normal forest. In a study on degraded forests, soil carbon was found to be about 69% of an undisturbed forest. However, when only the top 30 cm soil was considered, then the degraded forest had only 42% of total carbon (Singh et al. 2011).

A degraded oak forest is expected to recover at a slower rate than an undisturbed forest after a clear-cutting because of soil deterioration and slow responses of trees with depleted photosynthetic surfaces. Tree stems may respire off more carbon than their lopped canopies fix carbon in photosynthesis. Global warming may increase the stress, particularly because of intensified pre-monsoon drought (Singh 2017). Warmer pre-monsoon temperatures are likely to enhance evapotranspiration loss and may combine with decreasing pre-monsoon rainfall to deteriorate the forest condition.

Regeneration through oak seeds will remain slow for a quite long period until tree crowns become large enough to have seeds in sufficient quantities. Here, global warming may surface up as a negative factor, as seeds fallen on the ground may rapidly desiccate during intensified pre-monsoon droughts (Singh et al. 2017). Then, for a quite long period, recovery of living biomass of trees would be based on coppicing and a slow build-up of foliage mass. Exotic species, shrubs, and coppiced oak shoots are likely to be the main constituents of a degraded oak forest given an opportunity to recover. A restored degraded forest is likely to be a novel community with characters different from those recovering after clear-cutting or other forms of acute disturbance. For example, several oak trees might have hemiparasites, surrounded by thickets of shrubs, overtopped by chir pine trees. Even if the species composition is similar to original forest stands, the forest structure is likely to be markedly different. The food base of birds in the restored degraded forest might include seeds of hemiparasites, and shrubs, both native and exotics.

Forest restoration can be speeded up by increasing water infiltration and sowing oak acorns through dribbling (Thadani 2008). Where soil condition is severely deteriorated, alder (*Alnus nepalensis*) seedlings could be planted. In a study on restoring landslide sites in Kumaon, alder was found to perform very well in improving soil. In the rhizosphere of planted alder, N concentration was from 0.38% to 0.65%, compared to 0.04–0.31% in other planted species, and the organic carbon was

1.28–1.47%, compared to 0.21–1.2% for others (Chaudhary et al. 2012). The total N in soil can be increased ten times within 3–4 years by planting alder.

3.6 Comparison with Forests Affected by Shifting Cultivation

In the Eastern Himalayas, forest disturbance is somewhere between acute and chronic forms. In shifting agriculture or "jhuming," a forest patch is burnt, and then mixed crops are cultivated for some years, until soil fertility is no more enough to grow crops. The return time to the burnt site which used to be 25–30 years in the past is reduced now to 5–6 years. The people living in or near the forest continue to have a constant impact on the neighboring forests, resulting in forest degradation at a regional scale. However, even with a shortened cycle, a forest land gets 5–6 years to recover, which is not the case with the settled agriculture.

While branch lopping leads to the formation of a forest stand having disproportionately higher biomass of stems and little foliage, the shifting cultivation with shortened cycles leads to a form of stands with higher proportions of crown than normal forest stands. Soil quality in shifting cultivation deteriorates, but not to the extent seen in forests with lopped trees. Tree stumps of young trees left in shifting cultivated fields may rapidly sprout in species genetically capable of sprouting (Khan and Tripathi 1989), and shoots grow faster than seed-originated individuals.

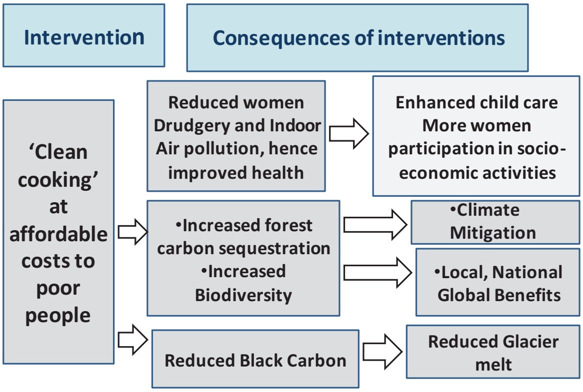
However, recovery of shifting agriculture-affected stands can be obstructed by invasive species and grasses like *Imperata cylindrical* which often establish after the abandonment of cultivation.

3.7 Interventions to Check Forest Degradation

Use of firewood for cooking is one of the principal causes of forest degradation. The pressure is the highest on oaks, because their dense wood is highly preferred as cooking fuel. In Himachal Pradesh firewood was the sole cooking energy in 90% of households during summer and 99% of households during winter season during 2000–2003. The remaining households used gas, but they too collected wood for cooking, and generally LPG was used for making tea and cooking a few other things. The percentage of households using LPG seems to have increased in recent years, but not many families depend for cooking entirely on LPG.

An exercise in Himachal Pradesh to find out the impact of gas subsidy indicated that about one-third subsidy can result in the use of cooking gas by 22% of families. The gains of providing cooking gas to people on an affordable cost can be enormous (Fig. 3.3) including women's health and child care, apart from recovery of forests and biodiversity. An additional gain of providing a clean cooking energy is the reduction in black carbon, which is considered one of the major causes of glacier

Fig. 3.3 A schematic representation of the beneficial effects of an energy intervention on forest recovery and women life



melt (Ramanathan and Carmichael 2008). Research indicates that air-borne particles including black carbon can reach high the Himalayas from distant areas including Gangetic Plains and other continents (Kopacz et al. 2011)

The other interventions which could recover a degraded forest including growing grass and legume-based fodder so that trees are not lopped for leaves to feed livestock. In the Lamgarha block of Uttarakhand, grass cultivation has resulted in reducing tree branch cutting (CHEA report 2015–2016). A proper composting of forest floor litter can also reduce litter removal.

3.8 Conclusion

Forest degradation due to "day-to-day" biomass extraction to support subsistence living, and shifting cultivation, is widespread in the Himalayas. Here, we have shown that its impact on ecosystem integrity is severe, adversely affecting several ecosystem processes with persistent scars. Not only forest structure and soil pools are adversely impacted by forest degradation; processes like litter decomposition and nutrient uptake with mycorrhizal aid are also impaired for a quite long time. We know very little about the recovery mechanism of such a degraded forest. Since the nature of forest degradation in the Himalayas is unique, it is not possible to apply knowledge based on forest degradation in other regions. There is a need to monitor how climate change is combining with forest degradation in the Himalayas and affecting various ecosystem services, such as pollination, hydrological regulation, and carbon sequestration.

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References

Asner G (2009) Automated mapping of tropical deforestation and forest degradation: CLASlite. J Appl Remote Sens 3(1):033543

Baland J (2009) Degradation forestière himalayenne: quel rôle des pouvoirs publics? Afr Contemp 229(1):43

Baland J, Mookherjee D (2014) Deforestation in the Himalayas: myths and reality. [online] SANDEE, Kathmandu. Available at: http://admin.indiaenvironmentportal.org.in/files/file/Deforestation%20in%20the%20

Baland J, Bardhan P, Das S, Mookherjee D (2010) Forests to the people: decentralization and forest degradation in the Indian Himalayas. World Dev 38(11):1642–1656

Bormann FH, Likens GE (1979) Pattern and process in a forested ecosystem: disturbance, development and the steady state based on the Hubbard brook ecosystem study. Springer Science & Business Media

Chaudhary D, Saxena J, Lorenz N, Dick L, Dick R (2012) Microbial profiles of rhizosphere and bulk soil microbial communities of biofuel crops switchgrass (Panicum virgatum L.) and jatropha (Jatropha curcas L.). Appl Environ Soil Sci 24:1–6

Cousens J, Borman F, Likens G (1980) Pattern and process in a forested ecosystem. J Ecol 68(2):697

DeFries R, Achard F, Brown S, Herold M, Murdiyarso D, Schlamadinger B, de Souza C (2007) Earth observations for estimating greenhouse gas emissions from deforestation in developing countries. Environ Sci Pol 10(4):385–394

Denman KL (2007) Couplings between changes the climate system and biogeochemistry climate change: the physical science basis. Contribution of working group I to the fourth assessment report of the intergovernmental panel on climate change. Cambridge University Press, Cambridge

FAO (2015) Global forest resources assessment. How are the world's forests changing? 2nd edn. FAO, Rome 2016

Garkoti S, Akoijam S, Singh S (2002) Ecology of water relations between mistletoe (Taxillus vestitus) and its host Oak (Q. floribunda). Trop Ecol 43(2):243–249

Gumber S, Tewari A, Tewari B (2017) Loranthus (Taxillus Vestitus) infestation in mixed oak Forest sites in and around Nainital. Indian Forester 143(7):671–675

Hosonuma N, Herold M, de Sy V, de Fries RS, Brockhaus M, Verchot L, Angelsen A, Romijn E (2012) An assessment of deforestation and forest degradation drivers in developing countries. Environ Res Lett 7(4):4009

Ives J (1989) Deforestation in the Himalayas. Land Use Policy 6(3):187–193

Joshi M, Mer G, Singh S, Rawat Y (1991) Seasonal pattern of total soil respiration in undisturbed and disturbed ecosystems of Central Himalaya. Biol Fertil Soils 11(4):267–272

Khan ML, Tripathi RS (1989) Effects of stump diameter, stump height and sprout density on the sprout growth of four tree species in burnt and unburnt forest plots. Acta Oecol 10(4):303–316

Kopacz M, Mauzerall DL, Wang J, Leibensperger EM, Henze DK, Singh K (2011) Origin and radiative forcing of black carbon transported to the Himalayas and Tibetan Plateau. Atmos Chem Phys 11:2837–2852

Prabhakar R, Somanathan E, Mehta BS (2006) How degraded are Himalayan forests? In: Current science, pp 61–67

Prabhakar R, Somanathan E, Singh B (2017) How degraded are Himalayan forests? on JSTOR. Available via Jstor.org. http://www.jstor.org/stable/24094176

Raikwal D (2009) Effect of leaf litter removal on soil nutrients in the Central Himalayan banj oak and chir pine forests with relation to carbon sequestration. Thesis, Kumaun University

Ramanathan V, Carmichael G (2008) Global and regional climate changes due to black carbon. Nat Geosci 1(4):221–227

Rathore SKS, Singh SP, Singh JS, Tiwari AK (1997) Changes in forest cover in central Himalayan catchments: inadequacy of assessment based on forest area alone. J Environ Manag 49:273–285

Rawat YS, Singh JS (1989) Forest floor biomass, litter fall and nutrient return in central Himalayan oak forests. Vegetation 82(2):113–125

Singh SP (1998) Chronic disturbance, a principal cause of environmental degradation in developing countries. Environ Conserv 25:1–2

Singh V (2009). Biomass stock and carbon sequestration rates in banj oak (Qusercus leucotrichophora, A. Camus.) forests under different disturbance regimes in Central Himalayas. Thesis, Kumaun University

Singh S (2017) Climate change in Himalayas: research findings, complexities and institutional roles- 23rd GB Pant Memorial Lecture- GBPNIHESD. Kosi-Katarmal, Almora

Singh JS, Singh SP (1987) Forest vegetation of Himalaya. Bot Rev 52:80–192

Singh JS, Singh SP (1992) Forests of Himalaya, structure, functioning and impact of man. Gyanodaya Prakashan, Nainital

Singh S, Thadani R (2015) Complexities and controversies in Himalayan research: a call for collaboration and rigor for better data. Mt Res Dev 35(4):401–409

Singh JS, Pandey U, Tiwari AK (1984) Man and forests: a central Himalayan case study. Ambio 13(2):80–87

Singh SP, Bassignana-Khadka I, Singh Karky B, Sharma E (2011) Climate change in the Hindu Kush-Himalayas: the state of current knowledge. International Centre for Integrated Mountain Development (ICIMOD)

Singh V, Thadani R, Tewari A, Ram J (2014) Human influence on banj oak (*Quercus leucotrichophora*, A. Camus) forests of central Himalaya. J Sustain For 33(4):373–386

Singh DS, Tangri AK, Kumar D, Dubey CA, Bali R (2017) Pattern of retreat and related morphological zones of Gangotri Glacier, Garhwal Himalaya, India. Quat Int 444:172–181

Skutsch M, Zahabu E, Lovett J, McCall M, Singh SP, Trines E, Verplanke J, Karky BS, van Laake P, Banskota K, Basnet R, Ba L (2008) Dealing with emissions from dry forest degradation; a low cost community-based approach. Think Global, Act Local project, Kyoto

Souza C (2003) Mapping forest degradation in the Eastern Amazon from SPOT 4 through spectral mixture models. Remote Sens Environ 87(4):494–506

Tewari A, Singh V, Phartiyal P (2008) The potential of community managed forests for carbon trade. LEISA Mag 24:332–333

Thadani R (1999) Disturbance, microclimate and competitive dynamics of tree seedlings in Banj oak (*Quercus leucotrichophora*) forest of central Himalayas India. Thesis, Yale University

Thadani R (2008) Direct sowing of Acorns: A low cost reforestation technique for the Himalaya. Centre for Ecology, Development and Research (CEDAR), Dehradun, New Delhi

Tucker R (1986) The evolution of Transhumant Grazing in the Punjab Himalaya. Mt Res Dev 6(1):17

Tucker BE, King JL (1984) Dependence of sediment-filled valley response on input amplitude and valley properties. Bull Seismol Soc Am 74(1):153–165

Upreti N, Tewari JC, Singh SP (1985) The oak forests of the Kumaun Himalaya (India): Composition, diversity and regeneration. Mt Res Dev 5:163–174

Zobel DB, Singh SP (1995) Tree water relations along the vegetational gradient in the Himalayas. Curr Sci:742–745

Zobel DB, Singh SP (1997) Himalayan forests and ecological generalizations. Bioscience 11:735–745

Management of Invasive Alien Plants in Nepal: Current Practices and Future Prospects

4

Bharat Babu Shrestha

Abstract

Management of invasive alien species is increasingly challenging mainly because of the failure of past global efforts to slow down the rate of invasion and increasing globalization of trade and transport. Developing countries like Nepal are further constrained due to the lack of adequate scientific knowledge to inform policy and management. This has resulted in weak policy and management responses, thereby exposing the country to a high threat of further invasions. This paper presents a brief review of the diversity, distribution, and impacts of invasive alien plants (IAPs), current management practices and policy responses, and future prospects for their management in Nepal. At least 183 vascular exotic plant species (4 pteridophytes and 179 flowering plants) are naturalized in Nepal, and among these are 26 invasive angiosperm species, including 4 from the list of 100 of the world's worst invasive species. The IAPs have invaded agroecosystems and the natural environment including protected areas and Ramsar sites from tropical lowland to temperate mountain zones. Impacts of a few IAPs have been examined, and they range from habitat degradation and species displacement to negative impacts on the livelihood of farming communities. Cultural and physical methods are the common control measures adopted, while a few biological control agents have also arrived fortuitously. As a policy response, the National Biodiversity Strategy and Action Plan 2014 prioritized inventory, impact assessment, identification of dispersal pathways, public education and participation, and biological control programs. Future prospects for the IAPs management in Nepal includes eradication of *Myriophyllum aquaticum*, prevention of *Mikania micrantha* and *Chromolaena odorata* from being spread to western Nepal, inclusion of IAPs management in community forestry programs and conservation management plans of protected

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areas, invasion risk assessment of species before introduction, government funding for education and research, strengthening biological control programs, and regional collaboration through common management strategies.

Keywords
Biological invasions · Control measures · Naturalized species · Policy response · Regional collaboration

4.1 Introduction

Biological invasion is an important component of anthropogenic global environmental changes (Vitousek et al. 1997) and a leading cause of biodiversity loss (Bellard et al. 2016) and decline in ecosystem services (Vila and Hulme 2017). Despite some controversies, the negative impacts of biological invasions on environment and biodiversity are now unambiguous in local, regional, and global levels (Courchamp et al. 2017; Russell and Blackburn 2017). The economic cost of invasions has been estimated to be several hundred billion dollars in terms of management and negative impacts (Pimentel et al. 2001). The biological invasions have been further aggravated by climate change (Hellmann et al. 2008; Bellard et al. 2013) and the rapid globalization of trade and transport (Hulme 2009). Despite past efforts to manage biological invasions, the number of exotic species has been increasing across continents and taxonomic groups (Seebens et al. 2017). Therefore, the management of invasive alien species (IAS) is likely to be still more challenging in the future and will require additional investment in developing innovative technologies and increasing interdisciplinarity of invasion biology (Ricciardi et al. 2017). Considering these current and future threats from the IAS, the Convention on Biological Diversity (CBD) has called for developing strategies by contracting Parties to “prevent the introduction of, control or eradicate those alien species which threaten ecosystems, habitats or species” (Article 8 (h); CBD 1992). Similarly, Aichi Biodiversity Target 9 of the CBD states that “By 2020, invasive alien species and pathways are identified and prioritized, priority species are controlled or eradicated, and measures are in place to manage pathways to prevent their introduction and establishment” (Strategic goal B, Target 9; CBD 2010).

The problem of biological invasions is also increasing in Nepal, a small South Asian country with extreme variation on climate and topography (Tiwari et al. 2005; MFSC 2014a). The threat from biological invasions to the agriculture sector, the backbone of national economy, is very high, globally the third most threatened country (Paini et al. 2016). While a wide range of impacts, from habitat degradation to native species displacement on natural ecosystems, have also been perceived (MFSC 2014b, Chaudhary et al. 2016), management responses are inadequate to address this emerging environmental and economic problem. Furthermore, scientific data appears to be inadequate to guide management as well as to contribute to science-policy interface. In this chapter, I briefly review the diversity, distribution, and impacts of invasive alien plants (IAPs) in Nepal based on published literature and my own observations and highlight the existing practices of the IAPs management and the policy responses. In the final section, future prospects are discussed with some suggestions to improve current management practices.

4.2 Overview of Invasive Plants in Nepal

4.2.1 Diversity and Distribution

Botanical exploration in Nepal has >200 years of history (Rajbhandari 2016), but preparation of the flora of Nepal has not been completed yet. In such situation, documentation of naturalized plant species (i.e., exotic plant species which has established self-replacing population in wild; Pysek et al. 2004) is not only challenging but also remains incomplete. However, based on the checklists of the flowering plants of Nepal (Press et al. 2000) and recent taxonomic publications, efforts have been made to list the naturalized plant species of Nepal by Tiwari et al. (2005) and Siwakoti (2012). These authors, nevertheless, included the species of "pantropical distribution" without known native distribution in the list of naturalized plant species of Nepal. Since the native range of such species is currently unknown, their inclusion in the list of naturalized plant species of Nepal is ambiguous. Considering this limitation, an effort has recently been made to update the list of naturalized flowering plant species of Nepal (Shrestha et al. 2018a). According to this list, at least 179 exotic species of flowering plants (gymnosperm, 1 sp. (*Pinus patula*), and angiosperms, 178 spp.) have been naturalized in Nepal. Among them, 26 species (dicotyledonous, 23; monocotyledonous, 3) have been reported as invasive in Nepal (Shrestha et al. 2017, Table 4.1). They belong to 14 families with the highest number of species in Asteraceae (10 spp.) followed by Fabaceae (3 spp.) and Amaranthaceae (2 spp.); the rest of the families have one species each. The invasive alien plants (IAPs) reported from Nepal are either herbs or shrubs; none of the exotic tree species in Nepal have become invasive. Eighteen species of IAPs reproduce by seeds, six by both seeds and vegetative parts, and two wetland species (*Alternanthera philoxeroides* and *Myriophyllum aquaticum*) exclusively by vegetative method.

Four IAPs of Nepal are among the 100 of the world's worst invasive alien species (Table 4.1, Lowe et al. 2000). The list of 100 species also includes *Leucaena leucocephala* (Lam.) de Witt (Fabaceae) which was introduced as fodder plant in the tropics including Nepal (Jackson 1994). A tree native to Mexico region, it has escaped to natural ecosystem in the introduced ranges (e.g., Asia, Australia) and become invasive (Sankaran and Suresh 2013). At the sites of introduction in Nepal, this exotic tree is regenerating well with high chances of being "invasive" to natural ecosystems in the near future (Tiwari et al. 2005, personal observation of author).

In addition to the flowering plants, four exotic species of pteridophytes (*Azolla filiculoides* Lam. subsp. *filiculoides*, *Salvinia molesta* D.S. Mitch, *Adiantum raddianum* C. Presl, and *Pityrogramma calomelanos* (L.) Link) have been also reported to be naturalized in Nepal (Fraser-Jenkins et al. 2015). *S. molesta* has been included

Table 4.1 Invasive alien plants (IAPs) in Nepal (Shrestha et al. 2017)

SN	Name of IAPs	Common name	Local name	Family	Habit	Reproduction	Primary habitats invaded	Native range	First report
1	*Ageratina adenophora* (Spreng.) R.M.King & H.Rob.	Crofton weed	Kalo banmara	Asteraceae	Perennial subshrub	Seed	Forest and shrub land	Mexico	1952
2	*Ageratum conyzoides* L.	Billygoat	Raunne/ gandhe	Asteraceae	Annual herb	Seed	Agroecosystem	Central and South America	1910
3	*Ageratum houstonianum* Mill.	Blue billygoat weed	Nilo gandhe	Asteraceae	Annual herb	Seed	Agroecosystem	Mexico and Central America	1929
4	*Alternanthera philoxeroides* (Mart.) Griseb.	Alligator weed	Jala jambhu, Patpate	Amaranthaceae	Perennial herb	Stem (stolon)	Wetland	South America	1994
5	*Amaranthus spinosus* L.	Spiny pigweed	Kande lude	Amaranthaceae	Annual herb	Seed	Grassland and residential areas	Tropical America	1954
6	*Argemone mexicana* L.	Mexican poppy	Thakal	Papaveraceae	Annual herb	Seed	Agroecosystem	Tropical America	1910
7	*Bidens pilosa* L.	Black-jack/ hairy beggar-tick	Kalo kuro	Asteraceae	Annual herb	Seed	Agroecosystem and grassland	Tropical America	1910
8	*Chromolaena odorata* (Spreng.) R.M.King & H.Rob.[a]	Siam weed	Seto banmara	Asteraceae	Shrub	Seed	Forest and shrub land	Mexico and central and South America	1825
9	*Eichhornia crassipes* (Mart.) Solms.[a]	Water hyacinth	Jalkumbhi	Pontederiaceae	Perennial herb	Seed + stem stolon	Wetland	South America	1966

10	*Erigeron karvinskianus* DC	Karwinsky's fleabane	Phule jhar	Asteraceae	Perennial herb	Seed	Agroecosystem	Mexico and Central America	?
11	*Galinsoga quadriradiata* Ruiz & Pav.	Shaggy soldier	Jhuse chitlange	Asteraceae	Annual herb	Seed	Agroecosystem	Mexico	1963
12	*Hyptis suaveolens* (L.) Poit.	Bushmint	Tulsi jhar	Lamiaceae	Annual herb	Seed	Shrub land and grassland	Tropical America	1956
13	*Ipomoea carnea* ssp. *fistulosa* (Mart. ex Choisy) D.F. Austin	Bush morning-glory	Besaram	Convolvulaceae	Shrub	Seed + stem	Wetland	Mexico and central and South America	1966
14	*Lantana camara* L.[a]	Lantana	Kirne Kanda	Verbenaceae	Shrub	Seed	Forest and shrub land	Central and South America	1848
15	*Leersia hexandra* Swartz.	Southern cut grass	Karaute ghans, navo dhan	Poaceae	Perennial grass	Seed (rarely) + stem (rhizome)	Wetland	Tropical America	1820
16	*Mikania micrantha* Kunth[a]	Mile-a-minute weed	Lahare banmara	Asteraceae	Perennial climber	Seed + stem	Shrub land and grassland	Central and South America	1963
17	*Mimosa pudica* L.	Sensitive plant	Lajjawati	Fabaceae	Annual herb	Seed	Agroecosystem	Mexico to South America	1910
18	*Myriophyllum aquaticum* (Vell.) Verdc.	Parrot's feather		Haloragaceae	Perennial herb	Stem (rhizome)	Wetland	South America	?

(continued)

Table 4.1 (continued)

SN	Name of IAPs	Common name	Local name	Family	Habit	Reproduction	Primary habitats invaded	Native range	First report
19	*Oxalis latifolia* Kunth.	Purple wood sorrel	Chari amilo	Oxalidaceae	Perennial herb	Seed + stem (underground bulb)	Agroecosystem	Central and South America	1954
20	*Parthenium hysterophorus* L.	Parthenium	Pati jhar	Asteraceae	Annual herb	Seed	Grassland	Southern USA to South America	1967
21	*Pistia stratiotes* L.	Water lettuce	Kumbhika, panibanda	Araceae	Perennial herb	Seed + stem (stolon)	Wetland	South America	1952
22	*Senna occidentalis* (L.) Link.	Coffee Senna	Panwar	Fabaceae	Perennial subshrub	Seed	Grassland	Tropical America	1910
23	*Senna tora* (L.) Roxb.	Sicklepod senna	Tapre	Fabaceae	Annual herb	Seed	Grassland	South America	1910
24	*Spergula arvensis* L.	Corn spurry	Thangne jhar	Caryophyllaceae	Annual herb	Seed	Agroecosystem	Europe	1952
25	*Spermacoce alata* Aubl.	Broadleaf buttonweed	Alu pate jhar	Rubiaceae	Perennial herb	Seed	Grassland	West Indies and tropical America	1967
26	*Xanthium strumarium* L.	Rough cocklebur	Bhede kuro	Asteraceae	Annual herb	Seed	Grassland	America	1952

[a] Species in the list of 100 of the world's worst invasive alien species (Lowe et al. 2000)

in the updated list of “100 of the world’s worst” invasive alien species (Luque et al. 2014), while other three species are also invasive elsewhere in the world (e.g., McConnachie et al. 2003). However, their invasive status in Nepal is currently unknown. Naturalized bryophytes (Nirmala Pradhan,[1] personal communication, 30 July 2017) and algae (Shiva K. Rai,[2] personal communication, 29 July 2017) have not been reported from Nepal.

The majority of naturalized flowering plants are native of the Americas (74%) followed by Europe (8%) (Bhattarai et al. 2014). All four species of naturalized pteridophytes are also native to the Americas (Fraser-Jenkins et al. 2015). Among the IAPs, all are native to the Americas except *Spergula arvensis*, which is a native of Europe (Table 4.1). They are invading a wide range of habitats with their greatest occurrence in agroecosystems followed by wetlands and grasslands. The southern half of the country (Tarai, Siwalik, and Middle Mountains zones) with tropical and subtropical climate has more IAPs than the northern part (High Mountains and High Himal) (Shrestha 2016). Nearly four-fifth of the IAPs are found below 2000 m asl. The IAPs found at the highest elevation (ca. 3300 m asl) in Nepal is *Ageratina adenophora* (Siwakoti et al. 2016). It appears that from the southern lowland, the IAPs are moving northward and upward. For example, Siwakoti et al. (2016) reported that upper elevation limit of the distribution of four IAPs (*Ageratina adenophora*, *Ageratum houstonianum*, *Amaranthus spinosus*, and *Bidens pilosa*) was >800 m higher in 2016 than previous reports (Press et al. 2000; Tiwari et al. 2005). Ecological niche modellings have also predicted invasion by IAPs to additional areas including high elevation zones under future climate change scenarios (Shrestha et al. 2018d, Thapa et al. 2018). Population of some IAPs first established in eastern Nepal and now are moving westward. *Mikania micrantha* and *Chromolaena odorata* were found in eastern and central Nepal (east of 83° E longitude) (Tiwari et al. 2005), but currently they can be found in western Nepal too (west of 83° E longitude) (Poudel 2016). *Myriophyllum aquaticum* has been reported only from the Kathmandu Valley (Tiwari et al. 2005), but all other IAPs are found in more than one ecological and physiographic zone of Nepal (Shrestha 2016).

There is no comprehensive assessment of protected areas and Ramsar sites of Nepal for the presence of IAPs. However, review of a few past studies and recent field observations revealed that the protected areas and Ramsar sites of Tarai, Siwalik, and Middle Mountains contain several IAPs. In Tarai and Siwalik zones, 18 IAPs have been reported from Chitwan National Park, 15 from Parsa, 12 from Banke, 8 from Bardia, and 3 from Suklaphanta (Shrestha et al. 2016). Similarly, seven species have been reported from Koshi Tappu Wildlife Reserve (REDD Cell 2012). In mountainous zones, five species have been reported from Langtang National Park, nine from Annapurna Conservation Area, and seven from Manaslu Conservation Area (Shrestha et al. 2016). *Lantana camara*, *Chromolaena odorata*,

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and *Mikania micrantha* are the major IAPs in the protected areas of Tarai and Siwalik zones, while *Ageratina adenophora* is the most common IAPs in the mountainous zones. In Ramsar sites of Tarai, Siwalik, and Middle Mountains, the major IAPs are *Eichhornia crassipes*, *Ipomoea carnea* ssp. *fistulosa*, *Pistia stratiotes*, and *Leersia hexandra* (Siwakoti and Karki 2009; Shrestha et al. 2016). *Eichhornia crassipes*, the most damaging wetland IAPs globally, has been reported from Koshi Tappu, Beeshajari, and Pokhara lake cluster but not from Mai Pokhari, Jagadishpur, and Ghodaghodi lakes.

4.2.2 Impacts

The IAPs have a wide range of impacts ranging from biodiversity loss and habitat degradation to health hazards (MFSC 2014b, Shrestha et al. 2015; Chaudhary et al. 2016). However, objective assessments of ecological and economic impacts have been completed for only a few species in a few locations. The IAPs evaluated for impacts are *Mikania micrantha* (Murphy et al. 2013; Shrestha and Dangol 2014), *Parthenium hysterophorus* (Karki 2009; Timsina et al. 2011), *Chromolaena odorata* (Thapa et al. 2016), and *Ageratina adenophora* (Thapa et al. 2017). These studies have shown that the impacts of IAPs include habitat degradation of endangered wildlife (e.g., one-horned rhinoceros; Murphy et al. 2013), changes in plant species composition (Timsina et al. 2011; Shrestha and Dangol 2014; Thapa et al. 2016), and interference in tree seedling regeneration by IAPs (Thapa et al. 2016, 2017). Additionally, health impacts of *Parthenium hysterophorus* (Shrestha et al. 2015) and negative impacts on rural livelihood by three IAPs – *Chromolaena odorata, Lantana camara* and *Mikania micrantha* (Rai et al. 2012; Rai and Scarborough 2015) – have been reported. For the rest of the IAPs, including all wetland IAPs, there is no assessment available with respect to their impacts. Furthermore, the reported data seem inadequate to draw national-level impacts of IAPs on biodiversity conservation, ecosystem functions, human health, and socioeconomic conditions. Therefore, impacts of IAPs in Nepal are largely anecdotal and require objective assessments considering the diversity of IAPs and the habitats they have invaded.

4.3 Management

Management of IAPs has three major components: prevention, eradication, and control (Holt 2004). Prevention is protection of any politically or ecologically defined areas from being invaded by IAPs through strict sanitary measures and quarantine rules. It is the first line of defense and cost-effective in the long term but equally challenging in the increasingly globalized world. Eradication involves the complete elimination of IAPs and their propagules from the areas where they have been established to prevent reinvasion in the future. Eradication is possible when the population is small and confined in a small geographic area. It requires early detection of potential IAPs through regular monitoring and economic assessment (Radosevich et al. 2007).

Though eradication of serious IAPs is cost-effective in the long term, this has never been implemented in Nepal due to the lack of early detection, economic assessment, and necessary resources. The last option left for IAPs management is control which aims to keep their abundance to a minimum level so that economic and ecological impacts are minimized. It includes cultural, physical, chemical, and biological methods and their appropriate integration (Radosevich et al. 2007).

4.3.1 Current Management Practices

Current management practices of IAPs in Nepal involve control measures that include cultural, physical, chemical, and biological methods.

4.3.1.1 Cultural Methods

Cultural methods include control of IAPs during routine activities of farming, silviculture, and rangeland management. In agroecosystem, prevention (e.g., use of weed-free seed, cleaning agricultural implements before moving to areas not infested by IAPs), crop rotation, smoother crops, cover crops, and biomass harvesting are cultural methods (Radosevich et al. 2007). In natural ecosystems, biomass utilization, rotational grazing, and controlled burning are the examples of cultural methods. Implementation of in-country quarantine rules and other legal directives by landowners and practitioners is also a part of cultural methods.

In Nepal, there is no effective in-country quarantine or regulatory mechanism that prevents transport and introduction of IAPs to new and non-infested areas. The Internal [Plant] Quarantine Guidelines 2013 issued by the Plant Quarantine and Pesticide Management Centre intends to regulate insect pests, nematodes, and pathogens of crops, but does not include IAPs (PQPMC 2013). Communities use biomass of *Ageratina adenophora* and *Chromolaena odorata* as animal bedding and subsequently for production of organic manure. *C. odorata* and *Lantana camara* are used as firewood (Fig. 4.1). Biomass of forest IAPs, together with other sources, has also been used for the manufacture of biobriquette (Singh 2013). However, the use of biomass by a few companies and occasionally by a small proportion of communities does not have any significant impact at landscape and regional scales toward the control of IAPs.

In community-managed forests, users regularly remove ground vegetation, which also includes some of the forest IAPs (e.g., *Chromolaena odorata*, *Lantana camara*, *Ageratina adenophora*) as part of one of the forest management practices to help tree regeneration. Previously degraded and open canopy forests have changed into closed canopy forests after handover to the communities for management (Shrestha et al. 2010). It appears that abundance of shade-intolerant IAPs such as *C. odorata* has declined locally in such forests due to closure of forest gaps and increase in canopy density (Khaniya 2017).

4.3.1.2 Physical Methods

Physical methods include techniques that uproot, cut, or remove IAPs biomass as an effort to reduce their abundance. In Nepal, this method has been used for the control

Fig. 4.1 Collection of stems of invasive alien plants for firewood. (A) *Chromolaena odorata* at Panchkhal area of Kavre district (December 2016), (B) *Lantana camara* at Hetauda of Makawanpur district (October 2011)

of wetland IAPs such as *Eichhornia crassipes* and *Pistia stratiotes*. For example, *E. crassipes*, *P. stratiotes*, and *Leersia hexandra* are being periodically removed from Beeshajari lake system (a Ramsar site) of Chitwan National Park (WWF Nepal 2013; Thapa 2016, Fig. 4.2) and *E. crassipes* from Lake Cluster of the Pokhara Valley, another Ramsar site in Kaski district. The periodically removed biomass of *E. crassipes* from these wetlands is used for preparing handicraft items (e.g., pen holders, handbags) by local women's groups (Anonymous 2015a).

4.3.1.3 Chemical Methods

Use of chemicals for the control of IAPs in natural ecosystems has not been observed in Nepal. In agroecosystems, use of glyphosate and 2,4-D for *Alternanthera philoxeroides* (Ranjit 2013) and pendimethalin 30 EC and 2,4-D sodium salt 80 WP for *Spergula arvensis* has been recommended (Ranjit et al. 2010). *P. stratiotes* in paddy, *S. arvensis* in wheat, and *A. philoxeroides* in multiple crops have been increasingly problematic in recent years. The 2,4-D has been also used to control *Ipomoea carnea* ssp. *fistulosa* in Jagadishpur Reservoir, a Ramsar site (Siwakoti and Karki 2009). However, effectiveness of these chemicals in controlling the target weeds and their undesired effects on non-target plants have not been evaluated in natural or managed ecosystems.

4.3.1.4 Biological Methods

Biological methods of IAPs control use living biological organisms, often from the native distribution range of the IAPs, to reduce their population and abundance and minimize ecological and economic impacts (Radosevich et al. 2007). Though the

Fig. 4.2 Removal of *Eichhornia crassipes* and *Leersia hexandra* from Beeshajari lake (a Ramsar site) of Chitwan National Park (July 2012)

initial cost is high and the process is lengthy, long-term economic gains and ecological benefits of biological control methods are higher than that of chemical control. In Nepal, biological control agents have not been released officially through quarantine screening for any of the IAPs. However, biological control agents for two IAPs have arrived naturally and established in Nepal from the regions where they were released (Fig. 4.3). They are leaf-feeding beetle *Zygogramma bicolorata* Pallister and winter rust *Puccinia abrupta* var. *partheniicola* (Jackson) Parmelee for *Parthenium hysterophorus* (Shrestha et al. 2015) and stem-galling fly *Procecidochares utilis* Stone and leaf spot fungus *Passalora ageratinae* Crous and A.R Wood for *Ageratina adenophora* (Winston et al. 2014). Potential sources of fungal agents could not be traced, but the distribution patterns of the insect agents clearly indicated that they entered into Nepal from India where they were released (*P. utilis* in 1963 and *Z. bicolorata* in 1984) as biological control agents (Sushilkumar 2015).

Effectiveness of these agents in controlling target IAPs has not been evaluated systematically, but field observations showed that the effect was only marginal. Distribution of fungal control agents (*Puccinia abrupt* var. *partheniicola* and *Passalora ageratinae*) of both species is highly localized with apparently no effect on the target species. *Zygogramma bicolorata* seems to be the most effective biological control agent of IAPs present in Nepal, but its population is still small, and their effectiveness is erratic with year-to-year variation (Shrestha et al. 2015). *Procecidochares utilis* has reached to almost all areas where *Ageratina adenophora* is present, but its impact on the weed is insignificant (BB Shrestha, unpublished

Fig. 4.3 Biological control of invasive alien plants. (A) Defoliation of *Parthenium hysterophorus* by *Zygogramma bicolorata* in Makawanpur (August 2009), (B) Stem galling in *Ageratina adenophora* by *Procecidochares utilis* in Sindhupalchowk (November 2012)

data). Failure of *P. utilis* to control *A. adenophora* is probably due to heavy infection by indigenous parasitoids (Sushilkumar 2015).

Recently, the Nepal Agricultural Research Council (NARC) imported two weevils *Neochetina eichhorniae* Warner and *N. bruchi* Hustache from the USA (Florida) as an effort to biologically control *Eichhornia crassipes* (Anonymous 2015b). After laboratory rearing, they were released to Begnas and Phewa lakes of Pokhara valley. It is not clear whether a standard quarantine screening was performed before releasing them into natural habitats as was done by India when *N. eichhorniae* was released there in 1984 (Jayanth 1988). Impacts of these weevils on the population of *E. crassipes* in Begnas and Phewal lakes have remained unknown.

4.3.2 Policy Responses

4.3.2.1 National Report to the Convention on Biological Diversity (CBD)

As a signatory to the CBD, Nepal submitted five national reports between 1997 and 2014 (Table 4.2). The initial report simply reported that invasive alien species were an issue with no detail. Subsequent reports have included more details about the threats of invasive alien species in Nepal, finally leading to a series of priority activities in the most recent report. Accomplishment of these activities by the target year

Table 4.2 Identification of biological invasion as an environmental problem and the priority actions set in the national reports submitted by the Nepal Government to the secretariat of the Convention on Biological Diversity (IAS, invasive alien species)

Report series	Identification of the problems	Priority actions
First report (MFSC 1997)	Introduction of exotic species is an issue for biodiversity conservation.	None.
Second report (MSFC 2001)	Growing need to study issues related to IAS and undertake policy measures to control them.	None.
Third report (MFSC 2006)	Identified *Eichhornia crassipes*, *Ageratina adenophora*, *Lantana camara*, and *Mikania micrantha* as problematic IAS. Inadequate scientific information on the impacts of IAS. Lack of innovative technologies and financial resources to manage IAS.	Reduce abundance of *Eichhornia crassipes* in wetlands.
Fourth report (MFSC 2009)	IAS as one of the root causes of biodiversity (species) loss in Nepal. Lack of data and monitoring for identification of dispersal pathways and control of IAS. No specific institution in IAS issue and become an overlooked environmental problem. Need to estimate abundance of IAS and their impact on native biodiversity.	Prepare and implement management plan for three IAS (*Eichhornia crassipes*, *Mikania micrantha*, and *Parthenium hysterophorus*). Develop method for monitoring and management of IAS.
Fifth report (MFSC 2014a)	Biological invasion, along with climate change, as an emerging threat to biodiversity and livelihood. *Eichhornia crassipes* as a major threat to wetlands in tropical and subtropical regions. There may be increase in the number of IAS and rate of spread due to climate change.	National level inventory and impact assessment. Strengthening national capacity for identification, early detection, prevention, and management of IAS. Identification of dispersal pathways. Increase public education and prepare identification kits. Promote biological control program. Provide technical support to communities for management of IAS. Control water hyacinth and other IAS in five major wetlands.

2020 will help mainstreaming of IAS management activities in biodiversity conservation and sustainable development plans of the country.

4.3.2.2 National Biodiversity Strategy

In Nepal's first Biodiversity Strategy, the IAS was identified as a threat to all levels of biodiversity, but management strategies were not specified (MFSC 2002). In the second National Biodiversity Strategy and Action Plan 2014–2020, the threats of IAS to biodiversity and ecosystem functions have been enumerated (MFSC 2014b). To manage IAS, the Strategy planned for a nationwide distribution survey of the five most problematic IAS, development of an atlas for the identification and early detection of IAS, enhancement of the capacity of custom and quarantine offices, use of appropriate biological control agents, and public education and community participation. The Nepal Government also expressed these commitments in the fifth national report submitted to CBD (MFSC 2014a). The Strategy identified national institutions such as the Department of Plant Resources, Forest Research and Training Centre, Nepal Agricultural Research Council, and Tribhuvan University (Central Department of Botany) to manage and study IAS problems in the country.

4.3.2.3 Invasive Species in Legal Documents

There are a number of international conventions and other legal instruments (e.g., Convention on Biological Diversity, Ramsar Convention, International Plant Protection Convention) related to IAS, and to which Nepal is a signatory (Siwakoti and Shrestha 2014). In line with international conventions and treaties, and acknowledging the increasing impacts of IAS, a few national policies of Nepal such as National Wetland Policy (MFSC 2012), Agro-Biodiversity Policy 2008 (First Amendment 2014) (http://www.moad.gov.np/en/legaltext; accessed on 2 August, 2017), and Forest Policy 2014 (MFSC 2014c) call for controlling invasive species which threaten native biodiversity and ecosystems. The Plant Protection Act 2007 (GoN 2007) and the Plant Protection Regulation 2010 (GoN 2010) have included provisions to regulate the import of plants, plant products, and biological control agents; however, effective implementation of these provisions remains a challenge due to porous border and weak regulatory mechanisms. Similarly, National Seed Vision 2013–2025 also identified uncontrolled flow of and increased dependency on seeds of exotic crops, fruits, and vegetables as a threat to the seed sector development of Nepal (Seed Quality Control Center 2013). However, some other pertinent national acts (e.g., Forest Act 1993, Seed Act 1988 – First Amendment 2002) and policies (e.g., Rangeland Policy 2010) remain silent on the issues of IAS. For example, diversity of IAPs was found the highest in grasslands used for grazing among the various vegetation types in Tarai and Siwalik regions of central Nepal (Dhakal 2017). Nepal's Rangeland Policy 2010, however, has not identified IAPs as a threat to the pastureland though the Policy intends to set up mechanism for introduction of exotic species (NPAFC 2010).

4.3.2.4 National Strategy for Management of Invasive Alien Species

The most recent development on the policy response of Nepal to the IAS problem is the drafting of the Invasive Alien Species Management Strategy by the Ministry of Forests and Environment. This national strategy aims to manage IAS through (1) prevention and risk reduction; (2) control and eradication; (3) inventory, research, and monitoring; (4) capacity building and community education; (5) early detection and rapid response; (6) information exchange and integration; (7) policy improvement and institutional development; and (8) international collaboration and networking (Rajesh Malla at the Department of Forest Research and Survey; personal communication on November 29, 2017). The Strategy also envisages the formation of a National Coordination Committee for IAS with representatives from government organizations, academia, and individual experts for effective implementation of the Strategy. Approval of this Strategy by the government will be a significant national response toward achieving the Aichi Biodiversity Target 9 of the CBD.

4.3.3 Future Prospects for Management

4.3.3.1 Opportunity for Eradication and Prevention

Eradication is possible when the population of IAPs is small and confined in a small geographic area. There is an opportunity for eradicating *Myriophyllum aquaticum* that has been reported only from Kathmandu Valley in Nepal (Tiwari et al. 2005). The species was not reported from Nepal prior to 2000 indicating that it is relatively a recent invader. A native of South America, this aquatic weed has been introduced primarily as aquarium and ornamental plants to the rest of the world where the plant is a nuisance for recreational use of water, obstructs irrigation channels, reduces dissolved oxygen, and favors the growth of *Anopheles* mosquitoes (Hussner and Champion 2012). Due to these impacts, the species has been included as one of the 23 IAPs of the European Union concern for management (European Union 2017). A number of cultural, physical, and chemical control methods are available for this species that can be used for eradication of these small populations (Hussner and Champion 2012).

Mikania micrantha and *Chromolaena odorata*, both being included in the 100 of the world's worst IAS (Lowe et al. 2000), are common in eastern and central Nepal but present as a few satellite populations in western Nepal (Poudel 2016; Shrestha et al. 2016). This has provided opportunity for eradicating satellite populations and prevent westward dispersal of these species (also refer to Sect. 4.3.3.3). Similarly, two Ramsar sites in western Tarai, Jagadishpur and Ghodaghodi, are still free from *Eichhornia crassipes*. The species has heavily invaded other Ramsar sites such as Beeshajari and Koshi Tappu in central and eastern Nepal, respectively. It is considered as the world's worst IAPs in freshwater wetlands with significant impact on water quality, biotic communities, and livelihood of local people (Villamagna and Murphy 2010). Since the dispersal of *E. crassipes* is mainly due to introduction by human for its ornamental value (Yan et al. 2017), a similar introduction of this weed to Jagadishpur and Ghodaghodi in the near future cannot be ruled out. Therefore, regular monitoring by the concerned authorities and increasing public awareness of local communities are essential to prevent the introduction of *E. crassipes* in these two Ramsar sites of western Nepal.

4.3.3.2 Integrating Management of Invasive Plant Species in Community Forestry

Community forestry is one of the most successful participatory natural resource management programs in Nepal (Shrestha et al. 2010). This program has not only empowered the communities for forest management but has also increased the awareness of communities toward biodiversity conservation. Most of the Community

Forest User Groups (CFUGs) are aware of the rapidly spreading IAPs and their impacts, but the CFUGs do not have any strategy included in the Operational Plans of their forest management (Shrestha et al. 2018b). Inclusion of IAP management components in Operational Plans would encourage CFUGs to control existing IAPs and prevent invasion by new ones. This would also help to accomplish additional management activities included in the National Biodiversity Strategy and Action Plan (MFSC 2014b) and the [draft of] Invasive Alien Species Management Strategy (Sect. 4.3.2.4).

4.3.3.3 Management of IAPs in Protected Areas

The diversity and abundance of IAPs are expected to be low in protected areas due to filtering effect of boundary vegetation and lower anthropogenic disturbances than in the surrounding landscape (Foxcroft et al. 2011). However, this filtering effect may be weakened in situations where the parks are transected by highways (e.g., Bardia National Park) and flooded by mass tourism (e.g., Chitwan National Park). A number of IAPs have been reported from the protected areas of Nepal (Sec. 4.2.1), and their negative impacts have been perceived (Murphy et al. 2013). However, there has been no specific program targeted to manage IAPs except occasional removal of wetland IAPs from Beeshajari lake of the Chitwan National Park (WWF Nepal 2013; Thapa 2016). Management of IAPs in the protected areas often includes prevention through the management of dispersal pathways and control measures integrating physical, chemical, and biological methods (Foxcroft and Freitag-Ronaldson 2007; Foxcroft et al. 2013). These approaches are also relevant to the management of IAPs in the protected areas of Nepal. *Mikania micrantha* and *Chromolaena odorata* are spreading westward, but *M. micrantha* has not been reported from Banke, Bardia, and Suklaphanta National Parks in western Nepal, while *C. odorata* has established a few satellite populations in Bardia National Park (Poudel 2016; Shrestha et al. 2016). Possibility of eradication of *C. odorata* from Bardia National Park needs to be further assessed, but the spread of *M. micrantha* to these national parks can be prevented through regular monitoring of potential dispersal pathways and the surrounding landscape. Preparation of a complete list of IAPs in each protected area and their spatial distribution is very crucial for long-term management planning (Foxcroft et al. 2013). A preliminary list of IAPs is available for protected areas of the Tarai and Siwalik zones, while other protected areas in the mountain zones are mostly unexplored for the presence of IAPs. Spatial distribution data is available only for *M. micrantha* in Chitwan National Park (Murphy et al. 2013). Therefore, preparation of a complete list of IAPs in protected areas is the first and most urgent task to be accomplished. This needs to be followed by impact assessment. Information generated from these activities will help to prioritize species and habitat for the management and selection of appropriate control measures. Furthermore, the management of IAPs needs to be integrated adequately in the conservation management plan of the protected areas.

4.3.3.4 Risk Assessment

Risk assessment of species can reduce the chances of intentional introduction of potentially invasive species. It requires additional financial and technological investment that may discourage international trade, but in the long term, the assessment provides net bioeconomic benefits (Keller et al. 2007). Since the eradication of invasive species is seldom successful, species-based risk assessment and enhancing the accuracy of assessment are being suggested as components of effective management (Kumschick and Richardson 2013). In Nepal, growth performance of exotic plant species is often assessed before large-scale plantation (e.g., performance of *Paulownia* spp., Neil 1990), but there is no report of invasion risk assessment of the species introduced for forestry, agroforestry, and aquaculture. Hundreds of plant species have already been introduced as ornamental plants in forestry and agroforestry sectors (Jackson 1994). There is clear evidence that some of these species are in the naturalization phase and may turn out to be invasive. For example, *Hypoestes phyllostachya* Back. (Acanthaceae), a garden plant native of Madagascar, has spread beyond gardens and naturalized in the wild (Rajbhandary et al. 2012). Similarly, *Leucaena leucocephala* was introduced to Nepal as fodder plant but is spreading to the wild (Sect. 4.2.1). *Paulownia tomentosa* (Thunb.) Steud. (Scrophulariaceae) is another example of recent unregulated introduction of potentially invasive tree for commercial cultivation (Joshi et al. 2015). The species is a native of China and reported as invasive in the USA (Lovenshimer and Madritch 2017). Since the lag period between introduction and naturalization may vary from years to decades, several introduced species currently in "casual" stage may be naturalized in the future as illustrated by the above examples. These examples indicate that the introduction of potentially invasive species has continued in Nepal. Since the threat from invasive species is very high for Nepal (Paini et al. 2016), a robust risk assessment would reduce the chances of introduction of potentially invasive species in the future. A species reported as invasive elsewhere in its introduced range is highly likely to be invasive in new areas with suitable climate (Kumschick and Richardson 2013). Therefore, a conservative approach would be to ban importation of any species to Nepal that has been reported as invasive elsewhere.

4.3.3.5 Education and Research

Public participation is important for the successful implementation of invasive species management strategies. It is also essential from the ethical point of view and to meet legal compliance (Boudjelas 2009). Before public's active participation is anticipated, communities need to be educated on the current and potential impacts of IAPs. In Nepal, farming communities, including executive members of the CFUGs, are not aware of the process (e.g., origin of IAPs, dispersal mechanisms) and management (e.g., biological control) related to biological invasions, but the communities are able to prioritize the IAPs for management based on their negative impacts (Shrestha et al. 2018b). In this context, producing community education materials (e.g., identification kit, booklets) and widely disseminating the relevant information using mass media can increase community awareness. Community education is also one of the priority activities targeted in the biodiversity strategy

(MFSC 2014b), but the progress in this direction is almost negligible. The issues of biological invasions have not been covered in biodiversity and environment curricula of secondary school and undergraduate levels. However, this topic has been included recently in the master's level curricula of botany, zoology, and environmental sciences of Tribhuvan University.

Scientific research on IAPs started during the 1950s in Nepal but only 43 studies, 25 of them being theses and dissertations, were undertaken up to 2012 (Poudel and Thapa 2012). The first national-level inventory was published in 2005 and reported 166 naturalized plant species including 21 IAPs (Tiwari et al. 2005). After 2012, nearly a dozen of research papers on IAPs have been published, and at least five studies (e.g., Shrestha et al. 2016; Siwakoti et al. 2016; Shrestha et al. 2018c) focusing on inventory and distribution mapping have been completed. These studies mostly focused on species inventory and autecology with a few of them on ecological impacts (Timsina et al. 2011; Murphy et al. 2013) and socioeconomic dimensions (Rai et al. 2012; Rai and Scarborough 2015). Therefore, future research should focus on quantifying ecological and economic impacts, searching suitable management options, and providing cost-benefit analysis of different management options including biological control. Furthermore, there is no regular government funding currently available for research and management of IAPs in Nepal. Most of the past research was conducted either by students (for thesis and dissertations) or from the support of international funding agencies. Government funding will help to improve knowledge base, increase community participation, and develop innovative techniques for IAP management within the country.

4.3.3.6 Biological Control Program

Biological control methods are a major component of integrated management of IAPs with significant long-term benefits (Radosevich et al. 2007). However, a biological control program has not been mainstreamed in IAPs management, and none of the biological control agents of IAPs found in Nepal was released officially with quarantine screening (Sec. 4.3.1.4). Suitability and effectiveness of fortuitously arrived biological control agents have also not been assessed adequately. Therefore, it is time to formally initiate biological control programs for IAPs through policy formulation and capacity building. The National Biodiversity Strategy and Action Plan 2014 has identified the Nepal Agriculture Research Council (NARC) as the main implementing agency for biological control programs and universities as supporting agencies (MFSC 2014b). Some of the developmental partners of Nepal (e.g., America, Australia) have well-established biological control programs with state-of-the-art technologies (Julien et al. 2012). These countries could provide both technical and financial support to NARC to initiate the program.

4.3.3.7 Regional Collaboration

Biological invasion is a trans-boundary issue and effective management of invasive species requires harmonization of the approaches implemented among the countries sharing borders. All South Asian countries share borders with India and have close sociocultural, political, economic, and trade relationships. India surrounds Nepal from three sides, shares an open border, and is a major international trade partner. Both intentional and non-intentional introductions of species are closely linked with trade, transport, and travel (Hulme 2009); thus most of the IAPs (and their biological control agents) found in Nepal spread from India (Tiwari et al. 2005; Shrestha et al. 2015; Shrestha et al. 2018c). Due to the large geographic area, diverse climatic conditions, and the international trade with all other continents, it appears that the IAPs establish first in India and subsequently spread to other countries including Nepal. For example, *Parthenium hysterophorus* invaded India first and then spread to Pakistan, Bangladesh, Nepal and Sri Lanka; the pattern was also followed by its biological control agent *Zygogramma bicolorata* (Dhileepan and Senaratne 2009; Shrestha et al. 2015). Management of IAPs in India will have direct influence on the introduction rate of IAPs to other South Asian countries including Nepal. Therefore, Nepal has to collaborate with India and other South Asian countries to manage IAPs through a common regional strategy. Since the South Asian Association for Regional Cooperation (SAARC) is already in place as a political and economic forum, the same platform can be used to develop a regional strategy for biological invasion and other environmental problems shared by the region. A similar situation also exists in Europe, and the region already has a "European Strategy on Invasive Alien Species" (Council of Europe 2004).

4.4 Conclusion

At least 183 exotic vascular plant species are naturalized in Nepal and 26 of them are invasive. They have invaded varied land use types in tropical lowland to temperate mountain zones. *Myriophyllum aquaticum* is still confined to Kathmandu Valley and *Mikania micrantha* and *Chromolaena odorata* are absent in the western part of Nepal. Remaining IAPs are widespread to the extent of climatic suitability of the country. Impacts of a few IAPs have been examined, and they range from habitat degradation, species displacement, to negative impacts on livelihoods of farming communities. Cultural and physical methods are the common control measures, while a few biological control agents of *Ageratina adenophora* and *Parthenium hysterophorus* have also arrived fortuitously. As a policy response, the National Biodiversity Strategy and Action Plan 2014 has prioritized inventory, impact assessment, identification of dispersal pathways, public education and participation, and biological control program. Future prospects for the IAP management in Nepal includes eradication of *M. aquaticum*, prevention of *M. micrantha* and *C. odorata* from being spread to western Nepal, inclusion of IAP management in community forestry program and the conservation management plans of the protected areas, invasion risk assessment of species before introduction, government funding for education and research, strengthening biological control program, and regional collaboration through common management strategy.

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References

Anonymous (2015a) Water hyacinth generating income. The Himalayan Times URL: http://thehimalayantimes.com/nepal/water-hyacinth-generating-income/. Accessed 31 July 2017

Anonymous (2015b) Indigenous natural enemies exploration for the biological control of invasive weeds. Unpublished progress report (March 2014–February 2015) of the Project, Entomology Division, Nepal Agriculture Research Council (NARC), Khumaltar, Lalitpur, Nepal

Bellard C, Thuiller W, Leroy B, Genovesi P, Bakkenes M, Courchamp F (2013) Will climate change promote future invasions? Glob Chang Biol 19(12):3740–3748

Bellard C, Cassey P, Blackburn TM (2016) Alien species as a driver of recent extinctions. Biol Lett 12:20150623. https://doi.org/10.1098/rsbl.2015.0623

Bhattarai KR, Maren IE, Subedi SC (2014) Biodiversity and invasibility: distribution patterns of invasive plant species in the Himalayas, Nepal. J Mt Sci 11:688–696

Boudjelas S (2009) Public participation in invasive species management. In: Clout MN, Williams PA (eds) Invasive species management: a handbook of principles and techniques. Oxford University Press, Oxford, pp 93–107

CBD (1992) Article 8 (in situ conservation) of the convention of the biological diversity (CBD). United Nations https://www.cbd.int/convention/articles/default.shtml?a=cbd-08. Accessed on 31 May 2018

CBD (2010) Strategic plan for biodiversity 2011–2020 and the Aichi biodiversity targets. (Decision X/2). Secretariat of the Convention on Biological Diversity (CBD), Canada. https://www.cbd.int/decision/cop/?id=12268. Accessed on May 31, 2018

Chaudhary RP, Upreti Y, Rimal SK (2016) Deforestation in Nepal: causes, consequences and responses. In: Sivanpillai R (ed) Biological and environmental hazards, risks, and disasters. Elsevier Inc., Amsterdam, pp 335–372

Council of Europe (2004) European strategy on invasive alien species. Council of Europe Publishing, Strasbourg

Courchamp F, Fournier A, Bellard C, Bertelsmeier C, Bonnaud E, Jeschke JM, Russell JC (2017) Invasion biology: specific problems and possible solutions. Trends Ecol Evol 32:13–22

Dhakal S (2017) Diversity of naturalized plant species across vegetation types in Tarai and Siwalik regions of Central Nepal. MSc thesis, Central Department of Botany, Tribhuvan University, Kathmandu, Nepal

Dhileepan K, Senaratne KADW (2009) How widespread is *Parthenium hysterophorus* and its biological control agent *Zygogramma bicolorata* in South Asia? Weed Res 49:557–562

European Union (2017) Invasive alien species of Union concern [Brochure]. Publications Office of the European Union, Luxembourg City

Foxcroft LC, Freitag-Ronaldson S (2007) Seven decades of institutional learning: managing alien plant invasions in the Kruger National Park, South Africa. Oryx 41:160–167

Foxcroft LC, Jarosik V, Pysek P, Richardson DM, Rouget M (2011) Protected-area boundaries as filters of plant invasions. Conserv Biol 25:400–405

Foxcroft LC, Richardson DM, Pysek P, Genovesi P (2013) Invasive alien plants in protected areas: threats, opportunities, and the way forward. In: Foxcroft LC, Pyšek P, Richardson DM, Genovesi P (eds) Plant invasions in protected areas: patterns, problems and challenges. Springer Science+Business Media, Dordrecht, pp 621–639

Fraser-Jenkins CR, Kandel DR, Pariyar S (2015) Ferns and fern-allies of Nepal, vol 1. National Herbarium and Plant Laboratories, Department of Plant Resources, Lalitpur

GoN (2007) Plant protection act 2007. Government of Nepal (GoN), Kathmandu. www.lawcommission.gov.np/en/documents/2015/08/plant-protection-act-2064-2007.pdf

GoN (2010) The plant protection regulation 2010. Government of Nepal (GoN), Kathmandu. www.npqpnepal.gov.np/downloadfile/The_Plant_Protection_Regulation_2010_Eng-Nepali%20_1438587311.pdf

Hellmann JJ, Byers JE, Bierwagen BG, Dukes JS (2008) Five potential consequences of climate change for invasive species. Conserv Biol 22(3):534–543

Holt JS (2004) Principles of weed management in agroecosystems and wildlands. Weed Technol 18:1559–1562

Hulme PE (2009) Trade, transport and trouble: managing invasive species pathways in an era of globalization. J Appl Ecol 46:10–18

Hussner A, Champion PD (2012) *Myriophyllum aquaticum* (Vell.) Verdcourt (Parrot feather). In: Francis RA (ed) A handbook of global freshwater invasive species. Earthscan, London, pp 103–111

Jackson JK (1994) Manual of afforestation in Nepal, vol 2, 2nd edn. Forest Research and Survey Center, Kathmandu

Jayanth KP (1988) Successful biological control of water hyacinth (*Eichhornia crassipes*) by *Neochetina eichhorniae* (Coleoptera: Curculionidae) in Bangalore, India. Trop Pest Manag 34:263–266

Joshi NR, Karki S, Adhikari MD, Udas E, Sherpa SL, Karky BS, Chettri N, Kotru R, Ning W (2015) Development of allometric equations for *Paulownia tomentosa* (Thunb.) to estimate biomass and carbon stocks: an assessment from the ICIMOD Knowledge Park, Godavari, Nepal. Working Paper 2015/10. International Center for Integrated Mountain Development (ICIMOD), Kathmandu

Julien M, McFadyen R, Cullen J (eds) (2012) Biological control of weeds in Australia. Commonwealth Scientific and Industrial Research Organization (CSIRO) Publishing, Melbourne

Karki D (2009) Ecological and socioeconomic impact of *Parthenium hysterophorus* L. invasion in two urban areas in Nepal. MSc thesis, Central Department of Botany, Tribhuvan University, Kathmandu, Nepal

Keller RP, Lodge DM, Finnoff DC (2007) Risk assessment for invasive species produces net bioeconomic benefits. PNAS 104:203–207

Khaniya L (2017) Status of invasive alien plant species in community forests: a case of Sundari and Dhuseri community forests of Nawalparasi district, Central Nepal. MSc thesis, Central Department of Botany, Tribhuvan University, Kathmandu, Nepal

Kumschick S, Richardson DM (2013) Species-based risk assessments for biological invasions: advances and challenges. Divers Distrib 19:1095–1105

Lovenshimer JB, Madritch MD (2017) Plant community effects and genetic diversity of post-fire princess tree (*Paulownia tomentosa*) invasions. Invasive Plant Sci Manag 10:125–135

Lowe S, Browne M, Boudjelas S, DePoorter M (2000) 100 of the world's worst invasive alien species: a selection from the Global Invasive Species Database. The Invasive Species Specialist Group (ISSG), a specialist group of the Species Survival Commission (SSC) of the World Conservation Union (IUCN), New Zealand

Luque GM, Bellard C, Bertelsmeier C, Bonnaud E, Genovesi P, Simberloff D, Courchamp F (2014) The 100th of the world's worst invasive alien species. Biol Invasions 16:981–985

McConnachie AJ, deWit MP, Hill MP, Byrne MJ (2003) Economic evaluation of the successful biological control of *Azolla filiculoides* in South Africa. Biol Control 28:25–32

MFSC (1997) National report on implementation of the convention on biological diversity in Nepal. Ministry of Forest and Soil Conservation (MFSC), Kathmandu

MFSC (2001) Second national report to the convention on biological diversity: Nepal. Ministry of Forest and Soil Conservation (MFSC), Kathmandu

MFSC (2002) Nepal biodiversity strategy. Ministry of Forest and Soil Conservation (MFSC), Kathmandu

MFSC (2006) Nepal third national report to the convention on biological diversity. Ministry of Forest and Soil Conservation (MFSC), Kathmandu

MFSC (2009) Nepal fourth national report to the convention on biological diversity. Ministry of Forest and Soil Conservation (MFSC), Kathmandu

MFSC (2012) National wetland policy 2012. Ministry of Forest and Soil Conservation (MFSC), Kathmandu

MFSC (2014a) Nepal fifth national report to the convention on biological diversity. Ministry of Forest and Soil Conservation (MFSC), Kathmandu

MFSC (2014b) Nepal national biodiversity strategy and action plan 2014–2020. Ministry of Forest and Soil Conservation (MFSC), Kathmandu

MFSC (2014c) Forest policy 2014. Ministry of Forest and Soil Conservation (MFSC), Kathmandu

Murphy ST, Subedi N, Gnawali SR, Lamichhane BR, Upadhyay GP, Kock R, Amin R (2013) Invasive Mikania in Chitwan National Park, Nepal: the threat to the greater one-horned rhinoceros *Rhinoceros unicornis* and factors driving the invasion. Oryx 47:361–368

Neil PE (1990) Early performance of *Paulownia* species. Banko Janakari 2:220–222

NPAFC (2010) Range land policy 2010. National Pasture and Animal Feed Center (NPAFC), Ministry of Agricultural, Land Management and Cooperatives, Kathmandu. http://www.npafc.gov.np/content.php?id=261. Accessed on May 31, 2018

Paini DR, Sheppard AW, Cook DC, De Barro PJ, Worner SP, Thomas MB (2016) Global threat to agriculture from invasive species. PNAS 113:7575–7579

Pimentel D, McNair S, Janecka J, Wightman J, Simmonds C, O'Connell C, Wong E, Russel L, Zern J, Aquino T, Tsomondo T (2001) Economic and environmental threats of alien plant, animal, and microbe invasions. Agric Ecosyst Environ 84:1–20

Poudel S (2016) Diversity and distribution of invasive alien plant species along road network in western Nepal. MSc thesis, Central Department of Botany, Tribhuvan University, Kathmandu, Nepal

Poudel BS, Thapa HB (2012) An assessment of existing studies on invasive alien plant species of Nepal. Banko Janakari 22:28–36

Press JR, Shrestha KK, Sutton DA (2000) Annotated checklist of the flowering plants of Nepal. The Natural History Museum, London

PQPMC (2013) Internal quarantine guidelines 2013. Plant Quarantinne and Pesticide Management Center (PQPMC), Ministry of Agriculture Livestock Development, Hariharbhawan, Lalitpur, Nepal

Pyšek P, Richardson DM, Rejmánek M, Webster GL, Williamson M, Kirschner J (2004) Alien plants in checklists and floras: towards better communication between taxonomists and ecologists. Taxon 53:131–143

Radosevich SR, Holt JS, Ghersa CM (2007) Ecology of weeds and invasive plants, 3rd edn. Wiley, New Jersey

Rai RK, Scarborough H (2015) Understanding the effects of the invasive plants on rural forest-dependent communities. Small-Scale For 14:59–72

Rai RK, Scarborough H, Subedi N, Lamichhane BR (2012) Invasive plants – do they devastate or diversify rural livelihoods? Rural farmers' perception of three invasive plants in Nepal. J Nat Conserv 20:170–176

Rajbhandari KR (2016) History of botanical exploration in Nepal: 1802–2015. In: Jha PK, Siwakoti M, Rajbhandary S (eds) Frontiers of botany. Tribhuvan University, Kathmandu, Nepal, Central Department of Botany, pp 1–99

Rajbhandary KR, Joshi L, Manandhar V, Bhatt GD, Phuyal N, Shrestha J (2012) Two new records of flowering plants (dicot) for Nepal. Bull Dep Plant Resour 34:2–3

Ranjit JD (2013) *Alternanthera philoxeroides*: ecology and management [Leaflet]. Agronomy Division, Nepal Agriculture Research Council (NARC), Lalitpur

Ranjit JD, Gautam DD, Bhattarai RK (2010) Management of *Spergula arvensis* in wheat [Leaflet in Nepali]. Agronomy Division, Nepal Agriculture Research Council (NARC), Lalitpur

REDD Cell (2012) Study on invasive alien species (IAS) as drivers to deforestation and degradation of forests in different physiographic regions of Nepal. Reducing Emission from Deforestation and Forest Degradation (REDD) Cell, Ministry of Forest and Soil Conservation, Kathmandu

Ricciardi A, Blackburn TM, Carlton JT, Dick JTA, Hulme PE, Iacarella JC, Jeschke JM, Liebhold AM, Lockwood JL, MacIsaac HJ, Pyšek P, Richardson DM, Ruiz GM, Simberloff D, Sutherland WJ, Wardle DA, Aldridge DC (2017) Invasion science: a horizon scan of emerging challenges and opportunities. Trends Ecol Evol 32:464–474

Russell JC, Blackburn TM (2017) The rise of invasive species denialism. Trends Ecol Evol 32:3–6

Sankaran KV, Suresh TA (2013) Invasive alien plants in the forests of Asia and the Pacific. Food and Agriculture Organization of the United Nations. Regional Office for Asia and the Pacific, Bangkok

Seebens H, Blackburn TM, Dyer EE, Genovesi P, Hulme PE, Jeschke JM, Pagad S, Pysˇek P, Winter M, Arianoutsou M, Bacher S, Blasius B, Brundu G, Capinha C, Celesti-Grapow L, Dawson W, Dullinger S, Fuentes N, Jager H, Kartesz J, Kenis M, Kreft H, Kuhn I, Lenzner B, Liebhold A, Mosena A, Moser D, Nishino M, Pearman D, Pergl J, Rabitsch W, Rojas-Sandoval J, Roques A, Rorke S, Rossinelli S, Roy HE, Scalera R, Schindler S, Stajerova K, Tokarska-Guzik B, vanKleunen M, Walker K, Weigelt P, Yamanaka T, Essl F (2017) No saturation in the accumulation of alien species worldwide. Nat Commun 8:14435. https://doi.org/10.1038/ncomms14435

Seed Quality Control Center (2013) National seed vision 2013–2025 (Seed sector development strategy). Seed Quality Control Center, National Seed Board, Ministry of Agriculture Development, Hariharbhawan, Lalitpur

Shrestha BB (2016) Invasive alien plant species in Nepal. In: Jha PK, Siwakoti M, Rajbhandary SR (eds) Frontiers of botany. Central Department of Botany, Tribhuvan University, Kathmandu, pp 269–284

Shrestha BK, Dangol DR (2014) Impact of *Mikania micrantha* H.B.K. invasion on diversity and abundance of plant species of Chitwan National Park, Nepal. J Instr Sci Technol 19:30–36

Shrestha UB, Shrestha BB, Shrestha S (2010) Biodiversity conservation in community forests of Nepal: rhetoric and reality. Int J Biodiver Conserv 2:98–104

Shrestha BB, Shabbir A, Adkkins SW (2015) *Parthenium hysterophorus* in Nepal: a review of its weed status and possibilities for management. Weed Res 55:132–144

Shrestha BB, Kokh M, Karki JB (2016) Mapping of invasive alien plant species in Tarai Arc Landscape (TAL) and Chitwan- Annapurna Landscape (CHAL), Nepal. Unpublished research report submitted to National Trust for Nature Conservation (NTNC), Kathmandu, Nepal

Shrestha BB, Ranjit JD, Siwakoti M (2017) Status of invasive alien plants in Nepal. In: Joshi, Bahadur KCH, Acharya AK (eds) Proceedings of second national workshop on conservation and utilization of agriculture plant genetic resources in Nepal, Dhulikhel, 22–23 May 2017. National Agriculture Genetic Resources Center (NAGRC), Fruit Development Directorate (FDD), Department of Agriculture (DoA) and Ministry of Agriculture Development (MoAD), Kathmandu, pp 446–452

Shrestha BB, Budha PB, Wong LJ, Pagad S (2018a) Global register of introduced and invasive species – Nepal. Version 2.2. Invasive species specialist group (ISSG). Checklist Dataset. https://doi.org/10.15468/4r0kkr. Accessed via GBIF.org on 2018.03.09

Shrestha BB, Shrestha UB, Sharma KP, Thapa-Parajuli RB, Devkota A, Siwakoti M (2018b) Community perception and prioritization of invasive alien plants in Chitwan-Annapurna Landscape. Nepal J Environ Manag 229:38–47

Shrestha BB, Joshi S, Bisht N, Yi S, Kotru R, Chaudhary RP, Wu N (2018c) Inventory and impact assessment of invasive alien plant species in Kailash Sacred Landscape. Working Paper 2018/3. International Center for Integrated Mountain Development (ICIMOD), Kathmandu

Shrestha UB, Sharma KP, Devkota A, Siwakoti M, Shrestha BB (2018d) Potential impact of climate change on the distribution of six invasive alien plants in Nepal. Ecol Indic 95:99

Singh RM (2013) Biobriquetting in Nepal – scope and potentials: a review. Kathmandu Univ J Sci Eng Technol 9:104–120

Siwakoti M (2012) Threats and opportunity of invasive alien plant species in wetland conservation of Nepal. In: Proceedings of international wetland symposium, Pokhara, 7–9 November, 2012. Ministry of Forest and Soil Conservation/Conservation and sustainable use of wetlands in Nepal, Kathmandu, Nepal, pp 66–72

Siwakoti M, Karki JB (2009) Conservation status of Ramsar sites of Nepal Tarai: an overview. Botanica Orientalis 6:76–84

Siwakoti M, Shrestha BB (2014) An overview of legal instruments to manage invasive alien species in Nepal. In: Thapa GJ, Subedi N, Pandey MR, Thapa SK, Chapagain NR, Rana A (eds) Proceedings of the international conference on invasive alien species management, Chitwan, March 25–27, 2014. National Trust for Nature Conservation, Kathmandu, pp 101–111

Siwakoti M, Shrestha BB, Devkota A, Shrestha UB, Thapaparajuli RB, Sharma KP (2016) Assessment of the effects of climate change on the distribution of invasive alien plant species in Nepal. In: Bhuju DR, McLaughlin K, Sijapati J, Devkota BD, Shrestha N, Ghimire GP, Neupane PK (eds) Building knowledge for climate resilience in Nepal: research brief. Nepal Academy of Science and Technology, Kathmandu, pp 5–8

Sushilkumar (2015) History, progress and prospects of classical biological control in India. Indian J Weed Sci 47:306–320

Thapa A (2016) Status and management of invasive alien plant species in Beeshazar Lake, Chitwan, Nepal. MSc thesis, Department of Environmental Science, Goldengate International College, Tribhuvan University, Kathmandu, Nepal

Thapa LB, Kaewchumnong K, Sinkkonen A, Sridith K (2016) Impacts of invasive *Chromolaena odorata* on species richness, composition and seedling recruitment of *Shorea robusta* in a tropical Sal forest. Nepal Songklanakarin J Sci Technol 38:683–689

Thapa LB, Kaewchumnong K, Sinkkonen A, Sridith K (2017) Plant invasiveness and target plant density: high densities of native *Schima wallichii* seedlings reduce negative effects of invasive *Ageratina adenophora*. Weed Res 57:72–80

Thapa S, Chitale V, Joshi-Rijal S, Bisht N, Shrestha BB (2018) Understanding the dynamics in distribution of invasive alien plant species under predicted climate change in Western Himalaya. PLoS One 13(4):e0195752. https://doi.org/10.1371/journal.pone.0195752

Timsina B, Shrestha BB, Rokaya MB, Munzbergova Z (2011) Impact of *Parthenium hysterophorus* L. invasion on plant species composition and soil properties of grassland communities in Nepal. Flora 206:233–240

Tiwari S, Adhikari B, Siwakoti M, Subedi K (2005) An inventory and assessment of invasive alien plant species of Nepal. IUCN Nepal, Kathmandu

Vila M, Hulme PE (eds) (2017) Impact of biological invasion on ecosystem services, Invading Nature – Springer Series in invasion ecology, vol 12. Springer International Publishing, Cham

Villamagna AM, Murphy BR (2010) Ecological and socio-economic impacts of invasive water hyacinth (*Eichhornia crassipes*): a review. Freshw Biol 55:282–298

Vitousek PM, D'Antonio CM, Loope LL, Rejmanek M, Westbrooks R (1997) Introduced species: a significant component of human caused global change. N Z J Ecol 21:1–16

Winston RL, Schwarzländer M, Hinz HL, Day MD, Cock MJW, Julien MH (eds) (2014) Biological control of weeds: a world catalogue of agents and their target weeds, 5th edn. USDA Forest Service, Forest Health Technology Enterprise Team, Morgantown

WWF Nepal (2013) Communities come together to protect wetland. Hario Ban Program, WWF Nepal. URL: http://www.wwfnepal.org/?207351/Communitiescome-together-to-protect-wetland. Assessed 31 July 2017

Yan SH, Song W, Guo JY (2017) Advances in management and utilization of invasive water hyacinth (*Eichhornia crassipes*) in aquatic ecosystems–a review. Crit Rev Biotechnol 37:218–228

Invasive Species and Their Impact on Tropical Forests of Central India: A Review

5

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Abstract

Tropical forests are the richest biodiversity hotspots and are under immense natural and anthropogenic pressures that lead to biodiversity loss. One such cause is alien plant invasion that alters the native forest stand structure and composition and disrupts the vital ecosystem functions. Central India, which mainly spans across the three states, viz. Madhya Pradesh, Chhattisgarh and some parts of Maharashtra, is well-known for its sprawling tropical deciduous forests, which are also no less immune to the present-day pressures, including the plant invasion. Alien invasive plants arrive via several pathways and possess unique traits that help them to surpass the barriers in the new habitats, where many influential

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factors might operate upon them. Once established, they may profoundly impact the invaded ecosystem. Most of the studies from Central India have been focused on floristics, forest structure, impact of disturbances, etc., and relatively few studies have addressed plant invasion. Overall, there are 179 invasive taxa in Central India, mostly from the Asteraceae (17.3%) and Fabaceae (14.5%) families. Majority of them are from Tropical America (52%), and most are herbs (69%). An outline of the most common top ten Central Indian invaders has been presented. Climate change might influence invasive plants, and constant monitoring and modelling is required to understand invasive species dynamics for effective management. Invasive alien species are to be tended with extreme caution and smart and novel approaches of putting them to use might help in better management for controlling them. This review will also provide a conceptual basis for improving our general understanding on invasive species and their impact on tropical forest ecosystems.

Keywords

Alien species · Plant invasion · Introduction pathway · Native habitats · Climate change · Management

5.1 Introduction

Tropical forests harbour diverse habitats and are rich in biodiversity (Whitmore 1990). Amid the different types of tropical forests, tropical dry forest, which once sprawled across more than half of the tropical forests (Janzen 1988a; Murphy and Lugo 1986; Raghubanshi and Tripathi 2009), is considered to be one of the most endangered forest types in the world (Janzen 1988b; Lerdau et al. 1991; Mittermeier et al. 1999; Raghubanshi and Tripathi 2009). According to UNEP-WCMC Forest Programme (2011), the tropical dry forest is just 17% of present-day tropical forests, and it serves a wide range of ecosystem functions such as climate regulation, watershed protection and management, prevention of soil erosion and provision of timber and other non-timber forest products. When forest structure gets altered due to natural or anthropogenic reasons, the ecosystem functions they render also change (Agarwala et al. 2016a). However, in the recent decades, these richly endowed tropical dry forests have experienced an accelerating loss of biodiversity. Among many factors that trigger biodiversity loss, such as climate change, habitat fragmentation, habitat destruction and over-exploitation (Millennium Ecosystem Assessment 2005; Morris 2010), alien plant invasion is also a significant factor.

A characteristic hallmark of the Anthropocene epoch (Steffen et al. 2011; Lewis and Maslin 2015) is species crossing biogeographical barriers by anthropogenic introductions of native species into new regions, where some of them colonize and naturalize (van Kleunen et al. 2015) by establishing self-sustaining populations (Blackburn et al. 2011; Dawson et al. 2017) and producing adverse effects on native biota. The demographic change in an area from having a few sporadic colonists to rampant and abundant invaders is witnessed as a dramatic ecological event (Ellstrand and Schierenbeck 2000). Biological invasions are now a worldwide phenomenon (Seebens et al. 2017a), deemed to be one of the major causes of biodiversity loss (Courchamp et al. 2017). Although it is a global phenomenon, megadiverse countries that are developing (like India) are impacted to a great extent (Frehse et al. 2016). However, introduced species are also known to stabilize degraded forest ecosystems, and they co-exist alongside the native species, forming 'novel forests' (Martinez 2009, 2010; Martinez and Lugo 2008; Martinez et al. 2010; Lugo 2013). It is even predicted that these novel forests could perform ecosystem functions similar to native forests (Lugo 2013). It has been estimated that around 10% of the vascular flora in the world could potentially invade and impact other ecosystems directly or indirectly (Singh et al. 2006; Shukla and Sinha 2012). Given this backdrop, it is now crucial to understand the traits and impacts of invasive species for effective management of the currently dwindling forest cover.

5.2 A Brief Profile of Central India

Tropical forests constitute approximately 86% of the total forest cover of India, and of this, tropical dry forest occupies 46%, which is thereby a major biome of the country (Singh and Singh 1988; Pande 2001; Sagar and Singh 2005). It is an integral vegetation type of Central India. Central India, with its diverse tropical deciduous forests and luxuriant natural vegetation (Kamble et al. 2013), is considered as one of the biggest and most important phytogeographical provinces of the country (Chauhan et al. 2013). Politically, Central India is comprised of the states Madhya Pradesh, Chhattisgarh and parts of Maharashtra, Rajasthan, Uttar Pradesh and Jharkhand. Madhya Pradesh is considered the 'forest heartland of India' (Thakur 2015a) and together with Chhattisgarh comprises the major portion of Central India (Fig. 5.1). The 2 states that lie between 17°48'–26° 52' N and 74° 2'–84° 24' E possess 13 agroclimatic zones and are rich in biodiversity (Chaubey et al. 2015). In terms of forest cover by state, Madhya Pradesh (77,462 km^2) ranks first, and Chhattisgarh (55,586 km^2) ranks third in the country (FSI 2015). Sal (*Shorea robusta* Gaertn.) and teak (*Tectona grandis* L.f.) forests are prominent in both the states.

Madhya Pradesh is the second largest state in India with tropical and subtropical climate (MoEF 2014). The forests of Madhya Pradesh are classified into three categories, reserve forest, protected forest and unclassified forest, that occupy 65.36%, 32.84% and 1.7% of the forest area, respectively (Madhya Pradesh Forest Department http://www.mpforest.org/forest.html). According to Champion and Seth (1968), this state has 3 major forest type groups, tropical moist deciduous, tropical dry deciduous and tropical thorn forests, which together comprise 18 forest types (FSI 2011). Some densely forested districts of Madhya Pradesh are Balaghat, Dindori, Chhindwara and Betul (Government of Madhya Pradesh http://www.mp.gov.in/en/web/guest/forest). Chhattisgarh is the tenth largest state in India with a dry subhumid climate (MoEF 2014). The forests of Chhattisgarh are also classified as reserve forest, protected forest and unclassified forests occupying 43.13%,

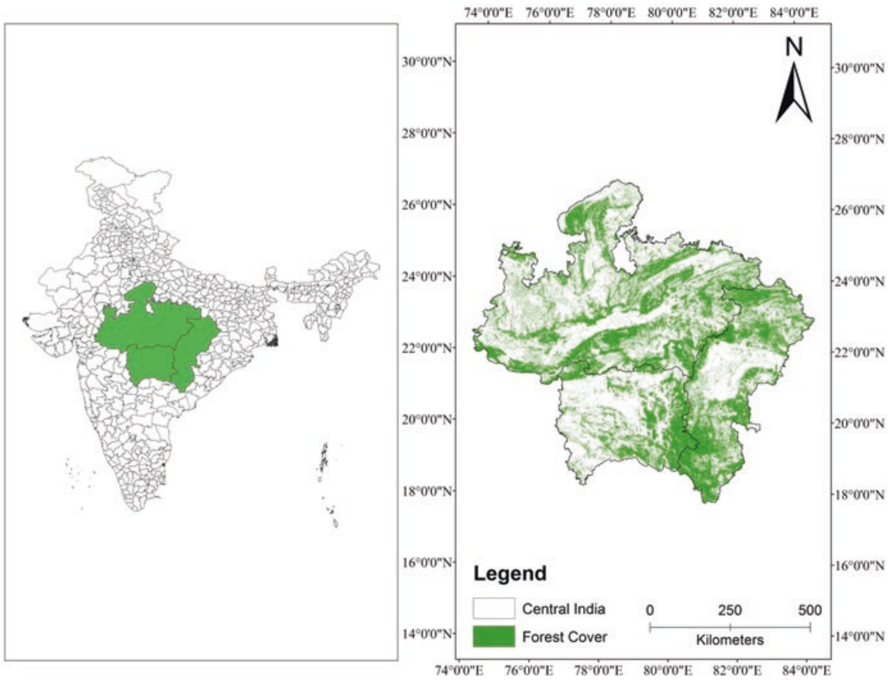

Fig. 5.1 Left: Central India, the location covered by this review. Right: Forest cover of different forest types in Central India

40.21% and 16.65% of the forest area, respectively (Chhattisgarh Forest Department, http://www.cgforest.com/English/Introduction.htm). According to Champion and Seth (1968), this state has two major forest 2 groups, tropical moist deciduous and tropical dry deciduous forests, which together comprise 10 forest types (FSI 2011). Some densely forested districts of Chhattisgarh are Bastar, Dantewada, Bilaspur and Sarguja (Chhattisgarh Forest Department, http://www.cgforest.com/English/Introduction.htm).

5.3 Plant Invasion

Alien species are important components that bring about a global environmental change as stated by Vitousek et al. (1996) and Gooden et al. (2009). Alien species may be invasive or non-invasive, and as such they are related to biodiversity in two ways: they may contribute to regional diversity increasing species richness, but if they are invasive, they may threaten the existence of the native biota (Jauni and Hyvonen 2010). Not all species that arrive are successful invaders. Williamson's (1996) 'tens rule' demonstrates that one in ten of imported species become introduced, one in ten of those introduced species become established and one in ten of those established species become a pest (Williamson and Fitter 1996; Ehler 1998).

As reported by Richardson and Rejmanek (2011), although woody plant invasions are on the rise globally, only 0.5–0.7% of the world's shrub and tree species are actually invasive beyond their native range. Invasive species may either act as 'drivers' or 'passengers' causing environmental change (Didham et al. 2005; MacDougall and Turkington 2005; Powell et al. 2011). If they are drivers, they dominate the native diversity with their unique traits and mechanisms, but if they are passengers, they dominate the region as a consequence of anthropogenic causes like disturbances or habitat degradation. Either way, they affect the native biodiversity (Byers 2002; Powell et al. 2011). They are sometimes perceived as good indicators of land use change or disturbance in a region (Maskell et al. 2006; Jauni and Hyvonen 2010). Invasion hotspots are mostly characterized by vegetation with less tree cover (Padalia and Bahuguna 2017).

Gray (2017) describes different processes in a biological invasion: uptake in the native range is followed by transport and introduction. Once an invasive species is introduced, it colonizes, establishes and spreads itself (Colautti et al. 2006; Theoharides and Dukes 2007; Hellmann et al. 2008). A region undergoes three stages during a biological invasion as summarized by Aung and Koike (2015) – a prior to establishment stage in which there is no regional population, an invading stage during which the number of invaded patches in a region increase and a steady or a saturated stage in which the invasive species had occupied all the suitable patches in the region. The dynamics of invaders such as their population structure and biotic and abiotic interactions vary with time (Strayer et al. 2006), thus producing unprecedented effects in a region. Many plant invaders remain benign for a long time before suddenly booming, which could be due to the changes in the biotic or abiotic interactions or selection within the invasive species itself (Crooks 2005; Ramaswami and Sukumar 2013). The three main foci that need to be paid heed to comprehend and successfully contain any biological invasion are the mode of introduction, traits of the invasive species and factors favouring successful ecesis.

5.3.1 Introduction Pathways

It is both interesting and important to understand the pathways of introduction of alien species along the time scale across different landscapes. Evidence on the different modes of introduction is relevant for prevention, monitoring and risk assessments (Clout and Williams 2009, Simberloff and Rejmanek 2011; Essl et al. 2015; Saul et al. 2017). African and Asian colonization from the fifteenth to nineteenth centuries by European nations led to increased introductions of many alien species for various purposes (Pysek and Richardson 2008; Khuroo et al. 2012), and the native landscape of India was greatly altered due to alien species invasions resulting from those colonial invasions (Kannan et al. 2013). Nevertheless, Seebens et al. (2017b) reported that 37% of the first-hand accounts on the established alien species were more recent (1970–2014) and forecasted that the numbers of new species would probably increase with time.

Introduction refers to a process whereby the plant or a propagule gets transported by humans beyond a geographical barrier, while naturalization refers to the surpassing of biotic and abiotic barriers of survival and other barriers of regular reproduction (Richardson et al. 2000). The invasion begins with introduction (Blackburn et al. 2011), and the pathways could be intentional or unintentional (Essl et al. 2015). Essl et al. (2015) recognized six pathways of introduction, which were originally proposed by Hulme et al. (2008) and later adopted by Convention on Biological Diversity (2014) after modification – release (intentional introduction in a natural environment), escape (escape of an alien species from confinement (zoos, botanical gardens, etc.), contaminant (unintentional transport of species that were present as contaminants from intentionally transported commodities), stowaway (transfer of organisms that were attached to the transporting vessels like ships, boats, etc.), corridor (movement of organisms to a new region due to the construction of a transport structure like opening up of international canals, linking rivers, constructing tunnels to connect mountain valleys, etc.) and unaided (secondary dispersal of a species across borders that was originally introduced by any of the above pathways). The number of individuals or propagules introduced via a particular pathway is directly related to the impact of the pathway (Wilson et al. 2009) as a pathway introducing greater numbers and higher diversity of alien species could result in the establishment of several alien species in the new region than another pathway that carried fewer individuals and less diversity of species (Lockwood et al. 2009; Essl et al. 2015; Pergl et al. 2017). Moreover, taxa introduced via multiple pathways in diverse habitats and different regions are more likely to be naturalized than taxa arriving by just a single pathway (Küster et al. 2008; Pergl et al. 2017).

Colonization and trade relations of India with Britain, Spain, France, Portugal, Middle East and Central Asian nations paved way to the introduction of several alien plant species (Pandey 2000; Khuroo et al. 2012). Some of the reasons highlighted by Richardson (1998) for the intentional introduction of alien species are as follows: alien trees have faster growth rates and are easier to manage silviculturally than native ones; they are thought to be more suitable for afforestation in grasslands, scrublands and degraded forests; they are more resistant to insect attacks and diseases; and their cultivation is profitable. Most of the species that threaten Indian ecosystems were mainly introduced to serve horticultural purposes (Reddy 2012) and also to meet the demand for fuelwood, to prevent desertification, and also for commercial benefits (Hiremath and Sundaram 2013). Trade is considered as a major factor for naturalization (Seebens et al. 2015; Kim and Kil 2016), and with increasing globalization, it is likely that many species would be introduced and would be naturalized outside their native ranges (van Kleunen et al. 2015; Dawson et al. 2017). As expressed by Banerjee and Dewanji (2017a), it is indeed a matter of concern that many native ecosystems now support several alien species (Johnson et al. 2009; Kuebbing et al. 2013) as a consequence of these increased introductions (and also subsequent invasions), due to which interspecific relationships and community interactions might become more complicated.

5.3.2 Traits of Invasive Species

To predict the fate of introductions is often very difficult as many of those introduced alien species interact with and integrate into the native communities (Buckley 2017). Furthermore, in order to be invasive, the introduced plants need to spread their reproductive offspring or propagules away from their introduced sites (Richardson et al. 2000), and this is facilitated by their unique traits. These biological traits enable them to be both good colonizers and good persisters (D'Antonio and Meyerson 2002; Hiremath and Sundaram 2005). Identification of traits pertaining to invasiveness helps in predicting the likelihood of a species becoming invasive in the future (Catford et al. 2016). Invader traits play a very crucial role in determining the large-scale patterns of bioinvasions (Iannone et al. 2016). Very often, the very same traits in alien plants which make them attractive for the purpose of their introduction also contribute to their invasiveness (Dodet and Collet 2012; Sitzia et al. 2016). For example, plants that pose a longer display of bright and showy flowers and fleshy fruits are selected for horticultural introductions, but those characteristics also attract generalist seed-dispersing agents (Reichard 2011). Similarly, woody species used in agroforestry are selected for their fast growth rates, prolific propagule production and tolerance to a broad spectrum of environmental conditions (Richardson and Rejmanek 2011), but these traits also increase potential invasiveness.

Some of the unique traits of invasive species are as follows: broad native ranges, being generalists in distribution, extensive and aggressive root systems, allelopathy, shorter regeneration periods, longer flowering and fruiting periods, being attractive to humans and pollinators, copious seed production that could be easily dispersed, ability to easily adapt and tolerance to different stresses in the introduced habitats (Rejmanek and Richardson 1996; Pyšek and Richardson 2007, 2008; van Kleunen et al. 2010; Sol et al. 2012; Iannone et al. 2016). Many alien plants produce lightweight seeds that are easily wind-dispersed (Egawa 2017), fast growing and more competitive in resource capture and use than native flora (Nagel et al. 2004; Burns 2006; Feng et al. 2008; Shen et al. 2011; Kumari and Choudhary 2016). Alien plants that are short-lived are the most successful invaders in human-transformed landscapes (Golivets 2014). Grasses and forbs get unintentionally introduced in many habitats as they profusely produce seeds that have dispersal aids such as hairs, awns and spines that stick to animals and clothes (Groves et al. 2005; Murray and Phillips 2012). Although the above-listed traits aid in easy dispersal of many invasive species, not all the invaders conform to this norm, as, for instance, even heavy-seeded tree species could become invasive (e.g. *Mangifera indica* in the neotropics).

These apart, as stated by Blackburn et al. (2011), Richardson and Pyšek (2012) and Zimmermann et al. (2017), traits related to reproduction and dispersal are essential and directly linked with the ability of the alien species to become naturalized or invasive. Alien plants that are capable of selfing have more chances of naturalization, and high selfing ability is directly related to the number of naturalized regions (Razanajatovo et al. 2016). Baker (1955) hypothesized that species capable of uniparental reproduction are more likely to establish after long-distance dispersal than those species that rely on suitable mates and pollinators, and this is known as Baker's Law or Baker's Rule. However, even though many naturalized plants have been shown to possess high selfing ability, some of them suffer inbreeding depression (Rodger and Johnson 2013; Razanajatovo et al. 2016). Another way of autonomous seed production is via apomixis, viz. the process of asexual, clonal production of seeds, which aids in avoiding meiosis and fertilization (Mazzolari et al. 2017). Many entomophilous invasive plants that usually possess showy flowers and are rich in floral resources (Albrecht et al. 2014; Albrecht et al. 2016) are known to form ecological interactions with native pollinators, and these associations are critical for their successful reproduction (Stout and Tiedeken 2017). If alien plants attract frugivores with fruit production that is brightly coloured and fleshy and has increased energy content and easy accessibility, then they have more chances of outcompeting the native flora for dispersal agents (Aslan and Rejmanek 2012).

Invaders are frequently known to possess preadaptations to climate or habitat matching (Murray and Phillips 2012), and they may also undergo local adaptations to match with native species (Oduor et al. 2016). They may also undergo changes in their genetic makeup as a result of selection pressures (Sakai et al. 2001; Kohli et al. 2006). As demonstrated by Blackburn et al. (2009) and Essl et al. (2015), the traits of invaders might have changed with time and also across pathways. Comparing and contrasting the characteristics between invasive and native species might not present a conclusive picture of traits of the former as the latter might also have the tendency to become invasive if introduced elsewhere (Feng et al. 2008). However, a database on plant traits consisting of about 5.6 million trait records of 100,000 plant species (as on July 2017) around the globe is being maintained by Max Planck Institute for Biogeochemistry and Future Earth in the TRY Plant Trait Database (https://www.try-db.org/TryWeb/Home.php).

5.3.3 Influential Factors and Successful Ecesis

Biological invasions are very dynamic, and most of them are due to human interventions, a phenomenon known as 'invasion debt' (Essl et al. 2011; Ööpik et al. 2013). Establishment or a successful ecesis refers to the existence of a persistent and self-perpetuating invasive population after an alien introduction (Gray 2017). Ramaswami and Sukumar (2013) demonstrated that the successful establishment of an invasive species depends upon the properties of both the invading species and the invaded environment. Invasibility and invasiveness are two strongly interwoven concepts (Hui et al. 2016). According to Colautti et al. (2014), the success of an invasive species depends on two factors – intrinsic factors by which some species are inherently good invaders and extrinsic factors by which a species becomes invasive due to ecological and evolutionary interactions.

In order to successfully establish in a new habitat, the non-native species must surpass barriers to both survival and reproduction as failing to do so might result in non-establishment or eventual extinction of the established population (Lockwood et al. 2013). The successful invaders must also overcome biotic resistance posed by the native, resident community (Levine et al. 2004) including soil microbes (Chen et al. 2017). The allelopathic effect of native plant communities on the introduced invaders as a means of biotic resistance, although seldom studied (Ning et al. 2016), is an important factor. Seebens et al. (2013) and Seebens et al. (2017a) observed that a number of alien species follows a 'hump-shaped' pattern with respect to geographic distance between the donor and the recipient regions. Climatic variables also influence the invasion of a species on the regional and continental scales (Csecserits et al. 2016).

Many hypotheses have been proposed regarding the invasion success such as 'enemy release hypothesis' which attributes the invasion success to the liberation of alien species from their native herbivores and pathogens (Darwin 1859; Williams 1954; Elton 1958; Hierro et al. 2005); 'empty niche hypothesis' that relates invasion success of alien species with the possibility of gaining access to resources in the introduced habitats that are not utilized by the native species (Elton 1958; MacArthur 1970; Levine and D'Antonio 1999; Mack et al. 2000; Hierro et al. 2005); 'resource availability hypothesis' which suggests that the invasion success of alien plants are brought about by the availability of resources such as water, light, soil nutrients, etc. and the resources might become available either when the supply of resources increases or when the capture of resources by other local plants decreases (Keane and Crawley 2002; Blumenthal 2005); 'species richness hypothesis' which speculates that species-rich communities are more resistant to invasion than species-poor communities (Elton 1958; Hierro et al. 2005); 'limiting similarity hypothesis' which proposes that invasive alien plants are unlikely to establish in communities where native species possess similar traits (Funk et al. 2008; Yannelli et al. 2017); 'novel weapons hypothesis' which posits that invasion success might be due to novel ways of interaction of alien species with native biota and this hypothesis highlights the role of allelopathy (Callaway and Aschehoug 2000; Bais et al. 2003; Hierro et al. 2005); and 'disturbance hypothesis' which advances the fact that disturbed ecosystems are easier targets to plant invasion than those that are less disturbed (Myers 1983; Hobbs 1989; Larson et al. 2001; Hiremath and Sundaram 2005).

Apart from these, there is also an on-going phenomenon called 'invasional meltdown', also called secondary invasion, whereby the successful ecesis of one invasive species facilitates the successful establishment of the successive invaders (Simberloff and Von Holle 1999; Catford et al. 2009; Bongard 2012; Jeschke et al. 2012; Kuebbing and Nunez 2016). Thus, many factors independently or synergistically play a vital role in determining the destiny of the newly introduced and subsequently invading species.

5.4 Invasive Species: Why Are They a Concern?

Invasive species can both promote and disrupt ecosystem functioning (Simberloff et al. 2013; Shackleton et al. 2016; Vaz et al. 2017), and their role is largely dependent on the spatiotemporal context, based on people's perceptions (Kueffer and Kull

2017; Vaz et al. 2017). As such the invasive species warrant attention, based on their interactions within new ecosystems. These interactions directly or indirectly impact the invaded ecosystems (Stout and Tiedeken 2017). Only 5–20% of all the alien species pose problems (Vila et al. 2010; Lockwood et al. 2013), but their impacts on ecosystem structure and functions are persistent and are on a large scale (McGeoch et al. 2016).

Most often, the negative impacts of invasive species are widely discussed as they are more apparent than the positive effects, although some researchers have worked on the latter as well. For example, Sandilyan and Klooster (2016) highlighted the medicinal values of invasive alien flora. Often, plant invaders support the daily livelihoods of local people (Kull et al. 2011; Vaz et al. 2017), and they also contribute positively to ecosystem function by attracting pollinators and dispersal agents that promote local/regional biodiversity and offer soil and coastal sediment protection (Vaz et al. 2017), besides providing other ecosystem services than the original purpose for which they were introduced like horticultural purposes, fuelwood production, etc. Introduced alien species could also help in restoring degraded forest lands (Lugo 2013) and enhance carbon sequestration (Dickie et al. 2014; Vaz et al. 2017). In India, some of the alien species are known for their current and potential benefits: soil management, ethnomedicine, insecticide preparation (e.g. *Lantana camara*; Rajashekar et al. 2014; Chatterjee 2015), preventing desertification and providing fuelwood (e.g. *Prosopis juliflora*; Hiremath and Sundaram 2013) and water effluent treatment (e.g. *Eichhornia crassipes*; Priya and Selvan 2017). Therefore, the pros or cons that an alien invader might have are directly related to its role in the new environment, site conditions (or requirements) and, also, the extent to which the plant has been studied and put to use.

As argued by Padalia and Bahuguna (2017), there exists a positive relationship between the dominance of non-native invaders and decline of native plants (Didham et al. 2005; Hulme 2008) in Central India. Invasive plant dominance on disturbed or open forests affects the recovery of those forests (Ghazoul and Sheil 2010; Morris 2010). Bioinvasions often result in altered community structure, and invasive plants may affect the faunal composition as well, especially those that are specialists by causing complete exclusion of their food plants from the invaded region (Chapin et al. 2000; Louman et al. 2010; Kumari and Choudhary 2016). Alien invasive plants also affect aboveground plant-pollinator mutualisms (Traveset and Richardson 2006; Sundaram and Hiremath 2012; Bruckman and Campbell 2016) and belowground root-mycorrhizal mutualisms of native species (Stinson et al. 2006; Sundaram and Hiremath 2012), disrupt water and nitrogen cycles (Gordon 1998; Ashton et al. 2005; Strayer et al. 2006; Sundaram and Hiremath 2012), cause skin irritations and health problems in humans due to the presence of allergens, convert non-fire-prone habitats to fire-prone (D'Antonio and Vitousek 1992; Vaz et al. 2017) and change soil properties by exuding allelochemicals and competing for nutrients. When forest structure gets transformed due to invasive tree dominance, above- and belowground carbon pool sizes also change, and the overall capability of carbon sequestration of the forest is affected (Peh et al. 2015), although it might be positive or negative, depending on the species. Apart from the fact that primary invasion might facilitate secondary invasion, the control or removal of a single primary invader might lead to the release of many other subordinate invaders (Penk et al. 2016) which might have remained subdued otherwise. Therefore, invasive species pose a great concern from conservational, ecological and economic perspectives.

5.5 Studies from Central India

Due to ever-increasing anthropogenic pressures, the tropical dry deciduous forest in many parts of Central India is being converted into dry deciduous scrub or dry grasslands or savannahs (Sagar and Singh 2005). Moreover, the recurrences of forest fires have degraded the forests, making it more susceptible to invasion. In the forests of Mendha village of Maharashtra, the fire return intervals are very short as they are burned annually to favour the collection of *Diospyros melanoxylon* leaves and *Madhuca longifolia* flowers (Saha 2002; Hiremath and Sundaram 2005). Irrespective of being perturbed and degraded by various pressures, the dry forests are believed to be more resilient with a greater potential to recover and reach maturity than the wet forests (Ewel 1977; Murphy and Lugo 1986; Sagar and Singh 2005). In this critical context, a few studies have attempted to unravel the current status of the Central Indian forests and to predict their future fate, so as to prioritize sites for conservation and management.

Quamar and Chauhan (2011) and Chauhan et al. (2013) reconstructed the vegetation history and changes in climate of some regions in Madhya Pradesh of the Holocene epoch based on palynological, geochemical and other evidence. Quamar and Chauhan (2011) found that the climate turned less humid since 1350 AD, due to low precipitation which is reflected by a decline in *Acacia*-scrub forest in the Sehore district of Madhya Pradesh. Chauhan et al. (2013) observed the present-day sal-dominated deciduous forests established since 2800 years BP, based on their studies from a deep sediment core of Padauna Swamp, Madhya Pradesh. Agarwala et al. (2016a) examined the factors that influence the species composition in the tropical dry forests around Kanha Tiger Reserve in Central India based on size-class proportions and observed that fire and grazing affect the long-term species composition in these forests. They also documented the changes these tropical dry forests underwent due to human-mediated disturbances (Agarwala et al. 2016b) and concluded that human use alters the relative abundance of a species, which may affect the forest composition in the long term. Joshi et al. (2009) assessed the impact of industrialization on the land use and land cover change of Chhattisgarh by remote sensing and recorded that around 57% of the forest area was affected. Salunkhe et al. (2016) estimated the tree biomass in seven districts of Madhya Pradesh and concluded that the dry deciduous forests contain more biomass than the mixed deciduous forests.

Pande (2001, 2005) studied the forest structure, biomass and productivity of three teak forest sites in Satpura Plateau and noted that disturbance adversely affects regeneration, while planting target species in forest gaps created by disturbances enhances productivity. Chaubey et al. (2015) studied the floral diversity and population structure of different forest types in the two Central Indian states, Madhya Pradesh and Chhattisgarh, and observed that the population structure of most of the forest types showed a progressive trend. Tree diversity and population structural inventory on tropical dry deciduous forests of Barnawapara Wildlife Sanctuary were undertaken by Lal et al. (2015), who demonstrated that the natural forests hold more species diversity than a plantation forest and hence the conservation measures should be more directed towards natural forests.

Although many studies have been reported on floristics and forest structure (Kushwah and Kumar 2002, Sagar and Singh 2005, Sharma and Raghubanshi 2006, Thakur and Khare 2006, Raghubanshi and Tripathi 2009, Patel 2012, Sahu et al. 2008, Sahu et al. 2012, Mujaffar et al. 2013, Sinha and Sinha 2013, Ray and Sainkhediya 2014, Sahu and Gupta 2014, Singh and Upadhyay 2014, Sandya and Ahirwar 2015, Thakur 2015a,b, and Vinodia and Dixit 2017) which are both highly and timely needed, very few studies have been carried out on plant invasion in Central India. Reshi et al. (2017) has documented 143 alien species comprising of 43 families from tropical dry deciduous forests of Sagar district, Madhya Pradesh. Suman et al. (2017) has enumerated 75 alien angiospermic plants from Panna Tiger Reserve, Madhya Pradesh. Shukla et al. (2009) documented 106 alien, invasive species from Achanakmar-Amarkantak Biosphere Reserve that spans across both the states, while Sahu and Singh (2008) assessed the effect of *Lantana camara* invasion on the structural attributes of the forest of the same reserve. Shukla and Sinha (2012) inventoried the alien flora of Chhattisgarh wherein they recorded 162 species. Wagh and Jain (2015) enumerated 102 alien invasive species from Jhabua district, Madhya Pradesh. Sharma et al. (2009) examined the effects of *Hyptis suaveolens* invasion in the Vindhyan range. Soni et al. (2014) assessed the effects of *Lantana camara* on the diversity and regeneration potential of a tropical dry forest in Ratanpur forest of Chhattisgarh, while Kumari et al. (2014) studied the effects of *Parthenium hysterophorus* on the agricultural fields of Bilaspur, also from Chhattisgarh. Dhileepan and Senaratne (2009) prepared a distribution map of *Parthenium hysterophorus* and its biological control agent, viz. *Zygogramma bicolorata*, in South Asia which also includes Central Indian states. Padalia and Bahuguna (2017) identified ‘hotspots of invasion’ in Central India to prioritize zones for conservation. Thus, on a comparative note, we find that more studies on alien plant invasion come from Chhattisgarh than Madhya Pradesh.

5.6 Diversity of Alien Invasive Taxa of Central India

5.6.1 Family-Wise Distribution of Invasive Taxa

In toto, there are 179 species belonging to 128 genera and 46 families that are invasive in Central India (Table 5.1). Asteraceae and Fabaceae are the largest contributors to the alien invasive flora of Central India (Fig. 5.2; Reshi et al. 2017; Suman et al. 2017; Rao and Murugan 2006, Khuroo et al. 2007, Singh et al. 2010, Khuroo et al. 2012, Reddy 2012, Deshmukh et al. 2015, Srivastava et al. 2014). Global

Table 5.1 List of invasive alien plant species in Central India with details of their nativity, life form and mode of introduction from various published sources

S.No.	Species	Family	Origin	Life form	Introduction
1	*Acacia farnesiana* (L.) Willd.	Fabaceae	SAM	Tree	Unintentional
2	*Acanthospermum hispidum* DC.	Asteraceae	BR	Herb	Unintentional
3	*Adenostemma lavenia* (L.) Kuntze	Asteraceae	SAM	Herb	Unintentional
4	*Ageratum conyzoides* L.	Asteraceae	TAM	Herb	Unintentional
5	*Alcea rosea* L.	Malvaceae	MR	Herb	Unintentional
6	*Alternanthera paronychioides* A.St. Hil.	Amaranthaceae	TAM	Herb	Unintentional
7	*Alternanthera philoxeroides* (Mart.) Griseb.	Amaranthaceae	TAM	Herb	Unintentional
8	*Alternanthera pungens* Kunth	Amaranthaceae	TAM	Herb	Unintentional
9	*Alternanthera sessilis* (L.) R.Br. ex DC.	Amaranthaceae	TAM	Herb	Unintentional
10	*Amaranthus spinosus* L.	Amaranthaceae	TAM	Herb	Unintentional
11	*Anagallis arvensis* L.	Primulaceae	EU	Herb	Unintentional
12	*Annona reticulata* L.	Annonaceae	TAM	Tree	Unintentional
13	*Antigonon leptopus* Hook. & Arn.	Polygonaceae	TAM	Climber	Unintentional
14	*Argemone mexicana* L.	Papaveraceae	SAM	Herb	Unintentional
15	*Aristida adscensionis* L.	Poaceae	NAM	Grass	Unintentional
16	*Asphodelus tenuifolius* Cav.	Liliaceae	TAM	Herb	Unintentional
17	*Bidens pilosa* L.	Asteraceae	TAM	Herb	Unintentional
18	*Blainvillea acmella* (L.) Philipson	Asteraceae	TAM	Herb	Unintentional
19	*Blumea eriantha* DC.	Asteraceae	TAM	Herb	Unintentional
20	*Blumea lacera* (Burm.f.) DC.	Asteraceae	TAM	Herb	Unintentional
21	*Blumea obliqua* (L.) Druce	Asteraceae	TAM	Herb	Unintentional
22	*Borassus flabellifer* L.	Arecaceae	TAF	Tree	Unintentional
23	*Bryophyllum pinnatum* (Lam.) Oken	Crassulaceae	TAF	Herb	Unintentional
24	*Caldesia oligococca* (F.Muell.) Buchanan	Alismataceae	AUS	Herb	Unintentional
25	*Calotropis gigantea* (L.) Dryand.	Apocynaceae	TAF	Shrub	Unintentional
26	*Calotropis procera* (Aiton) Dryand.	Apocynaceae	TAF	Shrub	Unintentional
27	*Cardiospermum halicacabum* L.	Sapindaceae	SAM	Climber	Unintentional
28	*Catharanthus pusillus* (Murray) G.Don	Apocynaceae	TAM	Herb	Ornamental
29	*Celosia argentea* L.	Amaranthaceae	TAM	Herb	Food
30	*Ceratophyllum demersum* L.	Ceratophyllaceae	TAM	Herb	Unintentional
31	*Chamaecrista absus* (L.) H.S. Irwin & Barneby	Fabaceae	TAM	Herb	Unintentional
32	*Chamaecrista pumila* (Lam.) K. Larsen	Fabaceae	TAM	Herb	Unintentional

(continued)

Table 5.1 (continued)

S.No.	Species	Family	Origin	Life form	Introduction
33	*Chenopodium album* L.	Chenopodiaceae	EU	Herb	Food
34	*Chenopodium murale* L.	Chenopodiaceae	TAM	Herb	Unintentional
35	*Chloris barbata* Sw.	Poaceae	TAM	Grass	Unintentional
36	*Chromolaena odorata* (L.) R.M. King & H. Rob.	Asteraceae	WI	Shrub	Unintentional
37	*Chrozophora rottleri* (Geiseler) A.Juss. ex Spreng.	Euphorbiaceae	TAF	Herb	Unintentional
38	*Cissampelos pareira* L.	Menispermaceae	SAM	Climber	Unintentional
39	*Cleome gynandra* L.	Capparaceae	TAM	Herb	Unintentional
40	*Cleome monophylla* L.	Capparaceae	TAF	Herb	Unintentional
41	*Cleome viscosa* L.	Capparaceae	TAM	Herb	Unintentional
42	*Clitoria ternatea* L.	Fabaceae	TAM	Climber	Unintentional
43	*Combretum indicum* (L.) DeFilipps	Combretaceae	ML	Climber	Unintentional
44	*Convolvulus arvensis* L.	Convolvulaceae	EU	Herb	Unintentional
45	*Corchorus aestuans* L.	Malvaceae	TAM	Herb	Unintentional
46	*Corchorus fascicularis* Lam.	Malvaceae	TAM	Herb	Unintentional
47	*Corchorus olitorius* L.	Malvaceae	TAF	Herb	Unintentional
48	*Corchorus tridens* L.	Malvaceae	TAF	Herb	Unintentional
49	*Corchorus trilocularis* L.	Malvaceae	TAF	Herb	Unintentional
50	*Cosmos sulphureus* Cav.	Asteraceae	SAM	Herb	Unintentional
51	*Crassocephalum crepidioides* (Benth.) S. Moore	Asteraceae	TAM	Herb	Unintentional
52	*Croton bonplandianus* Baill.	Euphorbiaceae	SAM	Herb	Unintentional
53	*Cryptostegia grandiflora* Roxb. ex R. Br.	Apocynaceae	MG	Liana	Unintentional
54	*Cuscuta chinensis* Lam.	Cuscutaceae	MR	Climber	Unintentional
55	*Cuscuta reflexa* Roxb.	Cuscutaceae	MR	Climber	Unintentional
56	*Cynodon dactylon* (L.) Pers.	Poaceae	TAF	Grass	Unintentional
57	*Cyperus difformis* L.	Cyperaceae	TAM	Herb	Unintentional
58	*Cyperus iria* L.	Cyperaceae	TAM	Herb	Unintentional
59	*Datura innoxia* Mill.	Solanaceae	TAM	Shrub	Unintentional
60	*Datura metel* L.	Solanaceae	TAM	Shrub	Unintentional
61	*Dicliptera paniculata* (Forssk.) I. Darbysh.	Acanthaceae	TAM	Herb	Unintentional
62	*Digera muricata* (L.) Mart.	Amaranthaceae	NAM	Herb	Unintentional
63	*Digitaria ciliaris* (Retz.) Koeler	Poaceae	TAM	Grass	Unintentional
64	*Dinebra retroflexa* (Vahl) Panz.	Poaceae	TAM	Grass	Unintentional
65	*Duranta erecta* L.	Verbenaceae	TAM	Shrub	Agroforestry
66	*Echinochloa colona* (L.) Link	Poaceae	SAM	Grass	Unintentional
67	*Echinochloa crus-galli* (L.) P.Beauv.	Poaceae	SAM	Grass	Unintentional
68	*Echinops echinatus* Roxb.	Asteraceae	AF	Herb	Unintentional
69	*Eclipta prostrata* (L.) L.	Asteraceae	TAM	Herb	Unintentional

(continued)

Table 5.1 (continued)

S.No.	Species	Family	Origin	Life form	Introduction
70	*Eichhornia crassipes* (Mart.) Solms	Pontederiaceae	TAM	Herb	Ornamental
71	*Emilia sonchifolia* (L.) DC. ex DC.	Asteraceae	TAM	Herb	Unintentional
72	*Erigeron canadensis* L.	Asteraceae	SAM	Herb	Unintentional
73	*Erigeron obliqum* L. Mant.	Asteraceae	NAM	Herb	Unintentional
74	*Euphorbia heterophylla* L.	Euphorbiaceae	TAM	Herb	Unintentional
75	*Euphorbia hirta* L.	Euphorbiaceae	TAM	Herb	Unintentional
76	*Euphorbia thymifolia* L.	Euphorbiaceae	TAM	Herb	Unintentional
77	*Euphorbia tirucalli* L.	Euphorbiaceae	TAF	Shrub	Unintentional
78	*Euphorbia umbellata* (Pax) Bruyns	Euphorbiaceae	TAM	Shrub	Ornamental
79	*Evolvulus nummularius* (L.) L.	Convolvulaceae	TAM	Herb	Unintentional
80	*Galinsoga parviflora* Cav.	Asteraceae	TAM	Herb	Unintentional
81	*Glossocardia bosvallia* (L.f.) DC.	Asteraceae	WI	Herb	Unintentional
82	*Gnaphalium pensylvanicum* Willd.	Asteraceae	TAM	Herb	Unintentional
83	*Gnaphalium polycaulon* Pers.	Asteraceae	TAM	Herb	Unintentional
84	*Gnaphalium purpureum* L.	Asteraceae	TAM	Herb	Unintentional
85	*Gomphrena celosioides* Mart.	Amaranthaceae	BR	Herb	Unintentional
86	*Grangea maderaspatana* (L.) Poir	Asteraceae	SAM	Herb	Unintentional
87	*Hyptis suaveolens* (L.) Poit.	Lamiaceae	SAM	Herb	Unintentional
88	*Impatiens balsamina* L.	Balsaminaceae	TAM	Herb	Ornamental
89	*Imperata cylindrica* (L.) Raeusch.	Poaceae	TAM	Grass	Unintentional
90	*Indigofera astragalina* DC.	Fabaceae	TAM	Herb	Unintentional
91	*Indigofera glandulosa* Wendl.	Fabaceae	TAM	Herb	Unintentional
92	*Indigofera linifolia* (L.f.) Retz.	Fabaceae	SAM	Herb	Unintentional
93	*Indigofera linnaei* Ali	Fabaceae	TAF	Herb	Unintentional
94	*Indigofera trita* L.f.	Fabaceae	TAF	Herb	Unintentional
95	*Ipomoea carnea* Jacq.	Convolvulaceae	TAM	Shrub	Unintentional
96	*Ipomoea eriocarpa* R. Br.	Convolvulaceae	TAF	Herb	Ornamental
97	*Ipomoea hederifolia* L.	Convolvulaceae	TAM	Herb	Unintentional
98	*Ipomoea nil* (L.) Roth	Convolvulaceae	NAM	Climber	Unintentional
99	*Ipomoea obscura* (L.) Ker Gawl.	Convolvulaceae	TAF	Herb	Unintentional
100	*Ipomoea pes-tigridis* L.	Convolvulaceae	TAF	Herb	Unintentional
101	*Ipomoea quamoclit* L.	Convolvulaceae	TAM	Climber	Ornamental
102	*Iseilema laxum* Hack.	Poaceae	TAM	Grass	Unintentional
103	*Jatropha curcas* L.	Euphorbiaceae	TAM	Shrub	Agroforestry
104	*Jatropha gossypifolia* L.	Euphorbiaceae	TAM	Shrub	Unintentional
105	*Lagascea mollis* Cav.	Asteraceae	TAM	Herb	Unintentional
106	*Lantana camara* L.	Verbenaceae	TAM	Shrub	Ornamental

(continued)

Table 5.1 (continued)

S.No.	Species	Family	Origin	Life form	Introduction
107	*Lathyrus aphaca* L.f.	Fabaceae	EU	Herb	Unintentional
108	*Leonotis nepetifolia* (L.) R. Br.	Lamiaceae	TAF	Herb	Unintentional
109	*Leucaena leucocephala* (Lam.) de Wit	Fabaceae	TAM	Tree	Fodder
110	*Ludwigia adscendens* (L.) H. Hara	Onagraceae	TAM	Herb	Unintentional
111	*Ludwigia octovalvis* (Jacq.) P.H. Raven	Onagraceae	TAF	Herb	Unintentional
112	*Ludwigia perennis* L.	Onagraceae	TAF	Herb	Unintentional
113	*Lysiloma latisiliquum* (L.) Benth.	Fabaceae	TAM	Tree	Fodder
114	*Malachra capitata* (L.) L.	Malvaceae	TAM	Herb	Unintentional
115	*Malvastrum coromandelianum* (L.) Garcke	Malvaceae	TAM	Herb	Unintentional
116	*Martynia annua* L.	Pedaliaceae	TAM	Herb	Unintentional
117	*Medicago lupulina* L.	Fabaceae	EU	Herb	Unintentional
118	*Medicago polymorpha L.*	Fabaceae	EU	Herb	Unintentional
119	*Medicago sativa* L.	Fabaceae	SWA	Herb	Unintentional
120	*Melilotus albus* Medik.	Fabaceae	EU	Herb	Fodder
121	*Melochia corchorifolia* L.	Malvaceae	TAM	Herb	Unintentional
122	*Merremia aegyptia* (L.) Urb.	Convolvulaceae	TAM	Climber	Unintentional
123	*Merremia emarginata* (Burm.f.) Hallier f.	Convolvulaceae	TAF	Climber	Unintentional
124	*Mikania micrantha* Kunth	Asteraceae	TAM	Climber	Unintentional
125	*Mimosa pudica* L.	Fabaceae	BR	Herb	Unintentional
126	*Mirabilis jalapa* L.	Nyctaginaceae	PU	Herb	Ornamental
127	*Monochoria vaginalis* (Burm.f.) C. Presl	Pontederiaceae	TAM	Herb	Unintentional
128	*Monstera deliciosa* Liebm.	Araceae	MX	Climber	Unintentional
129	*Nicotiana plumbaginifolia* Viv.	Solanaceae	TAM	Herb	Unintentional
130	*Ocimum americanum* L.	Lamiaceae	TAM	Herb	Unintentional
131	*Opuntia elatior* Mill.	Cactaceae	SAM	Shrub	Unintentional
132	*Opuntia ficus-indica* (L.) Mill.	Cactaceae	NAM	Shrub	Unintentional
133	*Oxalis corniculata* L.	Oxalidaceae	EU	Herb	Unintentional
134	*Oxalis debilis* Kunth	Oxalidaceae	SAM	Herb	Unintentional
135	*Panicum maximum* Jacq.	Poaceae	TAF	Grass	Unintentional
136	*Parkinsonia aculeata* L.	Fabaceae	MX	Shrub	Unintentional
137	*Parthenium hysterophorus* L.	Asteraceae	TAM	Herb	Unintentional
138	*Passiflora foetida* L.	Passifloraceae	SAM	Herb	Ornamental
139	*Pedalium murex* L.	Pedaliaceae	TAM	Herb	Unintentional
140	*Peperomia pellucida* (L.) Kunth	Piperaceae	SAM	Herb	Unintentional
141	*Physalis minima* L.	Solanaceae	TAM	Herb	Unintentional
142	*Pilea microphylla* (L.) Liebm.	Urticaceae	SAM	Herb	Unintentional
143	*Pistia stratiotes* L.	Araceae	TAM	Herb	Unintentional
144	*Pithecellobium dulce* (Roxb.) Benth.	Fabaceae	TAM	Tree	Unintentional

(continued)

Table 5.1 (continued)

S.No.	Species	Family	Origin	Life form	Introduction
145	*Portulaca oleracea* L.	Portulacaceae	SAM	Herb	Food
146	*Portulaca quadrifida* L.	Portulacaceae	TAM	Herb	Unintentional
147	*Prosopis juliflora* (Sw.) DC.	Fabaceae	MX	Tree	Unintentional
148	*Ricinus communis* L.	Euphorbiaceae	TAF	Tree	Unintentional
149	*Ruellia tuberosa* L.	Acanthaceae	TAM	Herb	Unintentional
150	*Saccharum spontaneum* L.	Poaceae	ML	Grass	Unintentional
151	*Scoparia dulcis* L.	Scrophulariaceae	TAM	Herb	Unintentional
152	*Senna alata* (L.) Roxb.	Fabaceae	WI	Shrub	Unintentional
153	*Senna obtusifolia* (L.) H.S. Irwin & Barneby	Fabaceae	TAM	Herb	Unintentional
154	*Senna occidentalis* (L.) Link	Fabaceae	SAM	Herb	Unintentional
155	*Senna tora* (L.) Roxb.	Fabaceae	SAM	Herb	Unintentional
156	*Sesbania bispinosa* (Jacq.) W. Wight	Fabaceae	TAM	Herb	Unintentional
157	*Sesbania sesban* (L.) Merr.	Fabaceae	TAF	Shrub	Unintentional
158	*Sida acuta* Burm.f.	Malvaceae	TAM	Herb	Unintentional
159	*Solanum torvum* Sw.	Solanaceae	WI	Shrub	Unintentional
160	*Solanum americanum* Mill.	Solanaceae	TAM	Herb	Unintentional
161	*Sonchus asper* (L.) Hill	Asteraceae	MR	Herb	Unintentional
162	*Sonchus oleraceus* (L.) L.	Asteraceae	MR	Herb	Unintentional
163	*Sorghum halepense* (L.) Pers.	Poaceae	TAF	Grass	Unintentional
164	*Spermacoce articularis* L.f.	Rubiaceae	TAM	Herb	Unintentional
165	*Spermacoce hispida* L.	Rubiaceae	TAM	Herb	Unintentional
166	*Stachytarpheta jamaicensis* (L.) Vahl	Verbenaceae	TAM	Climber	Unintentional
167	*Synedrella nodiflora* (L.) Gaertn.	Asteraceae	WI	Herb	Unintentional
168	*Torenia fournieri* Linden ex E. Fourn.	Scrophulariaceae	AUS	Herb	Unintentional
169	*Trema orientalis* (L.) Blume	Ulmaceae	TAF	Shrub	Unintentional
170	*Tribulus terrestris* L.	Zygophyllaceae	TAM	Herb	Unintentional
171	*Tridax procumbens* (L.) L.	Asteraceae	TAM	Herb	Unintentional
172	*Triumfetta rhomboidea* Jacq.	Malvaceae	TAM	Herb	Unintentional
173	*Turnera ulmifolia* L.	Turneraceae	TAM	Herb	Unintentional
174	*Typha angustifolia* L.	Typhaceae	TAM	Herb	Unintentional
175	*Urena lobata* L.	Malvaceae	TAM	Shrub	Unintentional
176	*Vaccaria hispanica* (Mill.) Rauschert	Caryophyllaceae	EU	Herb	Unintentional
177	*Waltheria indica* L.	Malvaceae	TAM	Shrub	Unintentional
178	*Xanthium strumarium* L.	Asteraceae	TAM	Herb	Unintentional
179	*Youngia japonica* (L.) DC.	Asteraceae	SAM	Herb	Unintentional

Origin (OR):*TAM* Tropical America, *NAM* North America, *BR* Brazil, *SWA* Southwest Asia, *EU* Europe, *MR* Mediterranean region, *MX* Mexico, *SAM* South America, *TAF* Tropical Africa, *WI* West Indies, *AF* Afghanistan, *ML* Malaysia, *AUS* Australia, *MG* Madagascar, *PU* Peru

Fig. 5.2 The ten families containing the most invasive species in Central India

family size is often perceived as a predictor of the number of alien plants (Weber 1997; Daehler 1998; Pyšek 1998; Li et al. 2001; Weber et al. 2008) as large families contribute too many species. As demonstrated by Khuroo et al. (2012), the dominance of species-rich families such as Asteraceae, Fabaceae and Poaceae are largely due to sampling effect as the relative percentage of invasive species occurring within the families might be disproportionate.

5.6.2 Nativity of Invasive Taxa

Around 53% of invasive taxa in Central India originate from Tropical America, followed by Tropical Africa (14%), South America (12.2%) and other regions (Fig. 5.3). Nayar (1977) stated that 55% of the Indian flora is American in origin. America has been reported to be the largest contributor of invasive taxa in other studies by Xu et al. (2004), Sharma et al. (2005), Khuroo et al. (2012), Sekar (2012), Wang and Niu (2016) and Sundarapandian and Subashree (2017). As species of the Northern Hemisphere have a more competitive evolutionary history, they are better competitors than those from Southern Hemisphere (Darwin 1859), which accounts for their successful naturalization, and as such the continents in the Northern Hemisphere are considered to be the major donors of alien species that are naturalized to all other continents (van Kleunen et al. 2015).

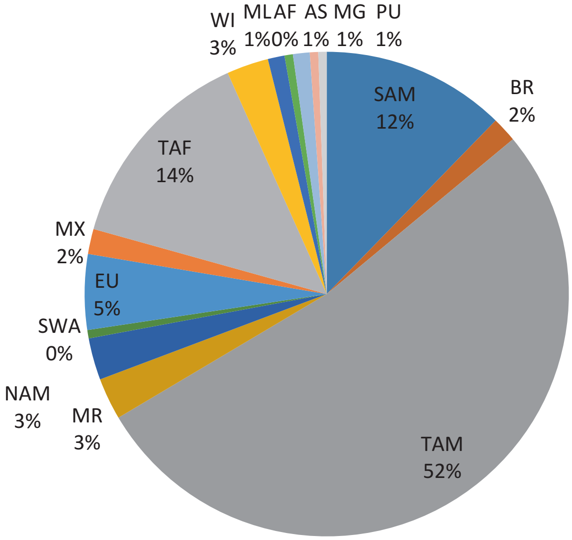

Fig. 5.3 Percentage contribution of different nativities to the invasive flora of Central India Abbreviations: *OR* Origin, *TAM* Tropical America, *NAM*, North America, *BR* Brazil, *SWA* Southwest Asia, *EU* Europe, *MR* Mediterranean region, *MX* Mexico, *SAM* South America, *TAF* Tropical Africa, *WI* West Indies, *AF* Afghanistan, *ML* Malaysia, *AUS* Australia, *MG* Madagascar, *PU* Peru

5.6.3 Life Forms of Invasive Taxa

Herbaceous elements dominate about 69% of the Central Indian invasive taxa (Fig. 5.4). The dominance of herbs in the alien flora has also been reported by Khuroo et al. (2007), Reddy (2012), Sekar (2012) and Srivastava et al. (2014). The ability to tolerate harsh environmental conditions and higher viability of propagules (Sharma et al. 2005; Shukla and Sinha 2012) are some of the attributes that help the herbaceous community to dominate the alien flora. Moreover, annual alien species may have high fecundity, early germination, fast growth rates, autonomous pollination and effective dispersal mechanisms that aid in successful invasion (Golivets 2014).

5.7 An Outline of the Most Common Top Ten Noxious Invaders of Central India

5.7.1 *Lantana camara* L.

Lantana camara, a member of Verbenaceae, is well-known as it is among the 'World's Worst' invaders list of the Global Invasive Species Database (2017a; http://www.iucngisd.org/gisd/speciesname/Lantana+camara), as it disrupts ecosystem functions and causes biodiversity loss to a great extent and as there are over 650 varieties that are naturalized in around 60 countries (Sundarapandian et al. 2015). This species, a native of Tropical America, was first introduced in India in 1809 as an ornamental plant in Kolkata from where it escaped and became invasive (Kohli et al. 2006). It dominates in diverse habitats and tolerates a wide range of climatic

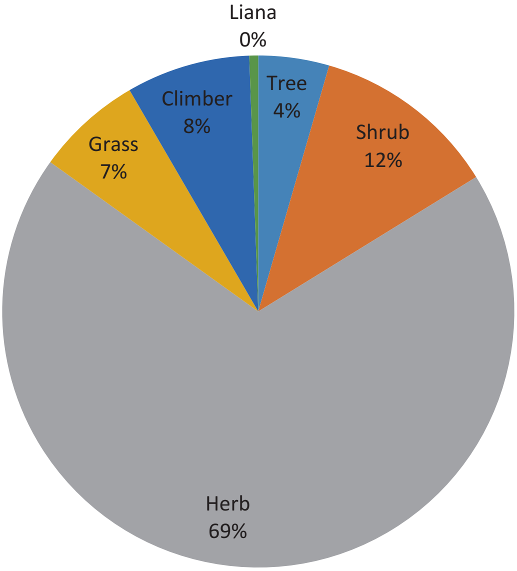

Fig. 5.4 Life form-wise contribution of invasive species in Central India

conditions (Pandey and Chauhan 2012) as it has prolific flowering, fruiting and seed dispersal mechanisms, besides allelopathy. Invasion by this plant is favoured by anthropogenic disturbances such as cattle grazing, fires and creation of canopy gaps (Gooden et al. 2009). According to Hiremath and Sundaram (2005, 2013), *Lantana camara* has the potential to recover faster in the event of a fire than the native vegetation, which would lead to a self-perpetuating *Lantana*-fire cycle. This invasive species is known to alter the fuel loads of forests causing the fires to be more intense and severe (Hiremath and Sundaram 2013). *Lantana camara* has climbing stems that could reach > 20 metres, almost up to the forest canopy, and could therefore cause crown fires when they burn (Tireman 1916; Hiremath and Sundaram 2005). Since this species possesses traits of fire-adapted species, its invasion is facilitated by fire, and once it gets established, it fuels further fires (Hiremath and Sundaram 2005). The authors have noted that in a span of 40 years, the scenario changed from *Lantana camara* suppression by fires to the promotion of invasion of *Lantana camara* by fires in Biligiri Rangaswamy Temple Tiger Reserve, India. Even though the above-said study is from Southern India, it highlights the potential of *Lantana camara* in dramatically changing forest characteristics, and it could entail similar consequences in Central Indian forests as well. Although it is considered as a noxious weed, it is sometimes used for firewood, medicine and as mulch (Pandey and Chauhan 2012). Control of this plant is often very difficult, especially due to its hybridization.

5.7.2 *Parthenium hysterophorus* L.

Parthenium hysterophorus, a member of Asteraceae, is of American origin but is now pantropical in distribution (Knox et al. 2011). This invasive plant is thought to have entered India between 1950s and 1960s along with the food grains imported from the United States, although the mode of introduction was not conclusive (Kohli et al. 2006). This prefers to invade those areas that are recently disturbed and where the top soil has been removed (Shabbir and Bajwa 2007). It has an extensive root system and produces abundant flowers and fruits. The seeds germinate year-round under favourable temperature and moisture (Dhileepan and Senaratne 2009). Apart from negatively impacting biodiversity, this plant is often known to deplete nutrients from the soil and exudates allelochemicals. *P. hysterophorus* also affects the health of humans and cattle, causing skin and respiratory illnesses (Kohli et al. 2006). Managing this weed is often difficult as the released biocontrol agent, a Mexican beetle (*Zygogramma bicolorata*), attacks nontarget species as well.

5.7.3 *Ageratum conyzoides* L.

Another member of Asteraceae, *Ageratum conyzoides*, is known to exist in India before 1882, although there is no authentic information on the mode or purpose of introduction (Kohli et al. 2006). This American weed adapts itself to a wide range of habitats, although it prefers disturbed sites. It copiously produces fruits with pappus that help in easy dispersal of photoblastic seeds that could remain viable for 1 year by wind, animals and water, and hence it is known to affect agricultural fields causing economic losses (Kohli et al. 2006). Some success has been achieved in controlling the infestation of this weed by chemical methods (Global Invasive Species Database 2017b; http://www.iucngisd.org/gisd/speciesname/Ageratum+conyzoides).

5.7.4 *Prosopis juliflora* (Sw.) DC.

A native of Mexico, *Prosopis juliflora* (Fabaceae), was introduced to India during the nineteenth century to prevent desertification and as a fuelwood tree. It exhibits wide tolerance to drought and salinity (Pasiecznik et al. 2001) and has fast growth rates (Shiferaw et al. 2004), abundant seed production with long viability (Aung and Koike 2015) and low palatability, which gives it a competitive edge over native species (Hiremath and Sundaram 2005). It is often perceived as a globally invasive alien taxon that threatens biodiversity (Shackleton et al. 2014; Shackleton et al. 2015) and large drought-prone areas, where the local people depend on groundwater are now under invasion by this plant (Dzikiti et al. 2017). Cutting stems to meet firewood requirements and root excavation is often the preferred method of controlling its spread.

5.7.5 *Hyptis suaveolens* (L.) Poit.

Originally of American origin and a Lamiaceae member, *Hyptis suaveolens* has caused considerable havoc in many diverse habitats. It is pantropical in distribution and has a pungent odour that repels cattle (Padalia et al. 2014). It exhibits allelopathy, fast growth and copious seed production and is an aggressive colonizer of disturbed habitats. It is known to alter species diversity in dry deciduous forests (Sharma et al. 2009; Sharma et al. 2017) and threatens the course of natural succession in invaded ecosystems (Padalia et al. 2013; Padalia et al. 2014). It has a tendency to form monocultures displacing native flora as observed by Sharma et al. (2017). Despite these negative impacts, *Hyptis suaveolens* also has anticancerous and tumorigenic substances in its vegetative parts (Mudgal et al. 1997; Peerzada 1997; Padalia et al. 2014) and could be used as a green manure (Rao et al. 2012).

5.7.6 *Xanthium strumarium* L.

Another tropical American invader of Asteraceae, *Xanthium strumarium*, is a ruderal, herbaceous annual plant that is commonly found on farmlands, over-grazed pastures and other disturbed sites. It has been recorded to be widespread in many states of India (Holm et al. 1991), especially Madhya Pradesh and Maharashtra (from Central India), and parts of this plant are used in the traditional siddha, ayurveda and folk medicinal systems (India Biodiversity Portal, https://indiabiodiversity.org/species/show/266687). It is also documented by Wagh and Jain (2015) from Jhabua district, Madhya Pradesh, Central India. It is known to reduce forage production (Seifu et al. 2017). This noxious weed can cause contact dermatitis and toxicity in humans and livestock when consumed, and it has herbicide resistance (Kaur et al. 2014). One prominent reason for its invasion success is its competitive genetic makeup (Gray et al. 1986; Chikuruwo et al. 2017). Another characteristic feature of this plant is that it produces burred seeds which facilitate dispersal and range expansion (Kaur et al. 2014).

5.7.7 *Argemone mexicana* L.

Argemone mexicana, a native of Mexico and a Papaveraceae member, is a noxious weed that has prickly stems, spiny leaves and bright yellow, showy flowers and a yellow sap. It is commonly found along roadsides, rail tracks, abandoned fallows, agricultural fields, wastelands and other disturbed areas. This species produces on an average of 60–90 capsules per plant, and each capsule bears 300–400 seeds (Holm et al. 1979, Namkeleja et al. 2014), and the seeds are often toxic (Das and Khanna 1997; Karlsson et al. 2003). The seeds often show dormancy, and when they germinate, they create deleterious effects on neighbouring native species (Alemayehu and Desalegn 2016). Besides prolific seed production and long seed dormancy that contributes to soil seed bank, this invader also exerts allelopathic effects on the native community (Namkeleja et al. 2014). Eradication of this invader is often difficult even in a well-managed field as it requires removal of the plant before shedding and periodic removal of the juveniles when they germinate from the soil seed bank (Karlsson et al. 2003).

5.7.8 *Chromolaena odorata* (L.) R.M. King & H. Rob.

Chromolaena odorata, a Fabaceae member and native to the West Indies, is a perennial shrub that is known to invade many commercial plantations and orchards and increases susceptibility to forest fires (Misra et al. 2012). It smothers the existing native plant communities, and it often threatens the livelihoods of farmers and plantation owners (Uyi et al. 2014). As stated by Kohli et al. (2012), this species has high regenerative and biotic potential with profuse seed production that is viable for a long time. This weed often spreads and grows rampantly causing huge economic losses (Pimentel 2002; Perrings et al. 2010; Uyi et al. 2014), and it is often difficult to completely eliminate the weed.

5.7.9 *Eichhornia crassipes* (Mart.) Solms

Eichhornia crassipes or water hyacinth is a member of Pontederiaceae. It is a very serious invader from Tropical America that was originally introduced in India in the early twentieth century for its beauty (Misra et al. 2012). However, it is now naturalized in many parts of the world and where it chokes a variety of freshwater aquatic ecosystems. It has a better competitive ability than most native aquatic plants, and it grows and propagates rapidly and easily (Tellez et al. 2008). The seeds of this plant sink in the mud, where they can remain dormant and viable for up to 20 years and under ideal conditions. *E. crassipes* can double its number within 10 days (Misra et al. 2012). This invader often dampens the aquatic ecosystem dynamics by affecting the water flow, blocking sunlight and causing eutrophication, which lead to the death of many aquatic flora and fauna (Frezina 2013). It can be controlled biologically by the introduction of weevils such as *Neochetina eichhorniae* and *Neochetina bruchi* (Misra et al. 2012).

5.7.10 *Mikania micrantha* Kunth

Mikania micrantha is another obnoxious invader of Asteraceae that was introduced from Tropical America. It is a terrestrial vine that is pantropical in distribution (Day 2012; Banerjee and Dewanji 2017b). This highly invasive species was introduced in India for camouflaging airfields during World War II and to serve as a ground cover in tea plantations (Tripathi et al. 2012). It spreads profusely by producing seeds, ramets and perennating rosettes (Swamy and Ramakrishnan 1987; Tripathi et al. 2012) and by effective seed dispersal modes via wind, water and animals (Swamy and Ramakrishnan 1987; Ismail and Mah 1993; Sundarapandian et al. 2015). It suppresses neighbouring plants by climbing onto them and smothering them, in addition to allelopathy. An important observation by Banerjee and Dewanji (2017a) reveals that *M. micrantha* could dominate other alien species as well, which makes it a superior alien. Another interesting observation (Banerjee and Dewanji 2017b) was *Mikania micrantha* exhibits intraspecific trait plasticity by which it maintains its population by season-specific modulation of its vegetative traits. Shukla and Sinha (2012) stated that this vine was previously unknown in Chhattisgarh and has started invading the state recently.

5.8 Invasive Species and Climate Change

One of the pressing problems faced by the world today is the increase in atmospheric CO_2 levels (408.35 ppm as in February 2018; https://www.co2.earth) associated with global warming and climate change. Climate change has a serious impact on biodiversity, and it may drive many species, especially those that are endemic, to extinction (Kueppers et al. 2005; Loarie et al. 2008). Hence, it is time to predict and understand the impact of the climate change on biodiversity, whether native or invasive.

Plants usually respond to climate change by evolving, acclimating, moving or dying. That is to say they either adapt by changing their phenology and life cycles or shift their ranges to more suitable habitats or else become extinct, when they could no longer adapt or move (Sullivan and Clark 2007; Rinawati et al. 2013). Rise in atmospheric CO_2 levels might favour woody plant invasion in grassland ecosystems (Kgope et al. 2010; McDougall et al. 2011). There are substantial evidences that show that the plants are shifting their ranges along altitudinal and latitudinal gradients to combat the effects of climate change (Parmesan and Yohe 2003; Saikia et al. 2016). It has been predicted that all species may shift polewards as a response to climate change and, consequently, invasive species might undergo range expansion (Katz and Ibanez 2016). Invasive species might invade into areas where they were previously unknown, and they may enhance their competitive abilities to outcompete the native species (Raizada et al. 2009; Kannan et al. 2013). Many invasive species express a gradual variation in traits along an environmental gradient, and there are documented cases of invaders showing local adaptations and phenotypic plasticity (Monty et al. 2013). As such, they might be ‘preadapted’ and better prepared than native species to face the future climate (Venette 2013). In addition to climate change, due to increasing trade and globalization, many plants will be introduced in new areas which may not have been hospitable to invasion prior to climate change, but may become now.

Alien plants may achieve a greater colonization success than the indigenous ones under a warmer climate, and those that are not invasive under the current climatic conditions might become invasive under future climates (Haeuser et al. 2017). However, not all invasive plants undergo range expansion as some invasive plants of the United States have shown range contractions as a result of climate change

(Bradley et al. 2009; Allen and Bradley 2016). Climate change may affect habitat suitability, which in turn might affect the establishment of invasive species (Walther et al. 2009; Seebens et al. 2015). Moreover, if an invasive species fails to adjust its phenology to match new climatic conditions, it may not be able to establish successfully (Sanderson and Antunes 2013). Tropical forests that are already under tremendous pressure could be expected to be adversely impacted by climate change, and slowing species invasion in these forests could improve their resiliency.

5.9 Monitoring and Modelling

Monitoring and modelling bioinvasions are essential to plan conservation programmes for risk assessment and prediction of the possible impacts of potential invaders in the future. Monitoring, together with identification of 'invasion hotspots' as done by Adhikari et al. (2015) would help us to better tackle the problem of invasion. Continuous monitoring could help in preventing secondary plant invasion and could predict the possibility of recovery of the ecosystem (Gooden et al. 2009), if coupled with active management. Hierro et al. (2005) and Yang et al. (2013) have suggested that comparing the performance of an invasive species in its native and introduced ranges could help us understand its mechanism of invasion. Using individual or a combination of species traits to envisage the potential invasiveness of species and preparing 'blacklists' of potentially harming species and 'whitelists' of species that are unlikely to cause any harm (Colautti et al. 2014) could enable easy and better monitoring. One extremely critical and very often overlooked problem in the preparation of lists or ecological datasets is the lack of taxonomic clearing, which make them scientifically unreliable. This stresses the necessity to provide the currently accepted scientific names and not synonyms or (even worse) common names, besides assigning the correct ecological status (such as rare, naturalized, invasive, etc.) wherever relevant (Murray et al. 2017).

Monitoring biological invasions has already been done in many regions, and the data are constantly updated in respective databases to provide easy access to information and early warning to land managers, forest department officials and the general public. Some of such online repositories are Global Invasive Species Database (GISD; http://www.iucngisd.org/gisd/), Island Biodiversity and Invasive Species Database (IBIS; http://ibis.fos.auckland.ac.nz/page/invasive-species.aspx), CABI's Invasive Species Compendium (http://www.cabi.org/isc/), Global Naturalized Alien Flora (GloNAF; https://glonaf.org/), Global Invasive Alien Species Information Partnership Gateway (http://giasipartnership.myspecies.info/en), Global Register of Introduced and Invasive Species (GRIIS; http://www.griis.org/), European Alien Species Information Network (EASIN; https://easin.jrc.ec.europa.eu/), Delivering Alien Invasive Species Inventories for Europe (DAISIE; http://www.europe-aliens.org/), Great Britain's Non-Native Species Information Portal (GBNNSIP; http://www.nonnativespecies.org/factsheet/), Southern African Plant Invaders Atlas (SAPIA; http://www.invasives.org.za/) and the Invasive Alien Plant Species in India (http://www.bsienvis.nic.in/Database/Invasive_Alien_species_15896.aspx). The

Directorate of Plant Protection, Quarantine and Storage (http://ppqs.gov.in/) under the Ministry of Agriculture and Farmers Welfare, Government of India, monitors the import and export of plants or plant materials so as to reduce the introduction of pests both in India and elsewhere.

Modelling has helped to better understand the possible implications of biological invasions. Modelling methods such as phenomenological, mechanistic and hybrid models were developed to understand the processes of spread and distribution of invasive plants, which would help effective management, although these are seldom used due to the lack of expertise and limited data (Gallien et al. 2010; Caplat et al. 2012; Guisan et al. 2013; Adams et al. 2015). Species distribution models (bioclimatic envelope models) help quicker and cost-effective identification of key environmental or climatic factors that influence the distribution of alien invasive species and prediction of their potential distribution under the current and future climatic scenarios (Guisan and Thuiller 2005; Shmueli 2010; Vicente et al. 2010,2011,2013). As described by Peterson et al. (1999), a species distribution model is a mathematical description of a species distribution in environmental space, which could be used to predict the species distribution in geographical space (Motloung et al. 2014). These bioclimatic envelope models are used to define the climate and habitats of both native and invasive species based on their geographic distribution (Franklin 1995; Guisan and Zimmermann 2000; Pearson and Dawson 2003; Guisan and Thuiller 2005; Bradley et al. 2010). Yet these models have a limitation that they focus only on species-climate relationships and do not include other determinants such as nutrient uptake, competition, disturbance and land use history, etc. (Bradley et al. 2010).

Adhikari et al. (2015) predicted the areas of invasion of 155 alien species based on the species occurrence data from Global Biodiversity Information Facility. The authors calibrated the model using bioclimatic data from North America, South America, Africa, Europe and Australia as well as the bioclimatic data for the projected range of invasion in India. They found that about 49% of the total geographical area of the country is predicted to be vulnerable to invasion. The states Madhya Pradesh and Chhattisgarh showed low and medium agreements to the model projections that depicted the climatic suitability for the alien invasive species. The projections from North America, Africa and Australia seemed to have more climatic suitability with Central India. Padalia et al. (2014) predicted the potential invasive range of the species, *Hyptis suaveolens*, and their results revealed that areas in Central India were found to be the most suitable for range expansion of this invasive species. Padalia and Bahuguna (2017) identified the potential hotspots of invasion by 98 alien invasive species across Central India. They proved that almost all the vegetation types of Central India are susceptible to invasion by multiple invasive species, especially the dominant types. The authors found that the hotspots of invasion in Central India are largely concentrated on the areas with high to very high species richness. As evident from the above, such ecological niche modelling studies would help in better tracking and tackling of invasive species in the future.

5.10 Charting the Course Ahead: Towards Efficient Management

Early detection and rapid response are the most effective steps to check the spread of invasive plants (Allen and Bradley 2016; Banerjee and Dewanji 2017a). Invaded areas are often cleared using physical (hand-pulling, cutting, mowing, burning, etc.), chemical (application of pesticides, herbicides, etc.), biological (release of fungi, insects, etc.) or cultural (mulching, planting native species, canopy closure, etc.) methods based on the context and type of invasion. Very often, even after an invasive species is removed, the ecosystem does not return to previous conditions due to other problems such as secondary invasions (Zavaleta et al. 2001; Loo et al. 2009; Pearson et al. 2016) or 'legacy effects', where the original invader left behind long-lasting effects, such as the alteration of soil nutrients (D'Antonio and Meyerson 2002; Marchante et al. 2009; Gaertner et al. 2012). At times the applied control methods attack the nontarget, native species as well. Although the dry forests are viewed to be more resilient than the wet forests (Ewel 1977; Murphy and Lugo 1986), control measures might not be successful if the demands and dependence of the local people for fuel, fodder and other income-generating opportunities (in the case of non-timber forest produce) are not met (Sagar and Singh 2005).

Successful management requires the ability to identify the factors that may lead to the spread of invasive species and predict their probable geographical range (Cabra-Rivas et al. 2016). Hence, it becomes mandatory for the concerned managers to gain expertise on modelling and apply the same to thwart the problems arising out of invasions, well in advance. Eradication of invasive species requires public participation as some people/communities do not want the species to be removed for its beauty or desirability (see Lindemann-Matthies 2016). Eradication also has a poor track record because action often occurs too late and entire populations of invasives cannot be killed. Therefore, there is a need to spread awareness among the residing communities regarding the potential problems that the invasive weeds may pose, if not in the present, in the future. Lower occurrence of invasive species in an area due to less-frequent introductions could be the result of increased monitoring and surveillance or the fact that many potential invasive species are still in the pre-expansion phase (Murray and Phillips 2012). Thus, landowners, managers and forest officials need to be on a constant watch to successfully confine the invaders and protect their respective regions from further invasion or other negative impacts arising from the control measure itself.

5.11 Conclusion

Invasion of species is a major cause of biodiversity loss and profoundly influences ecosystem processes. Invasive species not only threaten biodiversity and stability of native ecosystems but also potentially affect ecosystem functioning and processes. Therefore, there is an urgent need to understand how invasive species influence the tropical forest ecosystems in changing climate. Thus, an attempt has been made to understand the phenomenon of plant invasion that seems to be gradually taking over the Central Indian forests, especially the states of Madhya Pradesh and Chhattisgarh. The adoption of an ecosystem approach to tackle the spread of alien invasive species across Central India could be the most effective and economical approach, rather than trying to manage each and every species individually. Also, smart and novel approaches of putting the alien invasive species to use might help in better management. The present review could help forest officials and policy-makers to devise and implement forest conservation and restoration measures to minimize the impact of plant invasion on the fragile Central Indian forests. This review would also provide a conceptual basis for improving our general understanding on invasive species and their impact on the structural and functional aspects of tropical forest ecosystems.

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References

Adams VM, Petty AM, Douglas MM, Buckley YM, Ferdinands KB, Okazaki T, Ko DW, Setterfield SA (2015) Distribution demography and dispersal model of spatial spread of invasive plant populations with limited data. Method Ecol Evol 6(7):782–794

Adhikari D, Tiwary R, Barik SK (2015) Modelling hotspots for invasive alien plants in India. PLoS One 10(7):pe0134665

Agarwala M, DeFries RS, Qureshi Q, Jhala YV (2016a) Factors associated with long-term species composition in dry tropical forests of Central India. Environ Res Lett 11(10):p105008

Agarwala M, DeFries RS, Qureshi Q, Jhala YV (2016b) Changes in the dry tropical forests in Central India with human use. Reg Environ Change 16(1):5–15

Albrecht M, Padrón B, Bartomeus I, Traveset A (2014) Consequences of plant invasions on compartmentalization and species' roles in plant–pollinator networks. P Roy Soc Lond B Biol 281(1788):p20140773

Albrecht M, Ramis MR, Traveset A (2016) Pollinator-mediated impacts of alien invasive plants on the pollination of native plants: the role of spatial scale and distinct behaviour among pollinator guilds. Biol Invasions 18(7):1801–1812

Alemayehu K, Desalegn T (2016) Antibiotic effects of *Argemone mexicana* (Papaveraceae) against field crops and pathogens causing mastitis in dairy cattle in three districts of Amhara region Ethiopia. J Adv Biol Biotech 5(1):1–9

Allen JM, Bradley BA (2016) Out of the weeds? Reduced plant invasion risk with climate change in the continental United States. Biol Cons 203:306–312

Anonymous (2014) MoEF (Ministry of Environment Forests and Climate Change Government of India). Ecosystems Service Improvement Project (GEF Assisted) Environment and Social Management Framework and Tribal Development Plan, p 1–98

Ashton IW, Hyatt LA, Howe KM, Gurevitch J, Lerdau MT (2005) Invasive species accelerate decomposition and litter nitrogen loss in a mixed deciduous forest. Ecol Appl 15(4):263–1272

Aslan C, Rejmanek M (2012) Native fruit traits may mediate dispersal competition between native and non-native plants. NeoBiota 12:1–24

Aung T, Koike F (2015) Identification of invasion status using a habitat invasibility assessment model: the case of Prosopis species in the dry zone of Myanmar. J Arid Environ 120:87–94

Bais HP, Vepachedu R, Gilroy S, Callaway RM, Vivanco JM (2003) Allelopathy and exotic plants: from genes to invasion. Science 301:1377–1380

Baker HG (1955) Self-compatibility and establishment after long distance dispersal. Evolution 9:347–349

Banerjee AK, Dewanji A (2017a) Native exotic relationships in plant communities: the role of exotic dominance in framing community composition. Ecol Res 32(5):653–665

Banerjee AK, Dewanji A (2017b) Role of intraspecific trait plasticity in *Mikania micrantha* Kunth growth and impact of its abundance on community composition. J Asia-Pac Biodiv 10(2):237–249

Blackburn TM, Lockwood JL, Cassey P (2009) Avian invasions: the ecology and evolution of exotic birds, vol 1. Oxford University Press, Oxford

Blackburn TM, Pyšek P, Bacher S, Carlton JT, Duncan RP, Jarošík V, Wilson JR, Richardson DM (2011) A proposed unified framework for Biological Invasions. Trends Ecol Evol 26(7):333–339

Blumenthal D (2005) Interrelated causes of plant invasion. Science 310(5746):243–244

Bongard C (2012) A review of the influence of root-associating fungi and root exudates on the success of invasive plants. NeoBiota 14:21–45

Bradley BA, Oppenheimer M, Wilcove DS (2009) Climate change and plant invasions: restoration opportunities ahead? Glob Change Biol 15(6):1511–1521

Bradley BA, Wilcove DS, Oppenheimer M (2010) Climate change increases risk of plant invasion in the Eastern United States. Biol Invasions 12(6):1855–1872

Bruckman D, Campbell DR (2016) Timing of invasive pollen deposition influences pollen tube growth and seed set in a native plant. Biol Invasions 18(6):1701–1711

Buckley YM (2017) Invasion ecology: Unpredictable arms race in a jam jar. Nature Ecol Evol 1:p0028

Burns JH (2006) Relatedness and environment affect traits associated with invasive and noninvasive introduced Commelinaceae. Ecol Appl 16(4):1367–1376

Byers JE (2002) Impact of non-indigenous species on natives enhanced by anthropogenic alteration of selection regimes. Oikos 97(3):449–458

CABI's Invasive Species Compendium, http://wwwcabiorg/isc/

Cabra-Rivas I, Saldaña A, Castro-Díez P, Gallien L (2016) A multi-scale approach to identify invasion drivers and invaders' future dynamics. Biol Invasions 18(2):411–426

Callaway RM, Aschehoug ET (2000) Invasive plants versus their new and old neighbors: a mechanism for exotic invasion. Science 290(5491):521–523

Caplat P, Nathan R, Buckley YM (2012) Seed terminal velocity wind turbulence and demography drive the spread of an invasive tree in an analytical model. Ecology 93(2):368–377

Catford JA, Jansson R, Nilsson C (2009) Reducing redundancy in invasion ecology by integrating hypotheses into a single theoretical framework. Divers distributions 15(1):22–40

Catford JA, Baumgartner JB, Vesk PA, White M, Buckley YM, McCarthy MA (2016) Disentangling the four demographic dimensions of species invasiveness. J Ecol 104(6):1745–1758

Champion SH, Seth SK (1968) A revised survey of the forest types of India. Government of India Publication, New Delhi

Chapin FS, Zavaleta ES, Eviner VT, Naylor RL, Vitousek PM, Reynolds HL, Hooper DU, Lavorel S, Sala OE, Hobbie SE, Mack MC (2000) Consequences of changing biodiversity. Nature 405(6783):234–242

Chatterjee R (2015) Impact of *Lantana camara* in the Indian society. Int J Environ 4(2):348–354

Chaubey OP, Sharma A, Krishnamurthy G (2015) Plant diversity edaphic status and population structure in different forest types of Madhya Pradesh and Chhattisgarh states in India. Int J Biosci Biotech 7(2):115–124

Chauhan MS, Sharma A, Phartiyal B, Kumar K (2013) Holocene vegetation and climatic variations in central India: a study based on multiproxy evidences. J Asian Earth Sci 77:45–58

Chen BM, Li S, Liao HX, Peng SL (2017) Do forest soil microbes have the potential to resist plant invasion? A case study in Dinghushan Biosphere Reserve (South China). Acta Oecol 81:1–9

Chhattisgarh Forest Department, http://wwwcgforestcom/English/Introductionhtm

Chikuruwo C, Masocha M, Murwira A, Ndaimani H (2017) Predicting the suitable habitat of the invasive Xanthium strumarium L in southeastern Zimbabwe. Appl Ecol Environ Res 15(1):17–32

Clout MN, Williams PA (2009) Invasive species management: A handbook of principles and techniques. Oxford University Press, Oxford

Colautti RI, Bailey SA, Van Overdijk CD, Amundsen K, MacIsaac HJ (2006) Characterised and projected costs of nonindigenous species in Canada. Biol Invasions 8(1):45–59

Colautti R, Parker JD, Cadotte MW, Pyšek P, Brown CS, Sax D, Richardson D (2014) Quantifying the invasiveness of species. NeoBiota 21:7–27

Courchamp F, Fournier A, Bellard C, Bertelsmeier C, Bonnaud E, Jeschke JM, Russell JC (2017) Invasion biology: Specific problems and possible solutions. Trends Ecol Evol 32(1):13–22

Crooks JA (2005) Lag times and exotic species: the ecology and management of Biological Invasions in slow-motion. Ecoscience 12(3):316–329

Csecserits A, Botta-Dukát Z, Kröel-Dulay G, Lhotsky B, Ónodi G, Rédei T, Szitár K, Halassy M (2016) Tree plantations are hot-spots of plant invasion in a landscape with heterogeneous land-use. Agric Ecosyst Environ 226:88–98

Daehler CC (1998) The taxonomic distribution of invasive angiosperm plants: ecological insights and comparison to agricultural weeds. Biol Cons 84(2):167–180

D'Antonio CM, Meyerson LA (2002) Exotic plant species as problems and solutions in ecological restoration: a synthesis. Restor Ecol 10(4):703–713

D'Antonio CM, Vitousek PM (1992) Biological invasions by exotic grasses the grass/fire cycle and global change. Ann Rev Ecol Syst 23(1):63–87

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

Das M, Khanna SK (1997) Clinicoepidemiological, toxicological and safety evaluation studies on argemone oil. Crit Rev Toxicol 27(3):273–297

Dawson W, Moser D, van Kleunen M, Kreft H, Pergl J, Pysek P, Weigelt P, Winter M, Lenzner B, Blackburn TM, Dyer EE, Cassey P, Scrivens SL, Economo EP, Guénard B, Capinha C, Seebens H, García-Díaz P, Nentwig W, García-Berthou E, Casal C, Mandrak NE, Fuller P, Meyeer C, Essl F (2017) Global hotspots and correlates of alien species richness across taxonomic groups. Nature Ecol Evol 1:0186

Day M (2012) *Mikania micrantha* Kunth: mile-a-minute Biological Control of weeds in Australia, pp 368–372

Delivering Alien Invasive Species Inventories for Europe (DAISIE), http://wwweurope-aliensorg/

Deshmukh UB, Shende MB, Rathor OS (2015) Invasive Alien angiospermic plants from Chamorshi Tahsil of Gadchiroli District of Maharashtra, India. Int Res J Biol Sci 4(12):40–45

Dhileepan K, Senaratne KADW (2009) How widespread is *Parthenium hysterophorus* and its biological control agent *Zygogramma bicolorata* in South Asia? Weed Res 49(6):557–562

Dickie I, Bennett B, Burrows L, Nuñez M, Peltzer D, Porté A, Richardson DM, Rejmánek M, Rundel P, van Wilgen B (2014) Conflicting values: ecosystem services and invasive tree management. Biol Invasions 16:705–719

Didham RK, Tylianakis JM, Hutchison MA, Ewers RM, Gemmell NJ (2005) Are invasive species the drivers of ecological change? Trends Ecol Evol 20(9):470–474

Dodet M, Collet C (2012) When should exotic forest plantation tree species be considered as an invasive threat and how should we treat them? Biol Invasions 14(9):1765–1778

Dzikiti S, Ntshidi Z, Le Maitre DC, Bugan RDH, Mazvimavi D, Schachtschneider K, Jovanovic NZ, Pienaar HH (2017) Assessing water use by Prosopis invasions and Vachellia karroo trees: Implications for groundwater recovery following alien plant removal in an arid catchment in South Africa. Forest Ecol Manag 398:153–163

Earth's CO_2 https://wwwco2earth/

Egawa C (2017) Wind dispersal of alien plant species into remnant natural vegetation from adjacent agricultural fields. Global Ecol Cons 11:33–41

Ehler LE (1998) Invasion Biology and Biological Control. Biol Control 13(2):127–133

Ellstrand NC, Schierenbeck KA (2000) Hybridization as a stimulus for the evolution of invasiveness in plants? Proc Natl Acad Sci 97(13):7043–7050

Elton CS (1958) The ecology of invasions by animals and plants. Metheun, London

Essl F, Dullinger S, Rabitsch W, Hulme PE, Hülber K, Jarošík V, Kleinbauer I, Krausmann F, Kühn I, Nentwig W, Vilà M (2011) Socioeconomic legacy yields an invasion debt. Proc Natl Acad Sci 108(1):203–207

Essl F, Bacher S, Blackburn TM, Booy O, Brundu G, Brunel S, Cardoso AC, Eschen R, Gallardo B, Galil B, García-Berthou E (2015) Crossing frontiers in tackling pathways of Biological Invasions. BioScience 65(8):769–782

European Alien Species Information Network (EASIN), https://easinjrceceuropaeu/

Ewel JJ (1977) Differences between wet and dry successional tropical ecosystems. Geo Eco Trop 1(2):103–117

Feng YL, Fu GL, Zheng YL (2008) Specific leaf area relates to the differences in leaf construction cost photosynthesis nitrogen allocation and use efficiencies between invasive and noninvasive alien congeners. Planta 228(3):383–390

Franklin J (1995) Predictive vegetation mapping: geographic modelling of biospatial patterns in relation to environmental gradients. Prog Phys Geogr 19(4) pp474-499

Frehse FDA, Braga RR, Nocera GA, Vitule JRS (2016) Non-native species and invasion biology in a megadiverse country: scientometric analysis and ecological interactions in Brazil. Biol Invasions 18(12):3713–3725

Frezina NCA (2013) Assessment and utilization of water hyacinth in the water bodies of Tamil Nadu. Int J Sci Res Rev 2(1):58–77

FSI (2011) India State of Forest Report. Forest Survey of India, Dehradun

FSI (2015) India State of Forest Report. Forest Survey of India, Dehradun

Funk JL, Cleland EE, Suding KN, Zavaleta ES (2008) Restoration through reassembly: plant traits and invasion resistance. Trends Ecol Evol 23(12):695–703

Gaertner M, Nottebrock H, Fourie H, Privett SD, Richardson DM (2012) Plant invasions restoration and economics: perspectives from South African fynbos. Perspect Plant Ecol 14(5):341–353

Gallien L, Münkemüller T, Albert CH, Boulangeat I, Thuiller W (2010) Predicting potential distributions of invasive species: where to go from here? Divers Distributions 16(3):331–342

Ghazoul J, Sheil D (2010) Tropical rain forest ecology diversity and conservation. Oxford University Press

Global Invasive Alien Species Information Partnership Gateway, http://giasipartnershipmyspeciesinfo/en

Global Invasive Species Database (2017a) Species profile: Lantana camara http://wwwiucngisdorg/gisd/speciesname/Lantana+camara

Global Invasive Species Database (2017b), Species profile: Ageratum conyzoides http://wwwiucngisdorg/gisd/speciesname/Ageratum%20conyzoides

Global Invasive Species Database (GISD), http://wwwiucngisdorg/gisd/

Global Naturalized Alien Flora (GloNAF), https://glonaforg/

Global Register of Introduced and Invasive Species (GRIIS), http://wwwgriisorg/

Golivets M (2014) Ecological and biological determination of invasion success of non-native plant species in urban woodlands with special regard to short-lived monocarps. Urban Ecosyst 17(1):291–303

Gooden B, French K, Turner PJ (2009) Invasion and management of a woody plant Lantana camara L alters vegetation diversity within wet sclerophyll forest in southeastern Australia. Forest Ecol Manag 257(3):960–967

Gordon DR (1998) Effects of invasive non-indigenous plant species on ecosystem processes: lessons from Florida. Ecol Appl 8(4):975–989

Government of Madhya Pradesh, http://wwwmpgovin/en/web/guest/forest

Gray DR (2017) Climate change can reduce the risk of biological invasion by reducing propagule size. Biol Invasions 19(3):913–923

Gray AJ, Mack RN, Harper JL, Usher MB, Joysey K, Kornberg H (1986) Do invading species have definable genetic characteristics? Philos Trans R Soc Lond B Biol Sci 314(1167):655–674

Great Britain's Non-Native Species Information Portal (GBNNSIP), http://wwwnonnativespeciesorg/factsheet/

Groves RH, Lonsdale M, Boden R (2005) Jumping the garden fence: invasive garden plants in Australia and their environmental and agricultural impacts. WWF-Australia Ultimo

Guisan A, Thuiller W (2005) Predicting species distribution: offering more than simple habitat models. Ecol Lett 8(9):993–1009

Guisan A, Zimmermann NE (2000) Predictive habitat distribution models in ecology. Ecol Model 135(2):147–186

Guisan A, Tingley R, Baumgartner JB, Naujokaitis-Lewis I, Sutcliffe PR, Tulloch AI, Regan TJ, Brotons L, McDonald-Madden E, Mantyka-Pringle C, Martin TG, Rhodes JR, Maggini R, Setterfield SA, Elith J, Schwartz MW, Wintle BA, Broennimann O, Austin M, Ferrier S, Kearney MR, Possingham HP, Buckley YM (2013) Predicting species distributions for conservation decisions. Ecol Lett 16(12):1424–1435

Haeuser E, Dawson W, van Kleunen M (2017) The effects of climate warming and disturbance on the colonization potential of ornamental alien plant species. J Eecol https://doi.org/101111/1365-274512798

Hellmann JJ, Byers JE, Bierwagen BG, Dukes JS (2008) Five potential consequences of climate change for invasive species. Cons Biol 22(3):534–543

Hierro JL, Maron JL, Callaway RM (2005) A biogeographical approach to plant invasions: the importance of studying exotics in their introduced and native range. J Ecol 93(1):5–15

Hiremath AJ, Sundaram B (2005) The fire-lantana cycle hypothesis in Indian forests. Cons Soc 3(1):26–42

Hiremath AJ, Sundaram B (2013) Invasive plant species in Indian protected areas: conserving biodiversity in cultural landscapes. In: Foxcroft L, Pyšek P, Richardson D, Genovesi P (eds) Plant invasions in protected areas. Invading Nature – Springer series in invasion ecology, vol 7. Springer, Dordrecht, pp 241–266

Hobbs RJ (1989) The nature and effects of disturbance relative to invasions. Biological Invasions: a global perspective. In: Drake JA, Mooney HA, Di Castri F, Groves RH, Kruger FJ, Rejmanek M, Williamson M (eds) Biological invasions: A global perspective. John Wiley and Sons, New York, pp 389–405

Holm LG, Pancho JV, Herberger JP, Plucknett DL (1979) A geographical atlas of world weeds. Wiley, New York

Holm LG, Pancho JV, Herberger JP, Plucknett DL (1991) A geographical atlas of world weeds. In: Krieger Publishing Company. Malabar, Florida

Hui C, Richardson DM, Landi P, Minoarivelo HO, Garnas J, Roy HE (2016) Defining invasiveness and invasibility in ecological networks. Biol Invasions 18(4):971–983

Hulme PE (2008) Contrasting alien and native plant species–area relationships: the importance of spatial grain and extent. Glob Ecol Biogeogr 17(5):641–647

Hulme PE, Bacher S, Kenis M, Klotz S, Kühn I, Minchin D, Nentwig W, Olenin S, Panov V, Pergl J, Pyšek P (2008) Grasping at the routes of Biological Invasions: a framework for integrating pathways into policy. J Appl Ecol 45(2):403–414

Iannone BV, Potter KM, Guo Q, Liebhold AM, Pijanowski BC, Oswalt CM, Fei S (2016) Biological invasion hotspots: a trait-based perspective reveals new sub-continental patterns. Ecography 39(10):961–969

India Biodiversity Portal, *Xanthium strumarium* L. Species page https://indiabiodiversityorg/species/show/266687

Invasive Alien Plant Species in India (ENVIS Centre on Floral Diversity), http://wwwbsienvisnicin/Database/Invasive_Alien_species_15896aspx

Island Biodiversity and Invasive Species Database (IBIS), http://ibisfosaucklandacnz/page/invasive-speciesaspx

Ismail BS, Mah LS (1993) Effects of *Mikania micrantha* HBK on germination and growth of weed species. Plant Soil 157(1):107–113

Janzen DH (1988a) Management of habitat fragments in a tropical dry forest: growth. Ann Mo Bot Gard 75(1):105–116

Janzen DH (1988b) Tropical dry forests: The most endangered major tropical ecosystem. In: Wilson EO, Peters FM (eds) Biodiversity. National Academy Press, Washington DC, pp 130–144

Jauni M, Hyvönen T (2010) Invasion level of alien plants in semi-natural agricultural habitats in boreal region. Agric Ecosyst Environ 138(1):109–115

Jeschke J, Aparicio LG, Haider S, Heger T, Lortie C, Pyšek P, Strayer D (2012) Support for major hypotheses in invasion biology is uneven and declining. NeoBiota 14:1–20

Johnson PT, Olden JD, Solomon CT, Van der Zanden MJ (2009) Interactions among invaders: community and ecosystem effects of multiple invasive species in an experimental aquatic system. Oecologia 159(1):161–170

Joshi PK, Kumar M, Paliwal A, Midha N, Dash PP (2009) Assessing impact of industrialization in terms of LULC in a dry tropical region (Chhattisgarh), India using remote sensing data and GIS over a period of 30 years. Environ Monit Assess 149(1):371–376

Kamble RB, Hate SD, Chaturvedi A (2013) New additions to the Flora of Nagpur District, Maharashtra. J New Biol Rep 2(1):9–13

Kannan R, Shackleton CM, Shaanker RU (2013) Playing with the forest: invasive alien plants policy and protected areas in India. Curr Sci 104(9):1159–1165

Karlsson LM, Tamado T, Milberg P (2003) Seed dormancy pattern of the annuals *Argemone ochroleuca* and *A. mexicana* (Papaveraceae). Flora Morph Distrib Funct Ecol Plants 198(4):329–339

Katz DS, Ibáñez I (2016) Biotic interactions with natural enemies do not affect potential range expansion of three invasive plants in response to climate change. Biol Invasions 18(11):3351–3363

Kaur B, Bhatia S, Sharma KK (2014) Diversity and impact of invasive alien plant species of family Asteraceae in Jammu district (Jammu and Kashmir India). Int J Interdis Multidis Stud 1(8):51–62

Keane RM, Crawley MJ (2002) Exotic plant invasions and the enemy release hypothesis. Trends Ecol Evol 17(4):164–170

Kgope BS, Bond WJ, Midgley GF (2010) Growth responses of African savanna trees implicate atmospheric [CO_2] as a driver of past and current changes in savanna tree cover. Austral Ecol 35(4):451–463

Khuroo AA, Rashid I, Reshi Z, Dar GH, Wafai BA (2007) The alien flora of Kashmir Himalaya. Biol Invasions 9(3):269–292

Khuroo AA, Reshi ZA, Malik AH, Weber E, Rashid I, Dar GH (2012) Alien flora of India: taxonomic composition invasion status and biogeographic affiliations. Biol Invasions 14(1):99–113

Kim CG, Kil J (2016) Alien flora of the Korean Peninsula. Biol Invasions 18(7):1843–1852

van Kleunen M, Dawson W, Essl F, Pergl J, Winter M, Weber E, Kreft H, Weigelt P, Kartesz J, Nishino M, Antonova LA, Barcelona JF, Cabezas FJ, Cardenas D Cardenas-Toro J, Castano N, Chacon E, Chatelain C, Ebel AL, Figueiredo E, Fuentes N, Froom QJ, Henderson L, Inderjit, Kupriyanov A, Masciadri S, Meerman J, Morozova O, Moser D, Nickrent DL, Patzelt A, Pelser PB, Baptiste MP, Poopath M, Schulze M, Seebens H, Shu WS, Thomas J, Velayos M, Wieringa JJ, Pysek P (2015) Global exchange and accumulation of non-native plants. Nature 525(7567):100-103

Knox J, Jaggi D, Paul MS (2011) Population dynamics of *Parthenium hysterophorus* (Asteraceae) and its biological suppression through *Cassia occidentalis* (Caesalpiniaceae). Turk J Bot 35(2):111–119

Kohli RK, Batish DR, Singh HP, Dogra KS (2006) Status invasiveness and environmental threats of three tropical American invasive weeds (*Parthenium hysterophorus* L *Ageratum conyzoides* L *Lantana camara* L) in India. Biol Invasions 8(7):1501–1510

Kohli RK, Batish DR, Singh JS, Singh HP, Bhatt JR, Singh SP, Tripathi RS (2012) Plant invasion in India: An overview. In: Bhatt JR, Singh JS, Singh SP, Tripathi RS, Kohli RK (eds) Invasive alien plants: An ecological appraisal for the Indian subcontinent. CAB International, pp p1–p9

Kuebbing SE, Nuñez MA (2016) Invasive non-native plants have a greater effect on neighbouring natives than other non-natives. Nature plants 2:p16134

Kuebbing SE, Nuñez MA, Simberloff D (2013) Current mismatch between research and conservation efforts: the need to study co-occurring invasive plant species. Biol Cons 160:121–129

Kueffer C, Kull CA (2017) Non-native species and the aesthetics of nature. In: Vilà M, Hulme P (eds) Impact of biological invasions on ecosystem services. Invading Nature - Springer series in invasion ecology, vol 12. Springer, Cham, pp 311–324

Kueppers LM, Snyder MA, Sloan LC, Zavaleta ES, Fulfrost B (2005) Modeled regional climate change and California endemic oak ranges. Proc Natl Acad Sci USA 102:16281–16286

Kull CA, Shackleton CM, Cunningham PJ, Ducatillon C, Dufour-Dror JM, Esler KJ, Friday JB, Gouveia AC, Griffin AR, Marchante E, Midgley SJ, Pauchard A, Rangan H, Richardson DM, Rinaudo T, Tassin J, Urgenson LS, von Maltitz GP, Zenni RD, Zylstra MJ (2011) Adoption use and perception of Australian acacias around the world. Divers distributions 17(5):822–836

Kumari P, Choudhary AK (2016) Exotic species invasion threats to forests: A case study from the Betla national park, Palamu, Jharkhand, India. Trop Plant Res 3(3):592–599

Kumari P, Sahu PK, Soni MY, Awasthi P (2014) Impact of *Parthenium hysterophorus* L invasion on species diversity of cultivated fields of Bilaspur (CG). India. Agric Sci 5(8):754–764

Kushwah RBS, Kumar V (2002) Status of flora in protected areas: the case studies of eight Pas of Madhya Pradesh (India). Ind For 128(3):271–288

Küster EC, Kühn I, Bruelheide H, Klotz S (2008) Trait interactions help explain plant invasion success in the German flora. J Ecol 96(5):860–868

Lal C, Singh L, Attri V, Sarvade S (2015) Tree species diversity distribution and population structure in a tropical dry deciduous forests of Chhatisgarh India. J Appl Nat Sci 7(2):681–685

Larson DL, Anderson PJ, Newton W (2001) Alien plant invasion in mixed-grass prairie: effects of vegetation type and anthropogenic disturbance. Ecol Appl 11(1):128–141

Lerdau M, Whitbeck J, Holbrook NM (1991) Tropical deciduous forest: death of a biome. Trends Ecol Evol 6(7):201–202

Levine JM, D'Antonio CM (1999) Elton revisited: a review of evidence linking diversity and invasibility. Oikos 87:15–26

Levine JM, Adler PB, Yelenik SG (2004) A meta-analysis of biotic resistance to exotic plant invasions. Ecol Lett 7(10):975–989

Lewis SL, Maslin MA (2015) Defining the anthropocene. Nature 519(7542):171–180

Li B, Xu B, Chen J (2001) Perspectives on general trends of plant invasions with special reference to alien weed flora of Shanghai. Biodiv Sci 9(4):446–457

Lindemann-Matthies P (2016) Beasts or beauties? Laypersons' perception of invasive alien plant species in Switzerland and attitudes towards their management. NeoBiota 29:15–33

Loarie SR, Carter BE, Hayhoe K, McMahon S, Moe R, Knight CA, Ackerly DD (2008) Climate Change and the Future of California's. Endemic Flora PLoS One 3(6):e2502. https://doi.org/101371/journalpone0002502

Lockwood JL, Cassey P, Blackburn TM (2009) The more you introduce the more you get: the role of colonization pressure and propagule pressure in invasion ecology. Divers distributions 15(5):904–910

Lockwood JL, Hoopes MF, Marchetti MP (2013) Invasion ecology. Wiley-Blackwell, West Sussex

Loo SE, Mac Nally R, O'Dowd DJ, Lake PS (2009) Secondary invasions: implications of riparian restoration for in-stream invasion by an aquatic grass. Restor Ecol 17(3):378–385

Louman B, DeClerck F, Ellatifi M, Finegan B, Thompson I (2010) Forest biodiversity and ecosystem services: drivers of change responses and challenges. IUFRO (International Union of Forestry Research Organizations) Secretariat 25:95–112

Lugo AE (2013) Novel tropical forests: Nature's response to global change. Trop Cons Sci Spl Issue 6(3):325–337

MacArthur R (1970) Species packing and competitive equilibrium for many species. Theor Popul Biol 1(1):1–11

MacDougall AS, Turkington R (2005) Are invasive species the drivers or passengers of change in degraded ecosystems? Ecology 86(1):42–55

Mack RN, Simberloff D, Lonsdale WM, Evans H, Clout M, Bazzaz FA (2000) Biotic invasions: causes epidemiology global consequences and control. Ecol Appl 10(3):689–710

Madhya Pradesh Forest Department, http://wwwmpforestorg/foresthtml

Marchante E, Kjøller A, Struwe S, Freitas H (2009) Soil recovery after removal of the N2-fixing invasive *Acacia longifolia*: consequences for ecosystem restoration. Biol Invasions 11(4):813–823

Martínez OJA (2009) Ecology of novel forests dominated by the African tulip tree (*Spathodea campanulata* Beauv) in northcentral Puerto Rico. Masters thesis, University of Puerto Rico, Río Piedras, Puerto Rico.

Martínez OJA (2010) Invasion by native tree species prevents biotic homogenization in novel forests of Puerto Rico. Plant Ecol 211:49–64

Martínez OJA, Lugo AE (2008) Post sugar cane succession in moist alluvial sites in Puerto Rico. In: Myster RW (ed) Post-agricultural succession in the Neotropics. Springer, New York, pp 73–92

Martínez OJA, Rodríguez MA, Rosario I, Soto N, López A, Lugo AE (2010) Structure and species composition of novel forests dominated by an introduced species in northcentral Puerto Rico. New Forest 39:1–18

Maskell LC, Firbank LG, Thompson K, Bullock JM, Smart SM (2006) Interactions between non-native plant species and the floristic composition of common habitats. J Ecol 94(6):1052–1060

Mazzolari AC, Marrero HJ, Vázquez DP (2017) Potential contribution to the invasion process of different reproductive strategies of two invasive roses. Biol Invasions 19(2):615–623

McDougall KL, Khuroo AA, Loope LL, Parks CG, Pauchard A, Reshi ZA, Rushworth I, Kueffer C (2011) Plant invasions in mountains: global lessons for better management. Mountain Res Develop 31(4):380–387

McGeoch MA, Genovesi P, Bellingham PJ, Costello MJ, McGrannachan C, Sheppard A (2016) Prioritizing species pathways and sites to achieve conservation targets for biological invasion. Biol Invasions 18(2):299–314

Millennium Ecosystem Assessment (2005) Ecosystems and human well-being: biodiversity synthesis. World Resources Institute, Washington DC

Misra SS, Sushi SN, Rajak DC (2012) Biocontrol strategy for major invasive weeds of neotropical origin in India. Bioherald: Int J Biodiv Environ 2(2):134–142

Mittermeier RA, Myers N, Mittermeier CG, Robles G (1999) Hotspots: Earth's biologically richest and most endangered terrestrial ecoregions. Cemex, Conservation International. The University of Chicago Press, Chicago, p 392

Monty A, Bizoux JP, Escarré J, Mahy G (2013) Rapid plant invasion in distinct climates involves different sources of phenotypic variation. PLoS One 8(1):pe55627

Morris RJ (2010) Anthropogenic impacts on tropical forest biodiversity: a network structure and ecosystem functioning perspective. Philos Trans R Soc Lond B Biol Sci 365(1558):3709–3718

Motloung R, Robertson M, Rouget M, Wilson J (2014) Forestry trial data can be used to evaluate climate-based species distribution models in predicting tree invasions. NeoBiota 20:31–48

Mudgal V, Khanna KK, Hazra PK (1997) Flora of Madhya Pradesh II. Botanical Survey of India, pp 403-404

Mujaffar S, Shukla SK, Mishra S (2013) Some angiospermic plants new to Central India. Sci Res Rep 3(2):102–105

Murphy PG, Lugo AE (1986) Ecology of tropical dry forest. Annu Rev Ecol Evol Syst 17(1):67–88

Murray B, Phillips M (2012) Temporal introduction patterns of invasive alien plant species to Australia. NeoBiota 13:1–14

Murray BR, Martin LJ, Phillips ML, Pyšek P (2017) Taxonomic perils and pitfalls of dataset assembly in ecology: a case study of the naturalized Asteraceae in Australia. NeoBiota 34:1–20

Myers RL (1983) Site susceptibility to invasion by the exotic tree *Melaleuca quinquenervia* in southern Florida. J Appl Ecol 20(2):645–658

Nagel JM, Huxman TE, Griffin KL, Smith SD (2004) CO_2 enrichment reduces the energetic cost of biomass construction in an invasive desert grass. Ecology 85(1):100–106

Namkeleja HS, Tarimo MT, Ndakidemi PA (2014) Allelopathic effects of *Argemone mexicana* to growth of native plant species. Am J Plant Sci 5:1336–1344

Nayar MP (1977) Changing patterns of the Indian flora. Nelumbo 19(1-4):145–155

Ning L, Yu FH, van Kleunen M (2016) Allelopathy of a native grassland community as a potential mechanism of resistance against invasion by introduced plants. Biol Invasions 18(12):3481–3493

Oduor AM, Leimu R, Kleunen M (2016) Invasive plant species are locally adapted just as frequently and at least as strongly as native plant species. J Ecol 104(4):957–968

Ööpik M, Bunce RGB, Tischler M (2013) Horticultural markets promote alien species invasions: an Estonian case study of herbaceous perennials. NeoBiota 17:19–37

Padalia H, Bahuguna U (2017) Spatial modelling of congruence of native biodiversity and potential hotspots of forest invasive species (FIS) in central Indian landscape. J Nat Cons 36:29–37

Padalia H, Kudrat M, Sharma KP (2013) Mapping sub-pixel occurrence of an alien invasive *Hyptis suaveolens* (L.) Poit using spectral unmixing technique. Int J Remote Sens 34(1):325–340

Padalia H, Srivastava V, Kushwaha SPS (2014) Modeling potential invasion range of alien invasive species *Hyptis suaveolens* (L) Poit in India: Comparison of MaxEnt and GARP. Ecol Inform 22:36–43

Pande PK (2001) Structures of the tropical dry deciduous teak (*Tectona grandis*) forests of Satpura plateau (India) with special emphasis on regeneration and disturbance. J Trop For Sci 13(2):329–343

Pande PK (2005) Biomass and productivity in some disturbed tropical dry deciduous teak forests of Satpura plateau, Madhya Pradesh. Trop Ecol 46(2):229–240

Pandey DS (2000) Exotics–introduced and natural immigrants weeds cultivated etc. In: Singh NP, Singh DK, Hajra PK, Sharma BD (eds) Flora of India (introductory volume Part II). Botanical Survey of India, Kolkata, pp 266–301

Pandey HP, Chauhan SK (2012) *Lantana camara*: a journey from eradication to adaptive management. Bioherald: Int J Biodiv Environ 2(2):99–109

Parmesan C, Yohe G (2003) A globally coherent fingerprint of climate change impacts across natural systems. Nature 421(6918):37–42

Pasiecznik NM, Felker P, Harris PJ, Harsh L, Cruz G, Tewari JC, Cadoret K, Maldonado LJ (2001) The *'Prosopis Juliflora'*-*'Prosopis Pallida'* Complex: A Monograph. (Vol 172), Coventry UK

Patel DK (2012) Vegetation structure and composition in Guru Ghasidas Vishwavidyalaya in Central India. Int J Biodiv Cons 4(15):621–632

Pearson RG, Dawson TP (2003) Predicting the impacts of climate change on the distribution of species: are bioclimate envelope models useful? Glob Ecol Biogeogr 12(5):361–371

Pearson DE, Ortega YK, Runyon JB, Butler JL (2016) Secondary invasion: the bane of weed management. Biol Cons 197:8–17

Peerzada N (1997) Chemical composition of the essential oil of *Hyptis suaveolens*. Molecules 2(11):165–168

Peh KSH, Balmford A, Birch JC, Brown C, Butchart SH, Daley J, Dawson J, Gray G, Hughes FM, Mendes S, Millett J, Stattersfield AJ, Thomas DHL, Walpole M, Bradbury RB (2015) Potential impact of invasive alien species on ecosystem services provided by a tropical forested ecosystem: a case study from Montserrat. Biol Invasions 17(1):461–475

Penk MR, Jeschke JM, Minchin D, Donohue I (2016) Warming can enhance invasion success through asymmetries in energetic performance. J Anim Ecol 85(2):419–426

Pergl J, Pyšek P, Bacher S, Essl F, Genovesi P, Harrower CA, Hulme PE, Jeschke JE, Kenis M, Kühn I, Perglová I (2017) Troubling travellers: are ecologically harmful alien species associated with particular introduction pathways? NeoBiota 32:1–20

Perrings C, Mooney H, Williamson M (2010) Bio-invasions and globalization: ecology economics management and policy. Oxford University Press, Oxford, p 288

Peterson AT, Soberón J, Sánchez-Cordero V (1999) Conservatism of ecological niches in evolutionary time. Science 285(5431):1265–1267

Pimentel D (2002) Biological invasions: Economic and environmental costs of alien plant animal and microbe species. CRC Press, Boca Raton, p 384. https://doi.org/101201/9781420041668

Powell KI, Chase JM, Knight TM (2011) A synthesis of plant invasion effects on biodiversity across spatial scales. Am J Bot 98(3):539–548

Priya ES, Selvan PS (2017) Water hyacinth (*Eichhornia crassipes*)–An efficient and economic adsorbent for textile effluent treatment–A review. Arab J Chem 10:S3548–S3558

Pyšek P (1998) Is there a taxonomic pattern to plant invasions? Oikos 82:282–294

Pyšek P, Richardson D (2007) Traits associated with invasiveness in alien plants: where do we stand? In: Nentwig W (ed) Biological Invasions. Ecological Studies (Analysis and Synthesis), vol 193. Springer, Berlin/Heidelberg, pp 97–125

Pyšek P, Richardson DM (2008) Invasive plants. In: Jorgensen SE, Brian DF (eds) Ecological engineering. Vol [3] of Encyclopedia of ecology, vol 5, pp 2011–2020

Quamar MF, Chauhan MS (2011) Late holocene vegetation climate change and human impact in southwestern Madhya Pradesh. India. Palaeobotanist 60(2):281–289

Raghubanshi AS, Tripathi A (2009) Effect of disturbance habitat fragmentation and alien invasive plants on floral diversity in dry tropical forests of Vindhyan highland: a review. Trop Ecol 50(1):57–69

Raizada P, Singh A, Raghubanshi AS (2009) Comparative response of seedlings of selected native dry tropical and alien invasive species to CO_2 enrichment. J Plant Ecol 2(2):69–75

Rajashekar Y, Ravindra KV, Bakthavatsalam N (2014) Leaves of *Lantana camara* Linn (Verbenaceae) as a potential insecticide for the management of three species of stored grain insect pests. J Food Sci Tech 51(11):3494–3499

Ramaswami G, Sukumar R (2013) Long-term environmental correlates of invasion by *Lantana camara* (Verbenaceae) in a seasonally dry tropical forest. PLoS One 8(10):pe76995

Rao RR, Murugan R (2006) Impact of exotic adventives weeds on native biodiversity in India: implications for conservation. In: Rai LC, Gaur JP (eds) Invasive alien species and biodiversity in India. Banaras Hindu University, Varanasi, pp 93–109

Rao RR, Sagar K, Sathyanarayana N (2012) Resource utilization and beneficial aspects of invasive alien weeds with special reference to the Western Ghats, India. In: Bhatt JR, Singh JS, Singh SP, Tripathi RS, Kohli RK (eds) Invasive alien plants: An ecological appraisal for the Indian subcontinent. CAB International, Wallingford, pp 271–281

Ray S, Sainkhediya J (2014) Some new record for the flora of Madhya Pradesh. Biosci Discov 5(2):187–192

Razanajatovo M, Maurel N, Dawson W, Essl F, Kreft H, Pergl J, Pyšek P, Weigelt P, Winter M, van Kleunen M (2016) Plants capable of selfing are more likely to become naturalized. Nat Commun 7:p13313

Reddy CS (2012) Assessment of plant invasions across different habitats of India. Bioherald: Int J Biodiv Environ 2(2):110–125

Reichard S (2011) Horticulture. In: Simberloff D, Rejmanek M (eds) Encyclopedia of Biological Invasions. University of California Press, Berkeley, pp 336–342

Rejmanek M, Richardson DM (1996) What attributes make some plant species more invasive? Ecology 77:1655–1661

Reshi MI, Chadhar BL, Khare PK (2017) Alien invasive plants of Central Indian tropical dry deciduous forests of Sagar district, Madhya Pradesh, India. Indian For 143(2):157–164

Richardson DM (1998) Forestry trees as invasive aliens. Cons Biol 12(1):18–26

Richardson DM, Pyšek P (2012) Naturalization of introduced plants: ecological drivers of biogeographical patterns. New Phytol 196(2):383–396

Richardson DM, Rejmánek M (2011) Trees and shrubs as invasive alien species–a global review. Divers Distributions 17(5):788–809

Richardson DM, Pyšek P, Rejmanek M, Barbour MG, Panetta FD, West CJ (2000) Naturalization and invasion of alien plants: concepts and definitions. Divers Distributions 6(2):93–107

Rinawati F, Stein K, Lindner A (2013) Climate change impacts on biodiversity – the setting of a lingering global crisis. Diversity 5(1):114–123

Rodger JG, Johnson SD (2013) Self-pollination and inbreeding depression in *Acacia dealbata*: Can selfing promote invasion in trees? S Afr J Bot 88:252–259

Sagar R, Singh JS (2005) Structure, diversity and regeneration of tropical dry deciduous forest of northern India. Biodiversity Conserv 14(4):935–959

Saha S (2002) Anthropogenic fire regime in a deciduous forest of central India. Curr Sci 82(9):1144–1147

Sahu PK, Gupta S (2014) Medicinal plants of morning glory: Convolvulaceae Juss of Central India (Madhya Pradesh and Chhattisgarh). Biolife 2(2):463–469

Sahu PK, Singh JS (2008) Structural attributes of lantana-invaded forest plots in Achanakmar–Amarkantak Biosphere Reserve, Central India. Curr Sci 94(4):494–500

Sahu PK, Sagar R, Singh JS (2008) Tropical forest structure and diversity in relation to altitude and disturbance in a Biosphere Reserve in central India. Appl Veg Sci 11(4):461–470

Sahu KP, Urmalia R, Mashi SK, Tiwari V (2012) Contribution to the flora of Umaria district of MP. J Eco Tax Bot 36:261–274

Saikia P, Kumar A, Khan ML (2016) Biodiversity status and climate change scenario in Northeast India. In: Nautiyal S, Schaldach R, Raju K, Kaechele H, Pritchard B, Rao K (eds) Climate change challenge (3C) and social-economic-ecological interface-building. Environmental science and engineering. Springer, Cham

Sakai AK, Allendorf FW, Holt JS, Lodge DM, Molofsky J, With KA, Baughman S, Cabin RJ, Cohen JE, Ellstrand NC, McCauley DE (2001) The population biology of invasive species. Annu Rev Ecol Evol Syst 32(1):305–332

Salunkhe O, Khare PK, Sahu TR, Singh S (2016) Estimation of tree biomass reserves in tropical deciduous forests of Central India by non-destructive approach. Trop Ecol 57(2):153–161

Sanderson LA, Antunes PM (2013) The exotic invasive plant *Vincetoxicum rossicum* is a strong competitor even outside its current realized climatic temperature range. NeoBiota 16:1–15

Sandilyan S, van't Klooster CI (2016) The other sides of invasive alien plants of India—With special reference to medicinal values. J Nat Cons 31:16–21

Sandya GS, Ahirwar RK (2015) Ethnobotanical studies of some wild food plants of District Umaria, Central India. Int J Sci Res 4(4):432–435

Saul WC, Roy HE, Booy O, Carnevali L, Chen HJ, Genovesi P, Harrower CA, Hulme PE, Pagad S, Pergl J, Jeschke JM (2017) Assessing patterns in introduction pathways of alien species by linking major invasion data bases. J Appl Ecol 54(2):657–669

Seebens H, Gastner MT, Blasius B (2013) The risk of marine bioinvasion caused by global shipping. Ecol Lett 16(6):782–790

Seebens H, Essl F, Dawson W, Fuentes N, Moser D, Pergl J, Pyšek P, Kleunen M, Weber E, Winter M, Blasius B (2015) Global trade will accelerate plant invasions in emerging economies under climate change. Glob Change Biol 21(11):4128–4140

Seebens H, Essl F, Blasius B (2017a) The intermediate distance hypothesis of Biological Invasions. Ecol Lett 20(2):158–165

Seebens H, Blackburn TM, Dyer EE, Genovesi P, Hulme PE, Jeschke JM, Pagad S, Pyšek P, Winter M, Arianoutsou M, Bacher S (2017b) No saturation in the accumulation of alien species worldwide. Nat Commun 8:14435

Seifu A, Seboka N, Misganaw M, Bekele T, Merawi E, Ayenew A, Faris G (2017) Impact of invasive alien plant *Xanthium strumarium* on species diversity and composition of invaded plant communities in Borena Zone. Ethiopia. Biodiv Int J 1(1):00004

Sekar KC (2012) Invasive alien plants of Indian Himalayan region – diversity and implication. Am J Plant Sci 3(2):177–184

Shabbir A, Bajwa R (2007) *Parthenium* invasion in Pakistan – A threat still unrecognized. Pak J Bot 39(7):2519–2526

Shackleton RT, Le Maitre DC, Pasiecznik NM, Richardson DM (2014) *Prosopis*: a global assessment of the biogeography benefits impacts and management of one of the world's worst woody invasive plant taxa. AoB Plants 6:plu027

Shackleton RT, Le Maitre DC, Richardson DM (2015) *Prosopis* invasions in South Africa: Population structures and impacts on native tree population stability. J Arid Environ 114:70–78

Shackleton CM, Ruwanza S, Sanni GS, Bennett S, De Lacy P, Modipa R, Mtati N, Sachikonye M, Thondhlana G (2016) Unpacking Pandora's box: understanding and categorising ecosystem disservices for environmental management and human wellbeing. Ecosystems 19(4):587–600

Sharma GP, Raghubanshi AS (2006) Tree population structure regeneration and expected future composition at different levels of *Lantana camara* L. invasion in the Vindhyan tropical dry deciduous forest of India. Lyonia 11(1):27–39

Sharma GP, Singh JS, Raghubanshi AS (2005) Plant invasions: emerging trends and future implications. Curr Sci 88(5):726–734

Sharma GP, Raizada P, Raghubanshi AS (2009) *Hyptis suaveolens*: an emerging invader of Vindhyan plateau India. Weed Biol Manag 9(3):185–191

Sharma A, Batish DR, Singh HP, Jaryan V, Kohli RK (2017) The impact of invasive *Hyptis suaveolens* on the floristic composition of the periurban ecosystems of Chandigarh, northwestern India. Flora 233:156–162

Shen XY, Peng SL, Chen BM, Pang JX, Chen LY, Xu HM, Hou YP (2011) Do higher resource capture ability and utilization efficiency facilitate the successful invasion of native plants? Biol Invasions 13(4):869–881

Shiferaw H, Teketay D, Nemomissa S, Assefa F (2004) Some biological characteristics that foster the invasion of *Prosopis juliflora* (Sw) DC at Middle Awash Rift Valley Area, north-eastern Ethiopia. J Arid Environ 58(2):135–154

Shmueli G (2010) To explain or to predict? Stat Sci 25(3):289–310

Shukla BK, Sinha GP (2012) An inventory of invasive alien species of Chhattisgarh India. Bioherald: Int J Biodiv Environ 2(2):126–133

Shukla AN, Singh KP, Singh JS (2009) Invasive alien species of Achanakmar-Amarkantak Biosphere Reserve Central India. Proc Natl Acad Sci India B Biol 79(4):384–392

Simberloff D, Rejmánek M (eds) (2011) Encyclopedia of Biological Invasions. University of California Press, Berkeley

Simberloff D, Von Holle B (1999) Positive interactions of nonindigenous species: invasional meltdown? Biol Invasions 1(1):21–32

Simberloff D, Martin JL, Genovesi P, Maris V, Wardle DA, Aronson J, Courchamp F, Galil B, García-Berthou E, Pascal M, Pyšek P (2013) Impacts of biological invasions: what's what and the way forward. Trends Ecol Evol 28(1):58–66

Singh KP, Singh JS (1988) Certain structural and functional aspects of dry tropical forest and savanna. Int J Ecol Environ Sci 14:31–45

Singh BP, Upadhyay R (2014) Medicinal pteridophytes of Madhya Pradesh. J Pharmacogn Phytochem 3(3):173–176

Singh JP, Singh SP, Gupta SR (2006) Ecology environment and resource conservation. Anamaya Publishers, New Delhi

Singh KP, Shukla AN, Singh JS (2010) State-level inventory of invasive alien plants their source regions and use potential. Curr Sci 99(1):107–114

Sinha MK, Sinha D (2013) Composition of forest vegetation of Koria district Chhattisgarh (India). Glob J Biol Agric Health Sci 2(4):160–168

Sitzia T, Campagnaro T, Kowarik I, Trentanovi G (2016) Using forest management to control invasive alien species: helping implement the new European regulation on invasive alien species. Biol Invasions 18(1):1–7

Sol D, Maspons J, Vall-llosera M, Bartomeus I, García-Peña GE, Piñol J, Freckleton RP (2012) Unraveling the life history of successful invaders. Science 337:580–583

Soni MY, Sahu PK, Kumari P, Somi I, Awasthi P (2014) Impact of *Lantana camara* invasion on species diversity and regeneration status of tropical forest in Bilaspur (CG). Int J Biol Sci 1(3):42–52

Southern African Plant Invaders Atlas (SAPIA), http://wwwinvasivesorgza/

Srivastava S, Dvivedi A, Shukla RP (2014) Invasive alien species of terrestrial vegetation of north-eastern Uttar Pradesh. Int J Forest Res 2014

Steffen W, Grinevald J, Crutzen P, McNeill J (2011) The Anthropocene: conceptual and historical perspectives. Philos Trans A Math Phys Eng Sci 369(1938):842–867

Stinson KA, Campbell SA, Powell JR, Wolfe BE, Callaway RM, Thelen GC, Hallett SG, Prati D, Klironomos JN (2006) Invasive plant suppresses the growth of native tree seedlings by disrupting belowground mutualisms. PLoS Biol 4(5):140

Stout JC, Tiedeken EJ (2017) Direct interactions between invasive plants and native pollinators: evidence impacts and approaches. Funct Ecol 31:38–46

Strayer DL, Eviner VT, Jeschke JM, Pace ML (2006) Understanding the long-term effects of species invasions. Trends Ecol Evol 21(11):645–651

Sullivan RG, Clark M (2007) Can biodiversity survive global warming? Chicago Wilderness Journal 5(1):2–13

Suman NR, Khare PK, Salunkhe O, Chadhar B (2017) Alien angiospermic plants of Panna tiger reserve, Madhya Pradesh, India. Indian For 143(1):19–24

Sundaram B, Hiremath AJ (2012) *Lantana camara* invasion in a heterogeneous landscape: patterns of spread and correlation with changes in native vegetation. Biol Invasions 14(6):1127–1141

Sundarapandian SM, Subashree K (2017) Status of invasive plants in Tamil Nadu India: their impact and significance. In: Ansari AA, Gill SS, Abbas ZK, Naeem M (eds) Plant biodiversity: monitoring assessment and conservation. CAB International, Wallingford, pp 371–387

Sundarapandian SM, Muthumperumal C, Subashree K (2015) Biological invasion of vines their impacts and management. In: Parthasarathy N (ed) Biodiversity of Lianas. Springer, Cham, pp 211–253

Swamy PS, Ramakrishnan PS (1987) Effect of fire on population dynamics of *Mikania micrantha* HBK during early succession after slash-and-burn agriculture (jhum) in northeastern India. Weed Res 27(6):397–403

Téllez TR, López EMDR, Granado GL, Pérez EA, López RM, Guzmán JMS (2008) The water hyacinth *Eichhornia crassipes*: an invasive plant in the Guadiana River Basin (Spain). Aquat Invasions 3(1):42–53

Thakur AS (2015a) Phytosociological analysis of Gopalpura forest in Sagar district, MP. Scholars Impact 1(2):8–15

Thakur AS (2015b) Floristic composition life-forms and biological spectrum of tropical dry deciduous forest in Sagar District, Madhya Pradesh, India. Trop Plant Res 2(2):112–119

Thakur AS, Khare PK (2006) Species diversity and dominance in tropical dry deciduous forest ecosystem. J Environ Res Develop 1(1):26–31

The Directorate of Plant Protection Quarantine and Storage, http://ppqsgovin/

Theoharides KA, Dukes JS (2007) Plant invasion across space and time: factors affecting nonindigenous species success during four stages of invasion. New Phytol 176(2):256–273

Tireman H (1916) Lantana in the forests of Coorg. Indian For 42:384–392

Traveset A, Richardson DM (2006) Biological Invasions as disruptors of plant reproductive mutualisms. Trends Ecol Evol 21(4):208–216

Tripathi RS, Khan ML, Yadav AS (2012) Biology of *Mikania micrantha* HBK: a Review. In: Bhatt JR, Singh JS, Singh SP, Tripathi RS, Kohli RK (eds) Invasive alien plants: An ecological appraisal for the Indian subcontinent. CAB International, Wallingford, pp 99–107

TRY Plant Trait Database, https://wwwtry-dborg/TryWeb/Homephp

Uyi OO, Ekhator F, Ikuenobe CE, Borokini TI, Aigbokhan EI, Egbon IN, Adebayo AR, Igbinosa IB, Okeke CO, Igbinosa EO, Omokhua GA (2014) *Chromolaena odorata* invasion in Nigeria: A case for coordinated biological control. Manag Biol Invasion 5(4):377–393

van Kleunen M, Weber E, Fischer M (2010) A meta-analysis of trait differences between invasive and non-invasive plant species. Ecol Lett 13(2):235–245

Vaz AS, Kueffer C, Kull CA, Richardson DM, Vicente JR, Kühn I, Schröter M, Hauck J, Bonn A, Honrado JP (2017) Integrating ecosystem services and disservices: insights from plant invasions. Ecosyst Services 23:94–107

Venette R (2013) Incorporating climate change into pest risk models for forest pathogens: a role for cold stress in an era of global warming? NeoBiota 18:131–150

Vicente J, Alves P, Randin C, Guisan A, Honrado J (2010) What drives invasibility? A multimodel inference test and spatial modelling of alien plant species richness patterns in northern Portugal. Ecography 33(6):1081–1092

Vicente J, Randin CF, Gonçalves J, Metzger MJ, Lomba Â, Honrado J, Guisan A (2011) Where will conflicts between alien and rare species occur after climate and land-use change? A test with a novel combined modelling approach. Biol Invasions 13(5):1209–1227

Vicente JR, Fernandes RF, Randin CF, Broennimann O, Gonçalves J, Marcos B, Pôças I, Alves P, Guisan A, Honrado JP (2013) Will climate change drive alien invasive plants into areas of high protection value? An improved model-based regional assessment to prioritise the management of invasions. J Environ Manag 131:185–195

Vilà M, Basnou C, Pyšek P, Josefsson M, Genovesi P, Gollasch S, Nentwig W, Olenin S, Roques A, Roy D, Hulme PE (2010) How well do we understand the impacts of alien species on ecosystem services? A pan-European cross-taxa assessment. Front Ecol Environ 8(3):135–144

Vinodia S, Dixit AK (2017) *Habenaria diphylla* (Nimmo) Dalzell: a new addition to the Orchid flora of Bilaspur district (CG), Central India. Curr Bot 8:60–65

Vitousek PM, Antonio CM, Loope LL, Westbrooks R (1996) Biological Invasions as global environmental change. Am Sci 84(5):468

Wagh VV, Jain AK (2015) Invasive alien flora of Jhabua district, Madhya Pradesh, India. Int J Biodivers Conserv 7(4):227–237

Walther GR, Roques A, Hulme PE, Sykes MT, Pyšek P, Kühn I, Zobel M, Bacher S, Botta-Dukát Z, Bugmann H, Czucz B (2009) Alien species in a warmer world: risks and opportunities. Trends Ecol Evol 24(12):686–693

Wang S, Niu S (2016) Do Biological Invasions by *Eupatorium adenophorum* increase forest fire severity? Biol Invasions 18(3):717–729

Weber EF (1997) The alien flora of Europe: a taxonomic and biogeographic review. J Veg Sci 8(4):565–572

Weber E, Sun SG, Li B (2008) Invasive alien plants in China: diversity and ecological insights. Biol Invasions 10(8):1411–1429

Whitmore TC (1990) An introduction to tropical rain forests. Clarendon Press, Oxford, p 226

Williams JR (1954) The biological control of weeds. Report of the Sixth Commonwealth Entomological Congress London United Kingdom, pp 95–98

Williamson M (1996) Biological Invasions. Chapman and Hall London

Williamson M, Fitter A (1996) The varying success of invaders. Ecology 77(6):1661–1666

Wilson JR, Dormontt EE, Prentis PJ, Lowe AJ, Richardson DM (2009) Something in the way you move: dispersal pathways affect invasion success. Trends Ecol Evol 24(3):136–144

Xu H, Qiang S, Han Z, Guo J, Huang Z, Sun H, He S, Ding H, Wu H, Wan F (2004) The distribution and introduction pathway of alien invasive species in China. Biodiv Sci 12(6):626–638

Yang Q, Carrillo J, Jin H, Shang L, Hovick SM, Nijjer S, Gabler CA, Li B, Siemann E (2013) Plant-soil biota interactions of an invasive species in its native and introduced ranges: Implications for invasion success. Soil Biol Biochem 65:78–85

Yannelli FA, Hughes P, Kollmann J (2017) Preventing plant invasions at early stages of revegetation: The role of limiting similarity in seed size and seed density. Ecol Eng 100:286–290

Zavaleta ES, Hobbs RJ, Mooney HA (2001) Viewing invasive species removal in a whole-ecosystem context. Trends Ecol Evol 16(8):454–459

Zimmermann TG, Andrade AC, Richardson DM (2017) Abiotic barriers limit tree invasion but do not hamper native shrub recruitment in invaded stands. Biol Invasions 19(1):109–129

Aboveground Tree Carbon Stocks Along a Disturbance Gradient in Wet Tropical Forests of South Assam, India

6

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Abstract

Human activities have been severely affecting forest structure and functions in humid tropics across the globe. In present study, we estimated aboveground biomass and carbon stocks along a disturbance gradient in wet tropical forests of southern Assam, India, using non-destructive sampling method. A total of 26 forest stands were surveyed and based on a disturbance index grouped into 4 categories, viz. undisturbed (UD), mildly disturbed (MLD), moderately disturbed (MD) and highly disturbed (HD) forests. Mean aboveground carbon (AGC) stocks and basal area decreased with increased disturbance index. Though phytosociological parameters such as species richness, Shannon-Wiener diversity index, tree density, basal area and AGC stocks showed a significant negative correlation with disturbance index, tree density (693 ± 11.6 trees ha^{-1}) and Shannon-Wiener diversity index (1.98 ± 0.07) were highest in mildly disturbed forests. Aboveground carbon stocks were positively correlated with basal area ($p < 0.01$) and diversity indices ($p < 0.01$) across disturbance regimes. Tree species such as *Cynometra polyandra, Mesua ferrea, Palaquium polyanthum, Mesua floribunda, Artocarpus chama* and *Stereospermum personatum* together contributed 41.3 ± 6.2 % and 42.4 ± 6.7% of the total AGC stocks in undisturbed and mildly disturbed forests, respectively, while *Artocarpu schama, Holarrhena pubescens, Mitragyna rotundifolia, Sapium baccatum, Schima wallichii* and

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Toona ciliata contributed 47.2 ± 3.5% in moderately and 55.4 ± 4.0% in highly disturbed forests.

Keywords
Tropical forests · C stock · Disturbance index · Basal area · Diversity

6.1 Introduction

Forests play a significant role in offsetting anthropogenic CO_2 emissions and hence in climate change mitigation and adaptation (Brown et al. 1996: IPCC 2013; Brienen et al. 2015). With the widespread concern about human activities increasing level of atmospheric CO_2, there is a need to assess the potential of native forests in carbon sequestration and storage (C) (Johnson and Kern 2002; Sharma et al. 2010; Borah and Garkoti 2011; Gandhi and Sundarapandian 2017). Forest biomass determines the potential amount of C that can be added to the atmosphere or sequestered in terrestrial ecosystems when they are managed for meeting emission targets (Brown et al. 1999; Brienen et al. 2015). Estimation of the existing C stocks in different forest ecosystems at national and local levels would help in appropriate decision-making on C management and country's intended nationally determined contributions under the United Nations Framework Convention on Climate Change (UNFCC) towards atmospheric carbon reduction targets as part of its international obligations in this regard (FCCC 2014; Sahu et al. 2016). In addition, forest C stock is also a useful measure for comparing structural and functional attributes of forest ecosystems across a wide range of environmental conditions (Brown 2002; Gandhi and Sundarapandian 2017).

Tropical forests are considered as the most diverse terrestrial ecosystems and largest pool of aboveground C stock covering only about 6% of the earth surface (Beer et al. 2010; Pan et al. 2013; Brienen et al. 2015). Agricultural expansion, urbanization and industrialization have been responsible for deforestation and forest degradation in the tropics (LaFrankie et al. 2006; Brienen et al. 2015) and changes in forest structure, which in turn influence ecosystem functions (Collins and Pickett 1987; Pawar et al. 2014; Sicard and Dalstein-Richier 2015). In view of the growing threats to tropical forests, it is important to understand how natural forests and their phytosociological attributes are affected by the progressive degradation due to anthropogenic activities and functional relationship between such attributes and C storage for developing suitable management plans (Chapin et al. 2000; Tilman 1988; Srivastava and Vellend 2005; Kirby and Potvin 2007).

Being part of Indo-Burma biodiversity hotspots, tropical forests of southern Assam are rich from a biodiversity standpoint (Borah and Garkoti 2011). Though such forests occupying inaccessible areas are still rich in biodiversity, those closer to the human settlements have been facing high level of pressure mainly through extraction of timber, fuel wood, grazing and overexploitation of non-timber forest products (NTFPs) by the local people (Borah 2012; Borah et al. 2014). Such anthropogenic activities have not only accelerated biodiversity loss in the forests of the study area but also impacted other ecological functions such as productivity, carbon stocks and regeneration (Borah 2012; Borah et al. 2014; Athokpam et al. 2014; Borogayary et al. 2018). In this paper we estimate the aboveground C stocks of the tree species in some wet tropical forests of southern Assam along a disturbance gradient and establish its relationships with various phytosociological attributes.

6.2 Materials and Methods

6.2.1 Study Site

The present study was conducted in the three districts, viz. Cachar, Hailakandi and Karimganj, of southern Assam (24°08′–25°05′N, 92°15′–93°15′E) collectively known as Barak Valley (Fig. 6.1). The valley is characterized by hot and humid climate and covering 6920 km^2 geographical area of which 55% is covered by forests (FSI 2017). According to Champion and Seth (1968), vegetation of the southern Assam is dominated by Cachar tropical evergreen forests and Cachar tropical semi-evergreen forests with *Cynometra polyandra*, *Mesua ferrea*, *Stereospermum personatum*, *Artocarpus chama*, *Palaquium polyanthum*, *Mesua floribunda*, *Dysoxylum binectariferum*, *Trewia nudiflora* and *Pterygota alata* being the most dominant tree species (Borah et al. 2014).

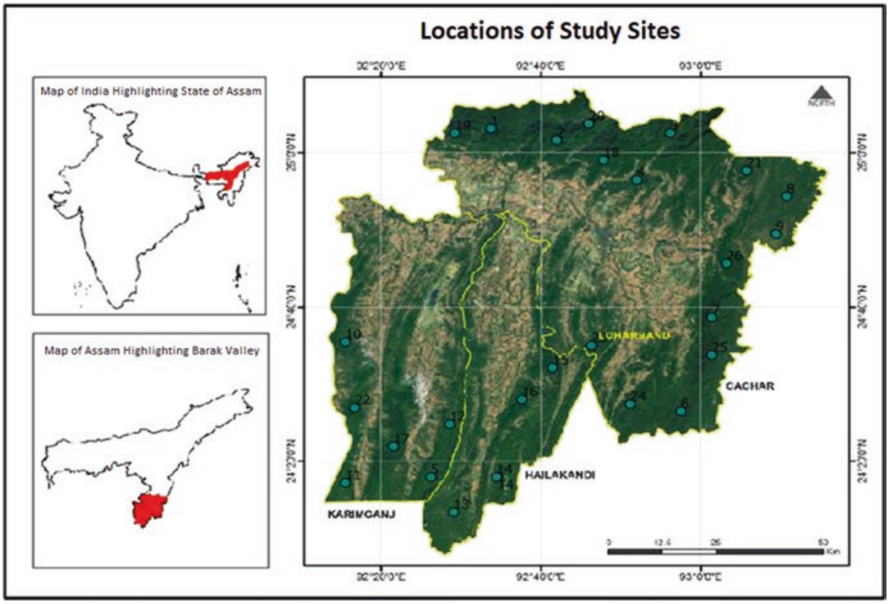

Fig. 6.1 Maps of the present study area

6.2.2 Vegetation Sampling

Based on visual reconnaissance, total 26 forest sites having different levels of disturbance (reflected by number of cut stumps, human trails and canopy cover) were selected (Fig. 6.1). The vegetation of each forest site was analysed using belt transect of 500 m × 10 m following Ganesh et al. (1996) and Borah and Garkoti (2011). Each transect was again subdivided into 50 quadrats of 10 m × 10 m size along its length. In each quadrat, all the woody plants (excluding lianas) with >10 cm circumference at breast height (CBH) were considered as tree and recorded with their CBH (Singh and Dadhwal 2009, Borah et al. 2014). The cut stumps and lopped trees in each quadrat were also counted, and their girths were measured at 10 cm from the ground for estimating the disturbance index. Plant specimens were brought to the laboratory and identified with the help of 'Flora of Assam' (Kanjilal 1934–1940), 'Assam's Flora' (Chowdhury et al. 2005) and the herbarium of the regional centre of Botanical Survey of India located at Shillong, Meghalaya.

6.2.3 Disturbance Index

A disturbance index for each forest site was calculated following Kanzaki and Kyoji (1968), Rao et al. (1990) and Borah (2012). The disturbance index (DI) was calculated as the basal area of cut trees measured at the ground level expressed as a fraction of total basal area of all trees

$$\mathrm{DI}\left(\%\right)=\frac{\text{Basal area of cut stumps}\times 100}{\text{Total basal area}\left(\text{cut stumps basal area}+\text{Standing tree basal area}\right)}$$

Based on the disturbance index, the forest sites were classified into (1) undisturbed forest (disturbance index 0%), (2) mildly disturbed forest (disturbance index up to 20%), (3) moderately disturbed forest (disturbance index 20–40%) and (4) highly disturbed forest (disturbance index above 40%) following Bhuyan et al. (2003). Of the 26 forest sites, 5 sites were recorded as undisturbed forests, 6 forest sites were mildly disturbed, 7 sites were moderately disturbed and 8 sites were highly disturbed. Generally undisturbed and mildly disturbed forests were approximately >6 kms, and moderately and highly disturbed forests were <6 km away from the human habitation. Thus, proximity to human habitation determined the level of disturbance in the forests of the study area.

6.2.4 Phytosociological Attributes

The phytosociological data were quantitatively analysed for frequency, density, basal area, relative density, relative frequency and relative dominance (Curtis and McIntosh 1950). Distribution of tree density and basal area in different DBH classes was estimated by following Mueller-Dombois and Ellenberg (1974). Trees were categorized into ten DBH classes starting with <10 cm and ending with >90 cm class, and tree density and basal area in each DBH class were estimated.

Diversity index was calculated following Shannon and Wiener (1963) as follows:

$$H = -\sum_{S}^{i=1} P_i In P_i$$

where p_i is the proportion of individuals of *i*th species and total number of individuals of all species.

The concentration of dominance was calculated following Simpson (1949) as follows:

$$\mathrm{CD} = \sum P_i^2$$

where p_i is the proportion of individuals of *i*th species and total number of individuals of all species.

6.2.5 Aboveground Biomass (AGB) and Aboveground Carbon (AGC)

Because of high species richness (132 tree species) in the studied forests, species-specific regression models were not used. Though there are several regression equations (Brown et al. 1989: Chambers et al. 2001; Chave et al. 2001; Brown and Iverson 1992) for estimating aboveground biomass (AGB), we used the following equation developed by Brown (1997) for simplicity, less prediction error and higher R^2 value. The regression model is

$$Y = 21.297 - 6.953(D) + 0.740(D^2) \quad \text{with } R^2 \text{ value } 0.87.$$

where Y is the aboveground biomass (AGB) and D is diameter of the tree.

We assumed vegetation carbon equal to 50% of biomass for all the tree components. The estimates are based on assumption of common carbon content per biomass unit as in many other similar studies (Brown and Lugo 1982; Montagnini and Porras 1998; Borah et al. 2013).

6.2.6 Statistical Analysis

ANOVA was performed to compare the average tree density, basal area, diversity index, AGB and AGC among the forest categories with the help of SPSS 16.0. Pearson correlation coefficient was calculated to determine the functional relationships among the phytosociological attributes of the forests.

6.3 Results and Discussion

6.3.1 Forest Structure

A total of 132 tree species was recorded in 26 forest stands. The highest number of species (92) was recorded in mildly disturbed forests and the lowest (47) in highly disturbed forests (Table 6.1). The Shannon diversity index of undisturbed forests ranged from 1.72 to 2.14, while it was recorded 1.78 to 2.19, 1.29 to 1.71 and 1.11 to 1.35, respectively, in mildly, moderately and highly disturbed forests. The average Shannon diversity index was highest (1.98) in mildly disturbed forests followed by undisturbed (1.94), moderately (1.51) and highly disturbed (1.23) forests (Table 6.1). The average tree density was highest in mildly disturbed forests (693 ± 12 trees ha^{-1}) followed by undisturbed (676 ± 10 trees ha^{-1}), moderately (675 ± 17 trees ha^{-1}) and highly disturbed (328 ± 6 trees ha^{-1}) forests. The average basal area was highest (42.9 ± 1.6 m^2 ha^{-1}) in undisturbed forests followed by mildly disturbed forests (39.6 ± 1.8 m^2 ha^{-1}), moderately disturbed forests (20.8 ± 1.0 m^2 ha^{-1}) and highly disturbed forests (14.4 ± 0.9 m^2 ha^{-1}). The Simpson dominance index was highest (0.075) in highly disturbed forests and lowest (0.047) in mildly disturbed forests (Table 6.1). All the phytosociological attributes (species number, tree density, diversity and basal area) were significantly different in different forest categories (Table 6.1). The forest structural analysis reveals that the species richness (47–92 species), Shannon index (1.11–2.19), tree density (328–693 trees ha^{-1}) and tree basal area (14.37–42.91 m^2 ha^{-1}) recorded in different forest categories are comparable with earlier studies (Table 6.2) in tropical forests of north-east India by Nath et al. (2005), Deb and Sundriyal (2008), Deb et al. (2009), Borah and Garkoti (2011), Thapa et al. (2011), Borah et al. (2013, 2014) and Nandy and Das (2013).

The highest species richness, diversity and density of tree species in the mildly disturbed forests were due to the favourable conditions (such as less competition for available resources, more sunlight penetration to the forest floor through canopy gaps, etc.) for the growth and regeneration of tree species (Boarh and Garkoti 2011). Pressures such as relentless extraction of fuel wood, tree felling, and non-timber forest products including bamboo collection were found responsible for lower species richness, density, diversity and basal area in moderately and highly disturbed forests (Boarh and Garkoti 2011; Borah et al. 2014).

6.3.2 Distribution of Density and Basal Area in Different DBH Classes

Density and basal area distribution in different DBH (diameter at breast height) classes can be used as indicators of changes in population structure and species composition (Newbery and Gartlan 1996). The distribution of tree density and basal area in different DBH classes of undisturbed and mildly disturbed forests revealed a reversed J-shaped and J-shaped curve, respectively. In undisturbed and mildly disturbed forests, tree density decreased, and basal area increased with increasing

Table 6.1 Phytosociological attributes of different wet tropical forests along disturbance gradient of southern Assam

	Forest categories				ANOVA	
Phytosociological attributes	Undisturbed	Mildly disturbed	Moderately disturbed	Highly disturbed	F-value	Significance
Number of species	78	92	68	47	72.3	$P < 0.001$
Number of genera	65	73	45	38	12.45	$P < 0.05$
Tree density (tree ha^{-1})	676.2 ± 10.1	693.3 ± 11.6	417.17 ± 17.2	328.6 ± 5.79	245.32	$P < 0.001$
Basal area ($\text{m}^2\ \text{ha}^{-2}$)	42.9 ± 1.6	39.6 ± 1.8	20.8 ± 1.0	14.4 ± 0.9	101.7	$P < 0.001$
Shannon diversity index (H)	1.94 ± 0.03	1.98 ± 0.07	1.51 ± 0.04	1.23 ± 0.02	64.69	$P < 0.001$
Simpson index (Cd)	0.051 ± 0.002	0.047 ± 0.001	0.052 ± 0.002	0.075 ± 0.001	57.25	$P < 0.05$
AGB (Mg ha^{-1})	361.6± 9.0	358.9 ± 9.0	240.9 ± 7.9	212.4 ± 6.9	115.81	$P < 0.001$
AGC-stock (Mg ha^{-1})	180.8 ± 4.5	179.4 ± 4.48	120.5 ± 3.9	106.2 ± 3.5	115.81	$P < 0.001$

Table 6.2 Comparison of the phytosociological attributes with other tropical forests of north-east India

Author	Location	Species number	Shannon index	Density (trees ha^{-1})	Basal area (m^2 ha^{-1})
Nath et al. (2005)	Namdapha, Arunachal Pradesh	18–50	0.96–1.45	340–610	7.8–98.5
Deb et al. (2009)	Arunachal Pradesh	130	–	245–418	18.3–49.7
Deb and Sundriyal (2008)	Arunachal Pradesh	20–73	1.71–3.85	245–418	–
Borah and Garkoti (2011)	Barak Valley, Assam	49–91	1.46–1.77	396–1110	9.4–42.1
Thapa et al. (2011)	Meghalaya, India	31–94	1.81–3.77	524–1110	38.3–93.7
Borah et al. (2013)	Barak Valley, Assam	22–62	1.02–1.62	295–965	5.37–37.09
Nandy and Das (2013)	Barak Valley	32–42	3.0–3.36	522–865	41.6–74.05
Borah et al. (2014)	Cachar District, Assam	22–62	1.02–1.62	295–965	5.37–37.09
Present study	Southern Assam	47–92	1.39–1.69	328–693	14.4–42.9

DBH classes (Fig. 6.2). In both undisturbed and mildly disturbed forests, tree density was high in <10 cm DBH class, whereas tree total basal area was high in >90 cm DBH class. High tree density in lower DBH classes indicates continuous regeneration of the forest ecosystems and higher total basal area in higher DBH classes in the forests represent relatively undisturbed and old growth forests. Similar trends of density and basal area distribution were also observed for different tropical forests of north-east India by Khamyong et al. (2004), Nath et al. (2005), Deb et al. (2009), Borah and Garkoti (2011) and Borogayary et al. (2018). In moderately disturbed and highly disturbed forests, both tree density and basal area showed fluctuating distribution curves. In moderately disturbed forests, the density of younger trees (DBH <10 cm) was recorded highest, whereas trees in DBH class between 40 and 50 cm contributed more to total basal area. Lower basal area in higher DBH classes indicates the selective extraction of larger trees from these forests. In highly disturbed forests, tree density in DBH class 20–30 cm was higher, whereas trees in 80–90 cm DBH class contributed highest to total basal area. Lower density of younger trees indicates the low regeneration in the highly disturbed forests.

6.3.3 Aboveground Biomass (AGB) and Aboveground Carbon Stocks (AGC)

The mean aboveground biomass (AGB) (361.6 ± 9.0 Mg ha^{-1}) and aboveground carbon (AGC) (180.8 ± 4.5 Mg ha^{-1}) were estimated highest for undisturbed forests

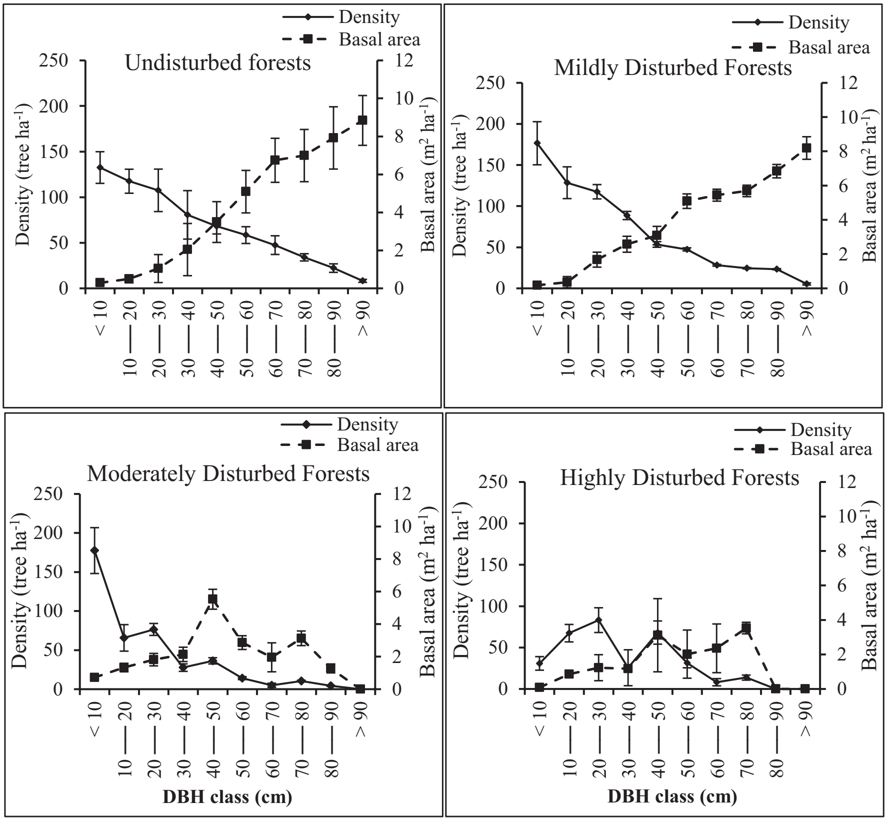

Fig. 6.2 Distribution of tree density and basal area in different DBH classes in undisturbed and differently disturbed forests. The error bars indicate the standard error of mean of tree density in each DBH class

followed by mildly disturbed forests, moderately disturbed forests and highly disturbed forests. The AGB and AGC varied significantly ($P < 0.001$) in different forest categories (Table 6.1). The AGB in undisturbed (361.6 Mg ha^{-1}) and mildly disturbed forests (358.0 Mg ha^{-1}) were within the range 236.0–425.7 Mg ha^{-1}) reported for tropical forests of India (Chaturvedi et al. 2011, Gogoi et al. 2017). Similarly, AGB of moderately (240.5 Mg ha^{-1}) and highly disturbed forests (212.4 Mg ha^{-1}) in the present study was comparable with disturbed tropical forests of Mizoram (116.8–278.5 Mg ha^{-1}) as reported by Singh and Sahoo (2018).

The lower AGB and AGC recorded in the disturbed forests may be mainly due to the lower tree basal area and tree density, especially in higher size classes. Local people living in the proximity of the forests constantly extract forest resources for meeting their subsistence needs of fuel wood, construction materials, timber and NTFPs from these forests, which resulted in a decline in tree basal area as well as lower AGB and AGC in disturbed forests.

Of the total 78 species recorded in the undisturbed forests, the top 5 contributors to aboveground carbon (AGC) were *Cynometra polyandra* (11.5%), *Palaquium polyanthum* (8.8%), *Artocarpus chama* (7.8%), *Mesua ferrea* (7.4%) and *Mesua floribunda* (5.5%) which cumulatively contributed 41.3 ± 6.2% of the total AGC (Fig. 6.3). In mildly disturbed forests, these were *C. polyandra* (12.9%), *M. ferrea* (9.6%), *S. personatum* (7.3%), *P. polyanthus* (6.5%) and *A. chama* (6.5%) and together contributed 42.4 ± 6.7% of the total AGC. In moderately disturbed forests, *A. chama* (15.8%) was the top contributor of AGC followed by *Tetrameles nudiflora*, *Schima wallichii*, *Toona ciliata* and *Bombax ceiba* (Fig. 6.2). These five species contributed 47.2 ± 3.5% of the total AGC in moderately disturbed forests. *S. wallichii*, *H. pubescens*, *S. baccatum*, *Mitragyna rotundifolia* and *T. ciliata* were five major contributors to AGC and aggregately contributed 55.2 ± 4.0% of the total AGC in highly disturbed forests (Fig. 6.3). In highly disturbed forests, *S. wallichii* alone contributed 26.2% of the total AGC implying that species contributed higher

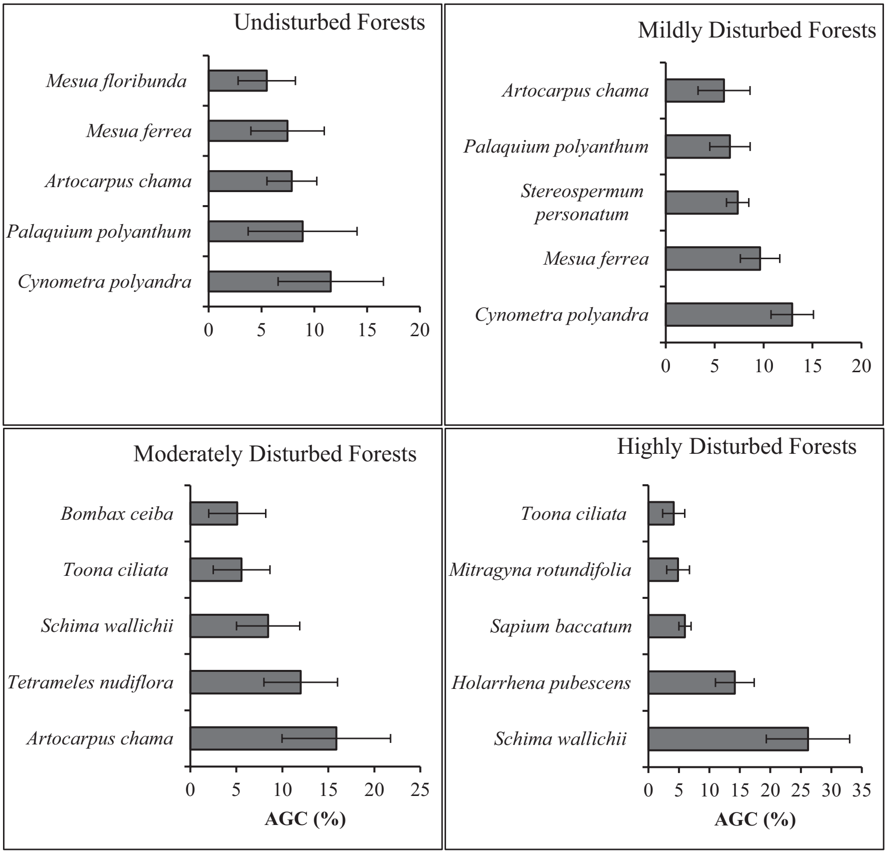

Fig. 6.3 Top five contributors to aboveground carbon (AGC) in undisturbed and differently disturbed forests. The error bars indicate the standard error of mean of AGCs in each species

Table 6.3 Correlation between AGC of dominant tree species with disturbance index

Species name	R^2	Significance
Artocarpus chama	0.09	ns
Bombax ceiba	0.1	ns
Cynometra polyandra	0.58	$P < 0.01$
Holarrhena pubescens	0.43	$P < 0.01$
Mesua ferrea	0.47	$P < 0.01$
Mesua floribunda	0.32	$P < 0.01$
Mitragyna rotundifolia	0.47	$P < 0.01$
Palaquium polyanthum	0.25	$P < 0.01$
Schima wallichii	0.71	$P < 0.01$
Stereospermum personatum	0.68	$P < 0.01$
Tetrameles nudiflora	0.06	ns
Toona ciliata	0.4	P < 0.01

in degraded forests were less preferred for local extraction. Removal or conservation of the top AGC contributor species greatly influenced total AGC across differently disturbed forests. Therefore, identification and conservation of such tree species are important for sustainable carbon management viewpoint in tropical forests (Kirby and Potvin 2007; Borah et al. 2013) (Table 6.3).

It was observed that the relative contribution to AGB and AGC by *C. polyandra*, *M. ferrea*, *M. floribunda*, *P. polyanthum* and *S. personatum* in the forest sites decreased along the disturbance gradient (Fig. 6.4). These species (*C. polyandra*, *M. ferrea*, *M. floribunda*, *P. polyanthum* and *S. personatum*) were dominant species in undisturbed and mildly disturbed forests (Borah 2012). However, these being the most preferred timber species in the study area, local people frequently extracted them from the forests resulting in a decrease in their presence in the disturbed forests and corresponding contribution in forest AGC. Statistical analysis revealed a significant ($p < 0.1$) negative correlation between disturbance and AGC of *C. polyandra*, *M. ferrea*, *M. floribunda*, *P. polyanthum* and *S. personatum* (Table 6.3). However, species such as *H. pubescens*, *S. wallichii*, *M. rotundifolia* and *T. ciliata* showed significantly ($p < 0.0$) increasing trend of AGC with the increasing levels of disturbance (Fig. 6.3; Table 6.3). Relatively higher AGC of these species in moderately and highly disturbed forests may be due to favourable conditions created by selective extraction of preferred species providing ample space for growth and regeneration and more light penetration through canopy gaps (Sahoo and Lalfakawma 2010; Asase et al. 2012). However, *Bombax ceiba* and *Trewia nudiflora* contributed higher to AGC in mildly and moderately disturbed forests (Fig. 6.4). Distribution of AGC in different DBH classes in undisturbed and mildly disturbed forests showed reversed J shaped curves while in moderately and highly disturbed forests it showed fluctuating curves (Fig. 6.5). Reversed J shaped curves in relatively undisturbed forests were due to higher total basal area in the higher DBH classes in these forests and fluctuating curves in moderately and highly disturbed forests may be due to excessive removal of certain DBH class trees from these forests.

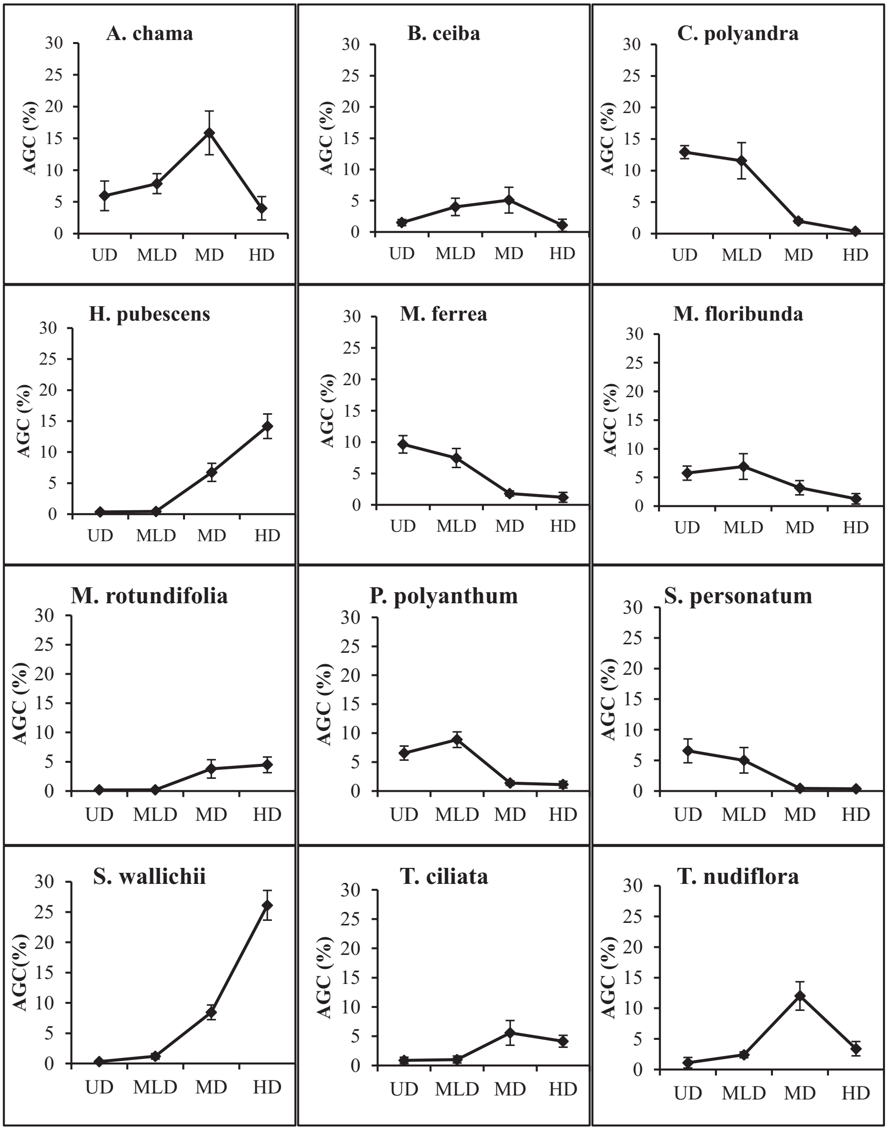

Fig. 6.4 Distribution of aboveground carbon (AGC) of dominant tree species in undisturbed and differently disturbed forests, viz. mildly disturbed (MLD), moderately disturbed (MD) and highly disturbed forests. The error bars indicate the standard error of mean of AGCs in each forest category

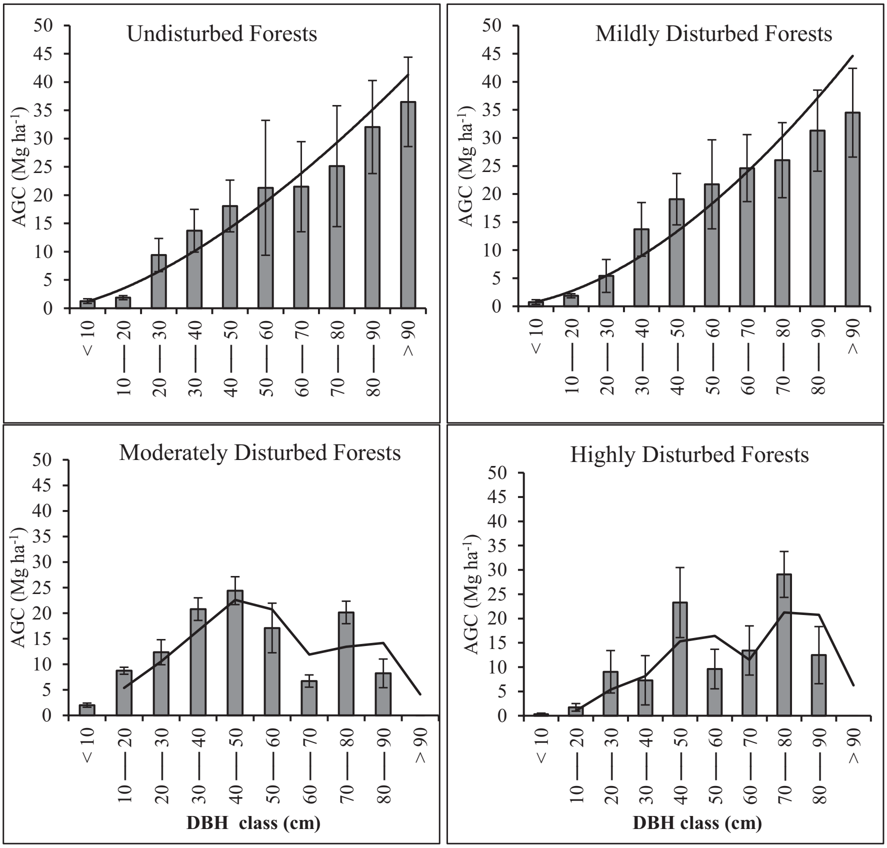

Fig. 6.5 Distribution of aboveground carbon (AGC) in different DBH classes in different forest categories of south Assam. The error bars indicate the standard error of mean of AGCs in each DBH class

6.3.4 Relationship of AGC with Phytosociological Attributes and Disturbance Index

AGC was positively correlated ($P < 0.001$) with basal area ($R^2 = 0.885$), total number of species ($R^2 = 0.696$) and Shannon diversity index ($R^2 = 0.867$) (Fig. 6.6) which is similar to earlier findings of various workers for tropical forests (Terakunpisut et al. 2007; Kumar et al. 2011; Borah et al. 2013; Con et al. 2013; Thokchom and Yadava 2013; Van de Perre et al. 2018). Contrary to the present findings, studies by Enquist and Niklas (2001) and Kirby and Potvin (2007) found no correlation between AGC and diversity indices. Frivold and Frank (2002) mentioned that there may be positive or negative relationship depending on the diversity index used and relative dominance of species in mixed forests. No significant relationship was found between AGC and tree density (Fig. 6.6). The phytosociological attributes such as species richness, diversity, tree density, basal

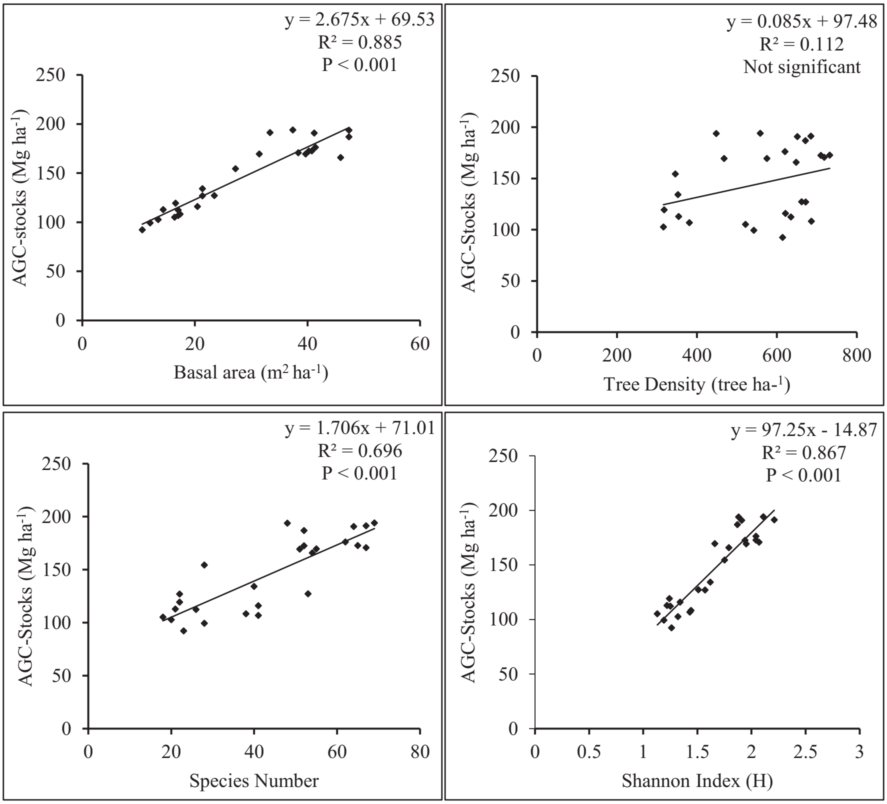

Fig. 6.6 Relationship of aboveground carbon (AGC) with basal area, tree density, species number and diversity index

Table 6.4 Relationship of disturbance index with different phytosociological attributes and aboveground carbon in tropical forests of south Assam

Attributes	r	P
Species number	−0.98	$P < 0.01$
Tree density	−0.81	$P < 0.05$
Tree basal area	−0.98	$P < 0.01$
Shannon index	−0.97	$P < 0.01$
AGC stocks	−0.96	$P < 0.01$

area and AGC in the present study showed significant negative correlation with disturbance index (Table 6.4). A similar relationship was also reported for different tropical forests by Bhuyan et al. (2003), Marín-Spiotta et al. (2007), Asase et al. (2012), Borah et al. (2013), Gogoi et al. (2017), and Van de Perre et al. (2018).

6.4 Conclusions

The aboveground tree carbon stocks were significantly lower in the disturbed forest stands than in undisturbed stands due to frequent removal of large trees from the formers. Locally preferred species, viz. *C. polyandra*, *M. ferrea*, *M. floribunda*, *P. polyanthum*, *A. chama* and *S. personatum*, contributed higher to the total AGC in undisturbed and mildly disturbed forests, whereas less preferred species, viz. *S. wallichii*, *H. pubescens*, *S. baccatum*, *M. rotundifolia* and *T. ciliata*, contributed higher in moderately and highly disturbed forests. Contribution of older trees with higher girths was higher to the total AGC stocks than the younger trees in lower girth classes irrespective of disturbance level. The phytosociological attributes, viz. tree diversity, species richness and basal area, revealed significant positive correlations with AGC stocks across disturbance gradient.

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References

Athokpam FD, Garkoti SC, Borah N (2014) Periodicity of leaf growth and leaf dry mass changes in the evergreen and deciduous species of southern Assam, India. Ecol Res 29(2):153–165

Asase A, Asitoakor BK, Ekpe PK (2012) Linkages between tree diversity and carbon stocks in unlogged and logged West African tropical forests. Int J Biodivers Sci Ecosyst Serv Manag 8(3):217–230

Beer C, Reichstein M, Tomeller E et al (2010) Terrestrial gross carbon dioxide uptake: global distribution and covariation with climate. Science 329:834–838

Borah N (2012) Community structure, tree regeneration and utilization of forest resources in Cachar and Hailakandi districts of Assam, India. Ph.D. thesis, Assam University, Silchar, Assam, India

Borah N, Garkoti SC (2011) Tree Species composition, diversity, and regeneration patterns in undisturbed and disturbed forests of Barak Valley, South Assam, India. Int J Ecol Environ Sci 37(3):131–141

Borah N, Nath AJ, Das AK (2013) Aboveground biomass and carbon stocks of tree species in tropical forests of Cachar district, Assam, northeast India. Int J Ecol Environ Sci 39(2):97–106

Borah N, Athokpam FD, Garkoti SC, Das AK, Hore DK (2014) Structural and compositional variations in undisturbed and disturbed tropical forests of Bhuban hills in south Assam, India. Int J Biodivers Sci Ecosyst Serv Manage 10(1):9–19

Borogayary B, Nath AJ, Das AK (2018) Tree species composition and population structure of a secondary tropical evergreen forest in Cachar district, Assam. J Environ Biol 39:67–71

Brienen RJW, Brienen RJW, Phillips OL et al (2015) Long-term decline of the Amazon carbon sink. Nature 519:344–348

Brown S (1997) Estimating biomass and biomass change of tropical forests: a primer. FAO forestry paper 134. Food and Agriculture Organization, Rome, p 55

Brown S (2002) Measuring, monitoring and verification of carbon benefits for forest-based projects. Philos Trans R Soc A 360:1669–1683

Brown S, Iverson LR (1992) Biomass estimates for tropical forests. World Resour Rev 4:366–384

Brown S, Lugo AE (1982) The storage and production of organic matter in tropical forests and their role in the global carbon cycle. Biotropica 14:161–187

Brown S, Gillespie A, Lugo A (1989) Biomass estimation methods for tropical forests with applications to forest inventory data. For Sci 35:881–902

Brown S, Sathaye J, Cannell M, Kauppi P (1996) Mitigation of carbon estimation to the atmosphere by forest management. Commonw For Rev 75(1):80–91

Brown S, Schroeder P, Kern JS (1999) Spatial distribution of biomass in forest of the eastern USA. For Ecol Manag 123:81–90

Bhuyan P, Biodiversity and Conservation 12 (8):1753–1773

Chambers JQ, Santos JD, Ribeiro RJ, Higuchi N (2001) Tree damage, allometric relationships, and aboveground net primary production in central Amazon forest. For Ecol Manag 152:73–84

Champion HG, Seth SK (1968) A revised survey of the forest types of India. Govt. of India Publications, New Delhi

Chapin FS, Zavaleta ES, Eviner VT, Naylor RL, Vitousek PM, Reynolds HL, Hooper DU, Lavorel S, Sala OE, Hobies SE, Mack MC, Diaz S (2000) Consequences of changing biodiversity. Nature 405:234–242

Chaturvedi RK, Raghubanshi AS, Singh JS (2011) Carbon density and accumulation in woody species of tropical dry forest in India. For Ecol Manag 262:1576–1588

Chave J, Rikra B, Dubois MA (2001) Estimation of biomass in a neotropical forest of French Guiana: spatial and temporal variability. J Trop Ecol 17:79–96

Chowdhury S, Nath AK, Bora A, Das PP, Phukan U (2005) Assam's flora. Assam Science Technology and Environment Council, Guwahati

Collins BS, Pickett STA (1987) Influence of canopy openings on the environment and herb layer in a northern hardwood forest. Vegetatio 70:3–10

Con TV, Thang NT, Ha DTT, Khiem CC, Quy TH, Lam VT, Do TV, Sato T (2013) Relationship between aboveground biomass and measures of structure and species diversity in tropical forests of Vietnam. For Ecol Manage 310:213–218

Curtis JT, McIntosh RP (1950) The interrelations of certain analytic and synthetic phytosociological characters. Ecology 31:434–455

Deb P, Sundriyal RC (2008) Tree regeneration and seedling survival patterns in old growth lowland tropical rainforest in Namdapha National Park, North-east India. For Ecol Manage 225:3995–4006

Deb P, Sundriyal RC, Shankar U (2009) Tree diversity and population structure in a lowland tropical rainforest in the Eastern Himalaya, India. Indian For 135(11):1526–1544

Enquist BJ, Niklas KJ (2001) Invariant scaling relation across tree dominated communities. Nature 410:655–660

FCCC-Framework Convention on Climate Change (2014) Report of the conference of the parties on its nineteenth session, held in Warsaw from 11 to 23 November 2013, 43pp

Frivold LH, Frank J (2002) Growth of mixed birch-coniferous stand in relation to pure coniferous stand at similar sites in south-eastern Norway. Scand J For Res 17:139–149

FSI (Forest Survey of India) (2017) State Forest report. Ministry of Environment and Forests, Dehradun, p 2017

Gandhi DS, Sundarapandian S (2017) Large-scale carbon stock assessment of woody vegetation in tropical dry deciduous forest of Sathanur reserve forest, Eastern Ghats, India. Environ Monit Assess 189:187–196

Ganesh T, Ganesan R, Devy MS, Davidar P, Bawa KS (1996) Assessment of plant biodiversity at mid-elevation evergreen forest of Kalak and Mundanthuari Tiger Reserve, Western Ghats, India. Curr Sci 71:379–392

Gogoi A, Sahoo UK, Singh SL (2017) Assessment of biomass and total carbon stock in a tropical wet evergreen rainforest of Eastern Himalaya along a disturbance gradient. J Plant Biol Soil Health 4(1):1–8

IPCC (Intergovernmental Panel on Climate Change) (2013) Climate change. The physical science basis. Working group I contribution to the IPCC fifth assessment report. Cambridge University Press, Cambridge

Johnson MG, Kern JS (2002) Quantifying the organic carbon held in forested soils of the United States and Puerto Rico. In: Kimble JM, Heath LS, Birdsey RA, Lal R (eds) The potential of U.S. forest foils to sequester carbon and mitigate the greenhouse effects. Lewis Publishers (CRC Press), Boca Raton, pp 47–72

Kanjilal UN (1934–1940) Flora of Assam, vol I–IV. Govt. of Assam, Shillong

Kanzaki M, Kyoji Y (1968) Regeneration in subalpine coniferous forests: mortality and pattern of death of canopy trees. Bot Mag Tokyo 99(1053):37–52

Khamyong S, Lykke AM, Seramethakun D, Barford AS (2004) Species composition and vegetation structure of an upper montane forest at summit of Mt. Doi Inthanon, Thailand. Nord J Bot 23:83–97

Kirby KR, Potvin C (2007) Variation in carbon storage among tree species: Implication for management of small scale carbon sink project. For Ecol Manag 246:208–221

Kumar R, Gupta SR, Singh S, Patil P, Dadhwal VK (2011) Spatial distribution of forest biomass using remote sensing and regression models in northern Haryana, India. Int J Ecol Environ Sci 37(1):37–47

LaFrankie JV, Ashton PS, Chunyong GB, Co L, Condit R, Davaries SJ, Fooster R, Hubbell SP, Kenfack D, Lagunzad D, Losos EC, Nor NSM, Tan S, Thomas DW, Valencia R, Villa G (2006) Contrasting structure and composition of the understory in species rich tropical rainforests. Ecology 87:2298–2305

Marín-Spiotta E, Ostertag R, Silver WL (2007) Long-term patterns in tropical reforestation: plant community composition and aboveground biomass accumulation. Ecol Appl 17(3):828–839

Montagnini F, Porras C (1998) Evaluating the role of plantations as carbon sink: an example of an integrative approaches the humid tropics. Environ Manag 22(3):459–470

Mueller Dombois D, Ellenberg H (1974) Aims and methods of vegetation analysis. Wiley, New York

Nandy S, Das AK (2013) Comparing tree diversity and population structure between a traditional agroforestry system and natural forests of Barak valley, Northeast India. Int J Biodivers Sci Ecosyst Serv Manage 9(2):104–113

Nath PC, Arunachalam A, Khan ML, Arunachalam K, Barbhuiya AR (2005) Vegetation analysis and tree population structure of tropical wet evergreen forest in and around Namdapha National Park, northeast India. Biol Conserv 14:2109–2136

Newbery DMC, Gartlan JS (1996) A Structural Analysis of Rain Forest at Korup and Douala-Edea, Cameroon. Proc R Soc Edinburgh 104(B):107–124

Pan Y, Birdsey RA, Phillips OL, Jackson RB (2013) The structure, distribution, and biomass of the world's forests. Annu Rev Ecol Evol Syst *44*:593–622

Pawar GV, Singh L, Jhariya MK, Sahu KP (2014) Effect of anthropogenic disturbances on biomass and carbon storage potential of a dry tropical forest in India. J Appl Nat Sci 6(2):383–392

Rao P, Barik SK, Pandey HN, Tripathi RS (1990) Community composition and tree population structure of subtropical humid forest along a disturbance gradient. Vegetatio 88:151–192

Sahoo UK, Lalfakawma (2010) Population dynamics of *Schima wallichii* in an undisturbed vs a disturbed tropical forest stand of North-East India. Int J Ecol Environ Sci 36(2):157–165

Sahu SC, Suresh HS, Ravindranath NH (2016) Forest structure, composition and above ground biomass of tree community in tropical dry forests of Eastern Ghats, India. Notul Sci Biologi *8*(1):125–133

Shannon CE, Weaver W (1963) The Mathematical theory of communication. University of Illinois Press, Urbana, p 117

Sharma CM, Baduni NP, Gairola S, Ghildiyal SK, Suyal S (2010) Tree diversity and carbon stocks of some major forest types of Garhwal Himalaya, India. For Ecol Manag 260:2170–2179

Sicard P, Dalstein-Richier L (2015) Health and vitality assessment of two common pine species in the context of climate change in southern Europe. Environ Res *137*:235–245

Simpson EH (1949) Measurement of diversity. Nature 63:688

Singh S, Dadhwal VK (2009) Manual on spatial assessment of vegetation carbon pool of India. Indian Institute of Remote Sensing (National Remote Sensing Centre), ISRO, Department of Space, Govt. of India, Dehradun

Singh SL, Sahoo UK (2018) Assessment of biomass, carbon stock and carbon sequestration potential of two major land uses of Mizoram, India. Inter J Ecol Environ Sci 44(3):293–306

Srivastava D, Vellend M (2005) Biodiversity-ecosystem function research: is it relevant to conservation? Annu Rev Ecol Syst 36:267–294

Terakunpisut J, Gajaseni N, Ruankawe N (2007) Carbon sequestration potential in aboveground biomass of Thong Pha Phun national forest, Thailand. Appl Ecol Environ Res 5:93–102

Thapa N, Upadhaya K, Baishya R, Barik SK (2011) Effect of plantation on plant diversity and soil status of tropical forest ecosystems in Meghalaya, northeast India. Int J Ecol Environ Sci 37(1):61–73

Thokchom A, Yadava PS (2013) Biomass and carbon stock assessment in the sub tropical forests of Manipur, north east India. Int J Ecol Environ Sci 39(2):107–113

Tilman D (1988) Plant strategies and the dynamics and structure of plant communities. Princeton University Press, Princeton

Van de Perre F, Willig MR, Presley SJ et al (2018) Reconciling biodiversity and carbon stock conservation in an Afrotropical forest landscape. Appl Ecol 4(3):1–9

Responses of Tropical and Subtropical Plants to Air Pollution

7

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Abstract

The tropical and subtropical regions of the world are facing strong negative effects of globalization, weakening the critical balance between ecosystem stability and socioeconomic development. Apart from increasing pressures of global climate change, deforestation, and shifts in land use pattern, air pollution is emerging as one of the major factors influencing ecosystem structure and function. Issues related to health, agricultural production, and economic losses due to air pollution in the tropical and subtropical regions are well known; however, information on air pollution-related effects on the tropical vegetation is limited. Therefore, based on the current literature, the status of air pollution and its effects on vegetation in the tropical and subtropical regions of the globe are explored in this chapter to understand the current scenario and to identify the knowledge gaps. Spatial and temporal variations were detected among different regions for particulate matter, its constituents, and gaseous pollutants including identification of the factors and sources influencing the air quality. Air pollution impacts were assessed based on changes in ecosystem structure and functions such as the patterns of biodiversity change, alteration in litterfall and decomposition, the response of leaf functional traits, and bioaccumulation in the community or individual plant species. Air pollution significantly influenced major ecological processes such as litterfall, decomposition, and plant diversity indirectly through changes in soil quality as well as through a direct effect on growth and physiol-

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ogy of native plants. Plants respond to changes in air quality through alternation in growth and morphology, physiological plasticity, and modifying leaf functional traits. These responses are both species- and pollutant-specific, as not all species responded equally to air pollution. Bioaccumulation pattern also showed a distinct relationship between pollutant accumulation and species specificity.

Keywords
Tropics · Air quality · Forest · Urban · Tree · Vegetation

7.1 Introduction

Ecosystem dynamics is regulated by different biotic and abiotic factors. The balance between these factors is the key for maintaining ecosystem structure and function. Climate change along with changing land use pattern has significantly altered this balance in recent times. Air pollution considerably adds to already existing stress and further influences the environment. Compared to other stresses, air pollution impacts are both local and regional and are directly toxic to plants and animals (Mukherjee and Agrawal 2017a). Air quality influences vegetation by altering species composition and structure (Koch et al. 2016; Pandey et al. 2014), rate of decomposition (Ferreira et al. 2017), growth and morphology (Moraes et al. 2003; Mukherjee and Agrawal 2016), physiological processes (Baek and Woo 2010), leaf functional traits (Baek and Woo 2010; Moraes et al. 2002; Mukherjee and Agrawal 2016), and foliar bioaccumulation of toxic chemicals (Nakazato et al. 2016; Breulmann et al. 2002).

The tropics surround the Earth's equator within the latitudes of Tropics of Cancer and Capricorn at $\pm 23.5^{\circ}$. These latitudes relate to the axial tilt of the Earth; in all areas between them, the sun reaches a point directly overhead at least once during the solar year. Although topography and other factors mostly contribute to local climatic variations, the tropical regions are typically warm and experience little seasonal changes in daily temperatures. Important features of the tropics are the prevalence of rain in the moist regions near the equator and increasing seasonality of rainfall with distance from the equator. The tropics are the biologically richest ecosystems on the Earth (Laurance et al. 2012). A significant proportion of this diversity is already under threat, which is most likely to amplify due to the current rate of climate change. Changes in climate are likely to have a greater impact in the tropics than elsewhere because many species are specialized to deal with a narrow range of environmental conditions. Species in the tropics are not as able to tolerate changes in climate as those accustomed to more significant changes in seasonal conditions in other parts of the world.

Emissions of sulfur and nitrogen-related compounds into the atmosphere over industrialized regions in Europe, North America, and Asia have strongly affected the climatic conditions within and around these regions throughout the twentieth century (Qu et al. 2016). The present fluxes of sulfur (SO_2, SO_4^{2-}) and nitrogen

(NOx, NO_3^-, NH_3, NH_4^+) to the atmosphere are much larger than the estimated natural fluxes (Seinfeld and Pandis 2016). These large-scale perturbations have resulted in significant adverse effects on the environment. The detrimental effects include widespread acidification of the Earth's surface and groundwaters with associated biological consequences, acidification of soils, increased rate of corrosion of buildings and monuments, and formation of secondary air pollutants, such as ozone (O_3), that are toxic to all life forms (Tian and Niu 2015). Such changes may occur as a result of biological and geological processes, short-term climatic fluctuations, or volcanic eruptions or by anthropogenic activities such as emissions of air pollutants in the atmosphere from vehicles and industries.

In most of the tropical countries, particulate matter (PM) and O_3 have been identified as major air pollutants due to increasing number of vehicles and biomass burning activities (Beig and Singh 2007; Cooper et al. 2014; Mukherjee and Agrawal 2017b). Historical trends have shown a decline in PM in most regions of the world, but levels are still critical in most of the tropical countries (Mukherjee and Agrawal 2017b; WHO 2016). A positive significant increment of 7–9% in tropospheric columnar O_3 per decade has been recorded across the tropical South Atlantic, India, Southeast Asia, and the tropical/subtropical regions downwind of China (Beig and Singh 2007).

Dey et al. (2012) recorded an increase in $PM_{2.5}$ (particles 2.5 μm or less in size) levels by more than 15 $\mu g\ m^{-3}$ from 2000 to 2010 in most parts of central, northern, and eastern India and northern Bangladesh. Gurjar et al. (2016) found decreasing trend in sulfur dioxide (SO_2) levels in most of the megacities in India, whereas opposite trend was observed in the case of nitrogen oxide (NOx). Heue et al. (2016) found an increase in tropical tropospheric column O_3 by 0.70 DU decade^{-1} or 0.35% year^{-1} from 1995 to 2015. As pollutant concentrations have significantly increased in most of the tropical regions, the responses of plants to ambient air quality have also changed (Pandey et al. 1992; Mukherjee and Agrawal 2018). Improvements in tools and techniques have considerably improved our knowledge of plant response with the addition of environmental modelling and geographical information system (Sicard et al. 2016). Recent reports have observed wide variations in pollution response in different species which are due to long-term evolutionary adaptation as well as due to increase in local stress factors. Bioaccumulation studies have also shown a linear relationship between pollution concentration and bioaccumulation in recent times (Mukherjee et al. 2016). Major shifts in biodiversity pattern have also caused large-scale variations in individual plants as well as in community patterns (Narayan et al. 1994; Pandey et al. 2014).

Evaluation of responses of the tropical plants to air pollution becomes more important due to modifying effects of climatic conditions on air quality and the tropics being biologically richest ecosystems already suffering from different natural and anthropogenic threats. The present chapter describes air quality status, sources of air pollution, factors influencing air quality in the tropics, and the responses of vegetation to air pollutants in different tropical regions.

7.2 Methodology

A systematic survey was first performed in World Wide Web with keywords such as air pollution, particulate matter, air pollution sources, plant responses, biodiversity, ecosystem functions, decomposition, and leaf traits with specific filters such as tropics, subtropics, tropical forest, tropical grassland, and tropical biodiversity in PubMed, Web of Science, Google Scholar, ScienceDirect, and Springer Link. Among the articles found, only those satisfying the criteria such as sound methodology, spatial-temporal variabilities, and large dataset were screened for further analysis. Papers for the first section of the article of air quality were screened which were specifically performed under a tropical or subtropical environment of Southeast Asia; West, East, and Central Africa; and South and part of North America (Mexico) in urban, suburban, and rural environments with relevant sampling protocols and large sample size. For assessment of responses of the tropical and subtropical plants to air pollution, papers were further screened for different response types such as foliar bioaccumulation, leaf functional traits, physiology, growth and morphology, biodiversity, litterfall and decomposition, and toxicity symptoms for trees, lichens, grasses, shrubs, and crop plants.

7.3 Air Quality in the Tropics

Pollution of air and water by anthropogenic activities is a noticeable feature of urban and industrial systems throughout the tropical regions. The airborne pollutants that are of major concern in these regions include toxic atmospheric gases such as SO_2, NOx, volatile organic compounds (VOCs), O_3, and carbon monoxide (CO); PM and its components such as polycyclic aromatic hydrocarbons (PAHs) and black carbon (BC); toxic elements such as lead (Pb) and fluoride (F); photochemical oxidants; and acid deposition. Most of the airborne substances are primary pollutants released directly from stationary and mobile sources, whereas secondary pollutants are formed in the atmosphere by chemical transformation of primary pollutants.

These airborne particles, gases, and their reaction products are carried out by winds and clouds to different directions in distant places and thereafter get deposited in various forms of gases, fine and coarse particles, or as dissolved or suspended substances in precipitation. These deposition processes transport the airborne chemicals to the surfaces of vegetation, soil, surface water, buildings, and cultural resources at short or long distances from the original emission sources. The direct and indirect effects of air pollution have been summarized in Fig. 7.1. SO_2 and NOx both directly and indirectly influence human health, vegetation, and soil. The direct effects are ill health, damage to plant surface, and degradation of environmental quality. These depend upon the concentration of pollutants and decline sharply with increasing distances from emission sources. Thus, the direct effects are more of a local in nature with a geographical extent of few kilometers.

Fig. 7.1 Schematic diagram showing sources and factors influencing air pollution and its impact on different ecological processes

7.3.1 Particulate Matter

Based on PM_{10} (particles 10 μm or less in size) and $PM_{2.5}$ data of the WHO (World Health Organization) for the tropical countries (WHO 2016), Asian countries were found to be highly polluted with PM_{10} values above 100 μg m^{-3} in India and Bangladesh while above the mean annual WHO standard of 20 μg m^{-3} in the Philippines, Malaysia, Sri Lanka, Bhutan, and Indonesia. Among Asian countries, Singapore showed the least PM_{10} value of 30 μg m^{-3} (Fig. 7.2). Higher values were also recorded in African countries although data of only a few countries were available. PM_{10} values in all the African countries were above the WHO standard. Compared to Asian and African countries, PM_{10} values were reasonably lower in countries of South America. PM_{10} concentration was 33.78 μg m^{-3} in Brazil, whereas in Colombia and Ecuador values were, respectively, 39.85 and 35.82 μg m^{-3}. In Central and North American countries, PM_{10} values were above the WHO annual standard in Mexico (56.14 μg m^{-3}), Costa Rica (28.96 μg m^{-3}), and Panama (31.1 μg m^{-3}). Among the tropical cities, PM_{10} value was the highest in the Indian city of Gwalior (329 μg m^{-3}) followed by Allahabad (317 μg m^{-3}), Raipur (268 μg m^{-3}), and Delhi (229 μg m^{-3}). Among other cities, PM_{10} value (μg m^{-3}) was 170 in Kampala, the capital city of Uganda; 169.7 and 158.05, respectively, in Sylhet and Dhaka, Bangladesh; and 141 in Bamenda, Cameroon. Cities like Tezpur, India (11 μg m^{-3}); Madre de Deuss, Brazil (12 μg m^{-3}); and Pasto, Colombia (18 μg m^{-3}), showed lower values for PM_{10} (WHO 2016). In São Paulo, Rio de Janeiro, and Piracicaba, Brazil, and in Bogotá, the capital of Colombia, PM_{10} levels were above the WHO standard (Mukherjee and Agrawal 2017b). Exceedances in PM levels above the WHO standard were more frequent in African cities like Accra, Ghana, and Bamenda, Bafoussam, and Yaounde, Cameroon,

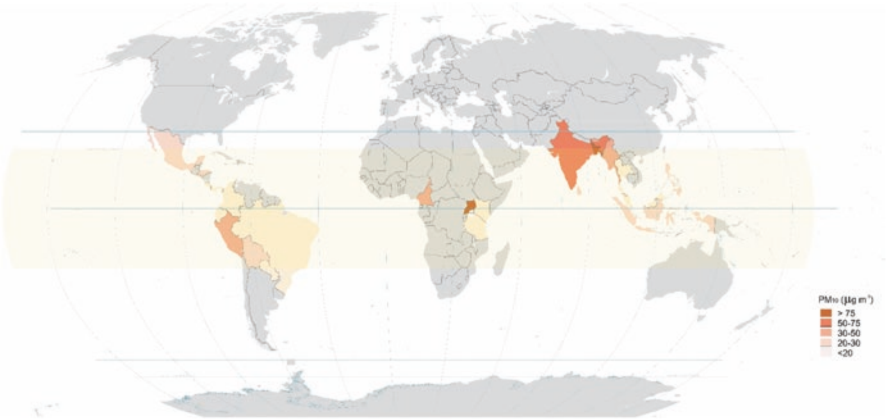

Fig. 7.2 Spatial variations in PM_{10} concentrations in different tropical countries, and shaded area represents the distribution of the tropical region

due to both natural and anthropogenic sources (Mukherjee and Agrawal 2017b). Mukherjee and Agrawal (2017b) reviewed the global status of PM_{10} and reported higher values in the tropical region compared to temperate regions of the world mostly attributed to increases in traffic and biomass burning in the tropics. The major reasons behind this trend are due to differences in the quality of vehicles and roads, density of roads, emission standard of engines, and strict regulation standards, apart from increase in vehicular density in developing countries. Although biomass burning is a traditional practice, its intensity and level have been tremendously increased in recent time due to strong economic pressure on farmers of developing countries lacking proper disposal plan of crop residues. Even in urban areas, biomass is frequently burned due to the lack of proper disposal facilities.

$PM_{2.5}$ levels showed the following trend: South Asia > Africa > Central America > North America > South America. For all the tropical countries, values were several times higher than the WHO annual limit of 10 µg m^{-3} (WHO 2016), indicating a severe problem of fine PM pollution in the tropics. $PM_{2.5}$ levels were above 50 µg m^{-3} in India, Bangladesh, Myanmar, and Cameroon (Fig. 7.3). Among the Indian cities, fine PM was maximum in Gwalior (176.14 µg m^{-3}) followed by Allahabad (169.72 µg m^{-3}), Patna (148.94 µg m^{-3}), Raipur (143.66 µg m^{-3}), and Delhi (122.10 µg m^{-3}) (WHO 2016). $PM_{2.5}$ value below the WHO annual mean standard was only recorded in 11 out of 294 cities or metropolitan areas in the world (WHO 2016). Mukherjee and Agrawal (2017a) also reported the critical condition of $PM_{2.5}$ in the Asian and African tropics. $PM_{2.5}$ levels in Dakar, Senegal; Nairobi, Kenya; and Dar es Salaam, Tanzania, were several times higher than the WHO standard.

In Delhi, the suspended particulate matter (SPM) and PM_{10} values have been continuously exceeding the standards in the last decade with a steep rise after 2005 (Gurjar et al. 2016). The observed trend is due to increased traffic load in the major cities in India. In Mumbai, SPM concentrations varied between the years with

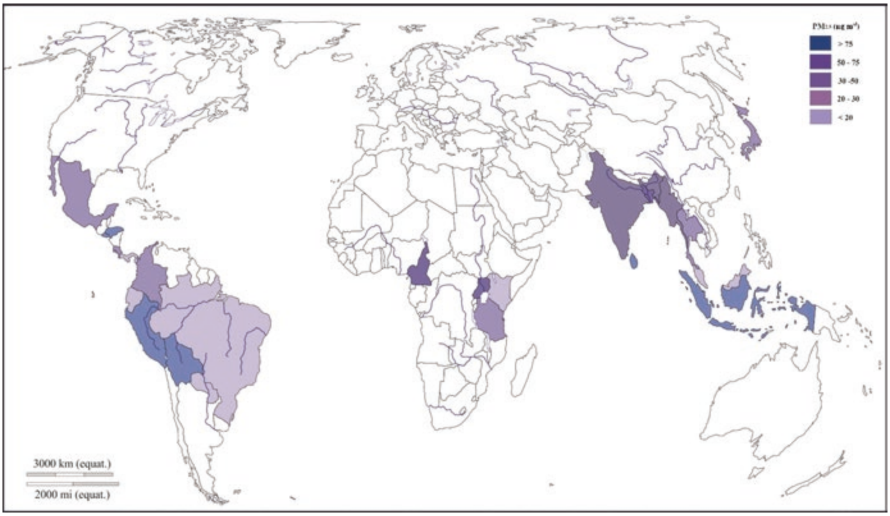

Fig. 7.3 Spatial variations in $PM_{2.5}$ concentrations in different tropical countries

increasing trend in concentrations from 2005 to 2012. SPM and PM_{10} emissions were higher in Kolkata compared to Mumbai and Delhi, whereas SPM concentration was maximum in Delhi followed by Kolkata and Mumbai (Gurjar et al. 2016). On comparison of SPM concentrations with earlier reports in Varanasi City, a clear trend of increment has been observed in the last two decades. At heavy traffic area, a 64% increase in SPM concentration was recorded during 2002–2012 (Trivedi and Agrawal 2003; Mukherjee and Agrawal 2016).

Sumit et al. (2012) assessed aerosol properties from different sites (Pune, Sinhgad, Ahmedabad, Trivandrum, and Gadanki) in India with the differences in urbanization, altitude, and land use. Aerosol optical depth, a marker of particulate matter pollution, varied from 0.23 to 0.67 between different sites, and values were significantly higher compared to reported values from Europe and the USA (<0.2) (Sumit et al. 2012; Provençal et al. 2017). Trivandrum site showed higher concentrations of small particles, whereas coarse PM was higher over Ahmedabad. The variations between different sites were mostly attributed to local emissions, meteorological conditions, and seasonal variations.

7.3.1.1 Composition of Particulate Matter

Polycyclic aromatic hydrocarbons (PAHs) are the products of fossil fuel combustion and are carcinogenic in nature (Ayi Fanou et al. 2006; Cavalcante et al. 2017). PAHs are mostly emitted from incomplete combustion of gasoline and diesel, crude oil, and organic materials. PAHs are weakly soluble in water and exist in atmosphere on the surface of particulate matter or in gaseous phase. Ayi Fanou et al. (2006) found significantly higher concentrations of benzene and PAHs associated with particulate matter in Cotonou, the largest tropical port city on the south coast of Benin in West Africa. Benzene and PAHs in ambient air were also found in the village of Cotonou, Benin, although concentrations were significantly lower than the values recorded in the city (Ayi Fanou et al. 2006). The total PAHs concentrations ranged from 1.73 to 2.83 ng m^{-3} in the metropolitan area of Fortaleza-CE, Brazil. High PAHs concentrations at the city center were correlated with the high vehicular flow (Cavalcante et al. 2017). Krauss et al. (2005) reported an average PAHs concentration of 35 ng m^{-3} in the tropical rainforest near Manaus, Brazil; however, the value was ten times lower than the nearby urban location. In the metropolitan area of Fortaleza-CE, Brazil, Cavalcante et al. (2017) reported that 62.93% variation in PAHs concentrations in PM was governed by constructed area and PAHs levels as well as PM concentrations, reduced with increasing tree cover.

In an urban area of Amritsar, Punjab, PAHs concentrations ranged from 37 to 274 ng m^{-3} due to higher biomass burning, coal combustion, and vehicular emissions, whereas concentrations were much lower with values ranging from 18.3 to 66.6 ng m^{-3} in Mumbai (Kaur et al. 2013; Sahu et al. 2008). As compared to Amritsar and Mumbai, significantly higher PAHs concentrations were recorded in Agra, India, and in Kandy, Sri Lanka (Masih et al. 2012; Wickramasinghe et al. 2011). As compared to Asian cities, values of PAHs were relatively lower in South American cities (Table 7.1).

Black carbon (BC) is formed due to incomplete combustion of fuel, biomass burning, and vehicular and industrial activities (Hung et al. 2014; Salako et al. 2012). Particle size of BC ranges from few nanometers to microns which are mostly measured in PM$_{2.5}$ range. The major portion of black carbon is made up of carbon (60%) with contribution of various aromatic compounds. Atmospheric lifetime of BC ranges from few days to weeks. BC is known for its strong radiative forcing with a stronger global warming potential than CO$_2$. BC levels showed significant spatial variations among the different tropical cities (Table 7.2). Higher values were recorded in Dhaka, Bangladesh, and Bangkok, Thailand, followed by Indian cities of New Delhi and Hyderabad. Values above 10 μg m^{-3} were recorded in Kathmandu (Nepal), Dakar (Senegal), and Hyderabad (India). Lower values of BC were recorded in Chihuahua and Tijuana, Mexico; Anantapur, India; and Kuala Lumpur, Malaysia (Table 7.2). Variations in BC concentrations depend upon sources and prevailing meteorological conditions (Salako et al. 2012).

Table 7.1 Spatial-temporal variations in PAHs concentrations in different locations/cities in different tropical countries

Location	Period	PAHs (ng m^{-3})	References
Agra, India	2006–2007	8.32–1036.79	Masih et al. (2012)
Amritsar, India	2011	37–274	Kaur et al. (2013)
Mumbai, India	–	18.3–66.6	Sahu et al. (2008)
Kandy, Sri Lanka	2008–2009	57.43–1246.12	Wickramasinghe et al. (2011)
Montenegro, Brazil	2004–2005	2.9–17.54	Pereira et al. (2013)
Porto Alegre, Brazil	2001–2002	0.04–2.30	Dallarosa et al. (2005)
São Paulo, Brazil	2003–2004	1.6–7.8	Vasconcellos et al. (2011)
Concepción, Chile	2007	26–230	Pozo et al. (2012)
Santiago, Chile	–	6.4	Romero et al. (2002)

Table 7.2 Spatial-temporal variations in black carbon concentrations in different locations/cities in different tropical countries

Location	Period	BC (μg m^{-3})	References
Ahmedabad, India	2003–2005	3.25	Ramachandran and Rajesh (2007)
Anantapur, India	2010	3.03	Reddy et al. (2012b)
Hyderabad, India	2006	12	Badarinath et al. (2009)
Hyderabad, India	2010	6.73	Swamy et al. (2012)
Kanpur, India	2006–2007	0.03–10	Baxla et al. (2009)
Mumbai, India	2008	3.58	Salako et al. (2012)
New Delhi, India	2007	14	Tiwari et al. (2009)
Pune, India	2004–2005	5.05	Panicker et al. (2010)
Dhaka, Bangladesh	2010–2011	22.77	Begum et al. (2012)
Dhaka, Bangladesh	2010	36.57	Salako et al. (2012)
Kathmandu, Nepal	2009–2010	8.6	Sharma et al. (2012)
Southern Himalaya, Nepal	2006–2008	0.16	Marinoni et al. (2010)
Bangkok, Thailand	2007–2008	9.4	Salako et al. (2012)
Bangkok, Thailand	2010	25.5	Hung et al. (2014)
Pathum Thani, Thailand	2008	4.65	Salako et al. (2012)
Kuala Lumpur, Malaysia	2008	3.85	Salako et al. (2012)
Chihuahua, Mexico	2008	2.14	Raysoni et al. (2011)
Tijuana, Mexico	2010	1.65	Shores et al. (2013)
Dakar, Senegal	2008–2009	10.4	Doumbia et al. (2012)

N deposition has been identified as the third greatest driver of biodiversity loss after land use and climate change as it alters species richness, dominance, evenness, and abundance (Xiankai et al. 2008). The tropical and subtropical areas of the world share almost 40% of the global applications of N fertilizers (Xiankai et al. 2008). In an urban atmosphere of New Delhi, India, Gupta et al. (2016) observed 2.5 times higher dust fall deposition at polluted site compared to reference site with higher concentrations of calcium ion (Ca^{+2}) and sulfate (SO_4^{-2}). Around highly industrialized region in Sonbhadra district of India, Singh and Agrawal (2005) studied N deposition at eight sites in the leeward side of thermal power plants on a 30 km transect and recorded N deposition rate of 9.67 kg ha^{-1} $year^{-1}$ at the distantly situated site from industry and 26.35 kg ha^{-1} $year^{-1}$ at the heavily polluted site, indicating the role of industrial pollution in nitrogen deposition. In the same study, SO_4^{-2} and nitrate (NO_3^-) were recorded as the major anions in the atmospheric deposits.

7.3.2 Gaseous Air Pollutants

7.3.2.1 Primary Gaseous Pollutants

After the industrial revolution, problems of gaseous air pollutants have been realized in all the developing and developed nations. Typical meteorological factors, intensive burning, and poorly maintained and higher numbers of vehicles in the tropics have resulted in a significant load of gaseous pollutants.

VOCs are gaseous air pollutants with short- to long-term adverse health effects. VOCs are emitted from combustion of fuels such as oil, gasoline, coal, wood, and natural gas. VOCs have a high vapor pressure at normal temperature and with lower water solubility. NOx is a primary air pollutant formed during combustion of fuels by the oxidation of atmospheric nitrogen within the fuel at high temperature. NOx denote the total concentrations of NO and NO_2 which are interconvertible in the presence of sunlight and other atmospheric gases especially O_3. Traffic is known to be the most significant source of NOx. SO_2 is a colorless, water-soluble gas formed by combustion of coal and oil. SO_2 is considered as one of the major harmful gases to plants as well as to public health. Major sources of SO_2 are coal-fired power plants and industries. CO is a colorless and odorless gas which is produced by the incomplete combustion of fossil fuels. Major sources of ambient CO are vehicular emissions and biomass burning.

Baumbach et al. (1995) found very high concentrations of VOCs and NOx in the large tropical West African city of Lagos, Nigeria, whereas average half-hour value of benzene was 250 pg m^{-3}. CO/NOx ratio was found to be 150–200 times higher than the previous reports in European cities (Baumbach et al. 1995). Valdés Manzanilla and de la Cruz Uc (2015) assessed spatial variability in SO_2 concentration in the tropical city of Villahermosa, Mexico, and found 363.2 tons of SO_2 emission in the metropolitan zone and the intersections of main roads in the city showed maximum concentrations of $SO_{2.}$

Dionisio et al. (2010) reported high CO concentrations in Accra, Ghana, which is mainly generated through vehicular combustion and biomass burning. CO levels were significantly lower in São Paulo, Brazil, compared to Accra. Azmi et al. (2010) found a correlation between traffic load and CO levels in Klang Valley, Malaysia. Awang et al. (2016) also recorded higher values of CO in three port cities in Malaysia. The industrial site showed maximum CO concentrations of 334 μg m^{-3}, whereas the lowest value of 42 μg m^{-3} was recorded at the semi-urban area in Dhaka, Bangladesh (Salam et al. 2008) (Table 7.3). Higher concentrations of CO around the industrial site are mainly due to incomplete combustion of fuels during industrial activities (Salam et al. 2008).

Gurjar et al. (2016) observed decreasing trend of SO_2 in all three megacities of India due to a decrease in the content of sulfur in coal and diesel, whereas increasing trend for NOx was attributed to increasing number of vehicles. Among major Indian cities, SO_2 emission was maximum in Mumbai followed by Delhi and Kolkata (Gurjar et al. 2016). Mukherjee and Agrawal (2016) also observed higher concentrations of NO_2 in different zones of Varanasi City, whereas the level of SO_2 was well below the NAAQS (CPCB 2009) of 50 μg m^{-3}. There is a remarkable reduction in SO_2 concentration in Varanasi City during the past 10 years, whereas NO_2 concentration has significantly increased by 25.6–72.2% in the last two decades (Pandey et al. 1992; Trivedi and Agrawal 2003; Mukherjee and Agrawal 2016). Both SO_2 and NO_2 concentrations were recorded below NAAQS of India in two tropical coastal cities in India (Guttikunda et al. 2015).

Table 7.3 Spatial-temporal variations in different gaseous air pollutant concentrations in different tropical countries

Location	Period	SO_2 (μg m^{-3})	O_3 (ppb)	NO_2 (μg m^{-3})	CO (ppm)	References
Agra, India	1999–2001	3.8		7		Kumar et al. (2004)
Anantapur, India	2010		20.35	23.948		Reddy et al.(2012a)
Chennai, India	2011–2012	10.85		20		Guttikunda et al. (2015)
Delhi, India	2005–2009	7.729		51.55		Mallik and Lal (2014)
Delhi, India	2013–2014		30.45			Kumar et al. (2015)
Durgapur, India	2005–2009	8.174		59.16		Mallik and Lal (2014)
Guwahati, India	2005–2009	7.074		16.28		Mallik and Lal (2014)
Hyderabad, India	2009		21			Yerramsetti et al. (2013)
Hyderabad, India	2010		14.33			Swamy et al. (2012)
Jodhpur, India	2005–2009	6.55		22.18		Mallik and Lal (2014)
Kannur, India	2009–2010		1.6–15.15			Nishanth et al. (2012)
Kolkata, India	2005–2009	10.79		58.14		Mallik and Lal (2014)
Nagpur, India	2005–2009	9.458		30.19		Mallik and Lal (2014)
Visakhapatnam, India	2011–2012	16.75		18.45		Guttikunda et al. (2015)
Dhaka, Bangladesh	2006	48.3	13.95	21.0	0.126	Salam et al. (2008)
Klang Valley, Malaysia	1997–2006	15.72	16.6	43.24	1.269	Azmi et al. (2010)
Klang, Malaysia	2009	10.48	20.3	38.16	0.952	Awang et al. (2016)
Pasir Gudang, Malaysia	2009	11.79	14.4	25.00	0.619	Awang et al. (2016)
Perai, Malaysia	2009	5.764	15.4	22.56	0.706	Awang et al. (2016)
São Paulo, Brazil	2000–2007	13.20	66	56.97	7.63	Rodrigues-Silva et al. (2012)
Santa Clara, Cuba	2010	5.95		11.35		Alejo et al. (2013)
Chihuahua, Mexico	2008			42.86		Raysoni et al. (2011)
Accra, Ghana	2006–2008				7–55	Dionisio et al. (2010)
Dar es Salaam, Tanzania	2005–2007	4.6	11.75	11.95		Mmari et al. (2013)

NO_2 concentrations were two times higher at semi-urban and coastal sites compared to a rural site, whereas SO_2 concentration was maximum at a semi-urban site (5.1 μg m^{-3}) and lowest at a rural site (0.94 μg m^{-3}) in and near Dar es Salaam, Tanzania (Mmari et al. 2013). In Dhaka, the average concentrations of SO_2 and NO_2 were much lower than the WHO standards. The highest concentration of NO_2 (40 μg m^{-3}) was found at a traffic site, whereas the highest SO_2 concentration of 76.8 μg m^{-3} was found in the commercial and heavy traffic areas (Salam et al. 2008) (Table 7.3).

VOCs are major precursors for O_3 formation. Apart from traffic and combustion sources, biogenic sources of VOCs also play a major role in O_3 formation in both urban and forested areas in the tropics. Singh et al. (2014) estimated VOCs (isoprene and monoterpene) emission potential of 60 common plant species in the Vidarbha region of Maharashtra and found maximum VOC emission rate of 75.2 g g^{-1} h^{-1} in *Dalbergia sissoo*, whereas maximum O_3-forming potential was observed in *Mangifera indica* (77 g O_3 $tree^{-1}$ day^{-1}). The study further found higher O_3-forming potential in 16 out of 60 species. Maximum monoterpene and isoprene emission rates were in *Murraya koenigii* and *D. sissoo*, respectively (Singh et al. 2014).

7.3.2.2 Secondary Gaseous Pollutants

Tropospheric O_3 is a secondary air pollutant, which is important from both atmospheric and biological perspectives. Tropospheric O_3 is a blue color gas, with a strong irritating smell formed by photochemical reactions of NOx, CO, and VOCs. O_3 concentration in the atmosphere is regulated by its precursor concentrations (hydrocarbons, CO, NOx) which are mostly emitted from combustion of fossil fuel, biomass burning, and industrial emissions and from natural sources such as lightning and microbial activity in soil. O_3 influences the oxidation capacity of the atmosphere since O_3 itself acts as a primary reactant and also due to its ability to produce hydroxyl radicals after photolysis, which play a very crucial role in the cycling of other trace gases in the atmosphere. O_3 injury in plants has been regularly reported from many areas of the tropical and temperate regions and on regional scales in North America and Europe.

Surface O_3 levels vary from 15 to 53 ppb in different tropical cities of India (Yerramsetti et al. 2013). O_3 levels were below the national and WHO standards in three port cities in Malaysia. NOx, CO, and meteorological conditions are identified as major factors behind diurnal variations in surface O_3 concentrations (Awang et al. 2016). Kumar et al. (2015) also reported O_3 concentrations below the national and WHO standards in the urban background, urban/traffic, and rural areas in Delhi-NCR, whereas concentrations were higher than both the Brazilian and WHO standards in São Paulo, Brazil (Rodrigues-Silva et al. 2012). Among the monitoring sites in Delhi-NCR, higher concentrations were observed at rural site followed by urban background and urban traffic site (Kumar et al. 2015). In an urban tropical site of Belo Horizonte, Brazil, O_3 concentrations varied from 8.7 to 96.1 ppb with an average of 38.17 ppb. Mmari et al. (2013) also reported higher O_3 concentrations at rural and coastal sites compared to a semi-urban site near Dar es Salaam, Tanzania.

Yerramsetti et al. (2013) found a role of the nocturnal chemistry of NOx in the formation of O_3 during daytime at a tropical urban site in Hyderabad (Table 7.3).

7.4 Factors Influencing Air Quality in the Tropics

7.4.1 Meteorological Factors

Air quality in the tropics is largely governed by meteorological factors apart from anthropogenic emissions. Major influencing factors that regulate spatial-temporal variations in the tropics are changing atmospheric dispersion conditions (wind direction, wind speed, atmospheric stability, relative humidity, and height of boundary layer) (Baumbach et al. 1995; Ferreira et al. 2017; Mukherjee and Agrawal 2017b; Reddy et al. 2012a). Gurjar et al. (2016) reported higher SO_2 concentrations in winter season followed by post-monsoon, pre-monsoon, and monsoon over Delhi, Mumbai, and Kolkata. PM_{10} and lead (Pb) levels were also lowered during monsoon season. In Mumbai, monthly mean SPM level also showed considerable reduction during the monsoon period from June to October (Gurjar et al. 2016). Long-term air pollution data in three Indian megacities showed ambient temperature, relative humidity, wind speed, and rainfall as the major determining meteorological variables influencing air quality. High humidity, precipitation, temperature, and light intensity have been reported to influence uptake rates and interactions of different chemical elements in tropical forests (Breulmann et al. 2002). Severe El Niño and quasi-El Niño are also responsible for forest fires in Indonesia, resulting in severe air pollution in the region (Hayasaka et al. 2014).

Among the gaseous air pollutants, surface O_3 concentration is mostly affected by meteorological factors apart from primary sources of O_3. Solar elevation, clouds, and aerosols along with temperature affect the photolysis and atmospheric reactions involved in the generation of O_3 (Silva and Tomaz 2013). Seasonal variations with the highest O_3 in the summer months and lowest during the rainy season are distinct throughout the tropical region, which are mainly due to higher temperature with long sunshine hours in summer providing an ideal condition for O_3 formation (Silva and Tomaz 2013). Nishanth et al. (2012) found positive correlation between O_3 and different meteorological variables such as solar radiation (0.91) and temperature (0.81) and negative correlation with relative humidity (0.88) and wind speed (0.77), indicating the role of meteorological factors in regulating surface O_3 concentrations at a tropical coastal site (Kannur) in India.

Boundary layer height (the lowermost part of the troposphere where the surface of the Earth interacts with the large-scale atmospheric flow) is one of the major factors influencing air pollutant concentrations in the tropical atmosphere. Reddy et al. (2012a) reported the influence of boundary layer dynamics in surface O_3 concentrations at a tropical rural site in the southern part of India. In tropics, daytime temperature and sunlight intensity are generally high all year round. The intensive sunlight intensifies photochemical processes in the atmosphere, resulting in the generation of high concentration of hydroxyl radicals (Crutzen and Giedel 1983). In large parts of the tropics, average rainfall rates are high at least during the rainy season. Therefore, wet deposition is likely to be the major deposition mechanism, but in the regions where precipitation rates are low, wet deposition processes are not significant. More studies related to atmospheric chemistry in the tropics are needed to understand both dry and wet deposition patterns with seasonal variations.

7.4.2 Sources

Air pollutants are emitted from a wide variety of sources. For air pollution control and mitigation, it is foremost important to identify and understand the nature of sources and factors responsible for regulating these sources. Baumbach et al. (1995) reported major sources of air pollution such as traffic (passenger cars and utility vehicles), industries (oil refineries with their area sources, gas flares, and electrical power stations), biomass burning (for cooking purposes and uncontrolled waste burning), and dust from Sahara Desert in large tropical West African city of Lagos, Nigeria. In the tropical city of Villahermosa, Mexico, Valdés Manzanilla and de la Cruz Uc (2015) identified mobile sources as the major sources of SO_2 pollution in the city. Among the mobile sources, trucks followed by public transportation vehicles were identified for higher emissions of SO_2.

Gurjar et al. (2016) identified industries and power plants as major sources of SPM and PM_{10} emissions (20–80%) among the anthropogenic sources in three megacities of India. Other important sources of SPM and PM_{10} in India are small-scale industries, transportation, construction, resuspended soil and road dust, domestic coal burning, and biomass burning (Gurjar et al. 2016). Coal combustion is one of the major sources of PM and NOx in Indian cities, whereas oil combustion is the predominant source of SO_2 (Gurjar et al. 2016). SO_2 concentration in various cities of India has drastically reduced with improvement in fuel quality and strict policies (Gurjar et al. 2016). The transportation sector is one of the major sources of NOx pollution in India contributing almost 50–70% of the total emission followed by industries (10–30%) (Gurjar et al. 2016).

Cavalcante et al. (2017) found light-duty vehicles as the most important emission source of PAHs in particulate matter followed by industrial activities (asphalt and steel production), combustion of wood and coal, and paved road dust in the metropolitan area of Fortaleza-CE, Brazil. Biomass or fossil fuel combustion was identified as the major source of atmospheric PAHs in a tropical rainforest near Manaus, Brazil (Krauss et al. 2005).

In Indo-Gangetic Plain, air pollution is heavily contributed by two-wheelers with two-stroke engines, and pollutants from vehicles are released at a very low height causing heavy pollution at the ground surface, which even gets worse with buildings around the roads that prevent dispersion of pollutants (Verma and Singh 2006). In two tropical coastal cities of Chennai and Visakhapatnam in India, Guttikunda et al. (2015) recorded vehicle exhaust accounting almost 60% and 43% of NOx emission, respectively. Further, 59%, 27%, and 5% contributions of the power plant, industries, and brick kilns, respectively, were calculated for total SO_2 emissions in

Chennai. The sources of O_3 precursors include vehicles and combustion of fossil fuels and biomass as well as natural processes such as NOx production by lightning and VOCs production by the tropical vegetation.

7.4.3 Forest Fire

With increasing human influence and overexploitation of forest resources and rise in temperature with uneven rainfall pattern due to climate change, there is a significant increase in the incidences of forest fires globally (Herawati and Santoso 2011; Le et al. 2014). The number of fire incidence in Vietnam was staggering with 16,086 fires per year during 2004–2012 (Le et al. 2014). Factors such as biomass accumulation, warmer and drier weather conditions, and anthropogenic influences in the tropical forests increase the fire incidences causing loss of biodiversity, degradation of natural habitat, reduction in natural decomposition process, and most importantly release of toxic atmospheric gases in large quantities causing severe air pollution in the tropics (Herawati and Santoso 2011). Le et al. (2014) identified several sources of forest fires in Vietnam such as the burning of crop residues, fire used by local people for hunting wild animals, use of smoke for honey collection, overexploitation of forest products, and trading conflicts for forest resources.

Peat fires are one of the major causes of air pollution in Central Kalimantan, Indonesia, causing a significant rise in PM and other gaseous pollutants during fire events. PM_{10} during fire events reached to a maximum value of 1905 $\mu g\ m^{-3}$, whereas NO_2 concentration averaged 30.5 $\mu g\ m^{-3}$, which was almost 2.24 times higher than in non-fire season. Similarly, SO_2 concentration reached 17.16 times higher during fire compared to non-fire season (Hayasaka et al. 2014).

Chir pine or blue pine (*Pinus roxburghii* Sarg) forests located between 1000 and 1800 m altitude in the Himalayan region are also prone to forest fire with incidences of 3908 fire per $year^{-1}$ (Vadrevu et al. 2012). Vadrevu et al. (2012) estimated total BC emissions of 431 Mg $year^{-1}$ from Himalayan region. Based on fire emission inventory by Heil et al. (2007), 55 teragram (Tg) of PM and 1098 Tg of carbon were estimated to be released during 1997 Indonesian vegetation fire. Sukitpaneenit and Kim Oanh (2014) also recorded increases in CO and PM_{10} levels during the events of forest fire in Northern Thailand.

7.5 Air Pollution Impacts on Ecosystem Structure and Function

The diversity of the tropical ecosystems and possible site-specificity together with limited resources for researching environmental problems in most tropical countries hinder better understanding of the problems. Airborne pollutants can affect ecosystems in two major ways: (1) by direct toxicity and (2) indirectly by changing soil nutrient availability. Present knowledge of each of these mechanisms is discussed below.

7.5.1 Direct Toxicity and Other Injurious Effects

Dust particles, aerosols, and other air pollutants are directly adsorbed on the large leaf surfaces of forest vegetation which ultimately influence plant function and structure (Breulmann et al. 2002). The deposition of particulate matter on the leaf surface causes blockage of solar radiation by shadowing the leaf surface, altering the pigment synthesis and the photosynthetic rate (Pereira et al. 2009). Air pollutant-stressed plants are more vulnerable to pest and fungal attack. In highly industrialized areas of Brazil, Moreira-Nordemann et al. (1988) observed damage to surrounding vegetation, but the direct effects of acidification cannot be adequately separated from those of gaseous pollutants.

Air pollutants such as SO_2, O_3, fluorides, and peroxyacyl nitrate (PAN) damage the leaves of plants. Chronic injury occurs on exposure of plants to low concentration of SO_2 at which the rate of accumulation of the ion is slow. The cells oxidize the sulfite ions and injury does not occur until sufficient sulfate ions get accumulated. This type of chronic injury is characterized by a general chlorotic appearance of the leaves. Cells are not directly harmed, but the chlorophyll is bleached which appears as a mild chlorosis or yellowing of the leaf or a silvering or bronzing of the lower leaf surface without necrosis (Agrawal and Agrawal 1999). Brownish necrotic streaks confined to the middle and upper laminar regions on *Panicum miliaceum* plant leaves were observed after 8 days of exposure with 0.5 ppm SO_2 (Agrawal et al. 1983). Shaw et al. (1993) reported the development of needle necrosis in Scots pine (*Pinus sylvestris* L.) due to SO_2 exposure of 34–58 $\mu g\ m^{-3}$. The acute injury was, however, resulted due to absorption of lethal quantities of SO_2 as full grayish green water-soaked areas which later converted as marginal or intercostal areas of dead tissues. In most plant species, these injuries develop as bleached areas, which upon drying or becoming dead or necrotic areas fall out leaving a ragged appearance to the leaf. In case of severe injury, abscission layer develops at the base of the petiole causing premature fall of the leaves (Mudd 2012). Pandey and Agrawal (1994) also recorded leaf injury symptoms in the form of bifacial chlorosis and necrosis mainly toward the tip and margin of the leaves in three tropical trees exposed to urban air pollution. Fluoride damages the edges of plants as brown or black pigmentation (Klumpp et al. 1996). Peroxyacyl nitrate causes a condition known as a silver leaf, in which the underside of the leaves turns silvery white or bronze (Oka et al. 2004).

As a strong oxidant, O_3 causes several types of symptoms including chlorosis and necrosis. The common symptoms associated with O_3 exposure include flecks (tiny light-tan irregular spots less than 1 mm diameter), stipples (small darkly pigmented areas approximately 2–4 mm diameter), bronzing, and reddening. O_3 damage on leaves appears as mottled spots which may be yellow, black, or brown. If the damage by O_3 is severe enough, the plant drops its leaves altogether. The reddish-brown stipples that develop on the leaves are the result of accumulation of black or red pigments in dead cells of the palisade tissue (Cho et al. 2011). Sanders et al. (1992) have also observed damaged chloroplast membranes, plasmalemma, and tonoplasts as a result of O_3 exposure. Swelling of thylakoid membranes leading to the breakdown of chloroplast integrity was shown to be the result of O_3 exposure (Crang and McQuattie 1986). Chaudhary and Agrawal (2014) also recorded O_3-induced visible foliar injury symptoms as small pale yellow and brown flecks on clover (*Trifolium alexandrinum*).

7.5.2 Changing Acidity of the Tropical Soils

Air pollution can cause severe acidity to soil due to the presence of acidic ions (Tian and Niu 2015). In the acidic soil, aluminum (Al) dominates the exchange complex up to a pH between 5.0 and 6.0. Al precipitates at pH values greater than 6.0 and causes Al toxicity which alters soil properties and root growth (Kunhikrishnan et al. 2016).

The tropical soils vary with regard to different types of ion exchange complexes. In many soils, Al and Fe oxides and some clay are responsible for anion absorption that influences the direct impact of sulfate loading and associated problems like phosphorus immobilization. Both phosphate and sulfate compete for anion absorption sites in tropical clay soils (Johnston and Chrysochoou 2016). Acidification of soils can lead to increased weathering rates of minerals, leaching of bases, and solubilization of trace metals. Soils having lower cation exchange capacity (Sanhueza et al. 1988) and low exchangeable Ca are often considered sensitive to acidification. Further increases in Al saturation and concentration in the soil solution are the major consequences of acidification.

Two types of soils most likely to be sensitive due to deposition of acidic or acidifying substances from the atmosphere are soils that are already acidic in nature and in which Al could be mobilized by additional anion leaching and soils having a substantial reduction in base saturation (Kunhikrishnan et al. 2016). The rapid deforestation in the tropics is also an important acidifying mechanism. Emissions of gases by burning of biomass, export of bases with biomass harvest, and increased leaching of acid-neutralizing cations are important factors in enhancing the soil acidification (Crutzen and Andreae 2016).

7.5.3 Effects on Litterfall and Decomposition

Studies related to the effect of air pollution on litterfall and decomposition in the tropical forests, in grasslands, and in vegetation growing in urban and other remote areas are limited. Decomposition is affected both by deposition of N and nutrients by air pollution, which alter soil properties, and by changing the foliar chemistry of trees. Emissions from industries and traffic alter the natural biogeochemical cycles, decomposition, nutrient balance, and soil fertility. Ferreira et al. (2017) found the influence of air pollution on the rate of litterfall decomposition in Guarapiranga forest in São Paulo, Brazil, and attributed these variations due to the proximity of air-polluting sources like vehicular and industrial activities around the forest site altering the dynamics of decomposition. Influence of air pollution from the urban area of São Paulo, Brazil, on litterfall stock and decomposition in the forest has been reported by Ferreira et al. (2017). It was observed that heavy metals (HMs) transported from the urban areas are the major causes of such changes apart from other factors. Singh et al. (2004) found a negative association between litter decomposition and air pollution around an industrial area in a dry tropical region of India; however, the turnover time of nutrients in the decomposing litter was higher at the site receiving maximum atmospheric depositions. Among the air pollutants, SO_2 showed significant negative correlation with mean annual litterfall. All the evidence clearly indicated that higher atmospheric depositions and gaseous and particulate pollutants significantly influence decomposition rates and nutrient cycling.

Powers and Salute (2011) compared the effect of the addition of nutrients on the decomposition of two leaf litter types from tropical dry forest trees (*Quercus oleoides* and *Gliricidia sepium*). The outcome of the experiment varied with the type of nutrient and species. The decomposition rate was enhanced with the addition of P and Zn, whereas Mg and N delayed the process while K and Ni showed no significant role in litter decomposition (Table 7.4).

7.5.4 Plant Biodiversity

Changes in species richness, dominance, evenness, density, and abundance are the first markers of a shift in biodiversity. To assess the impact of air pollution on lichen diversity, Shukla and Upreti (2011) explored lichen biodiversity in the city of Pauri and Srinagar in Garhwal Himalayas, Uttarakhand, and found a decline in lichen diversity at sites with higher pollution load. The dominance of physcoid lichens (pollution-tolerant) compared to parmeliod lichens (less pollution-tolerant) at polluted sites indicated a strong effect of air pollution on lichen diversity. The decrease in population of fruticose lichens which only accounted for 6% of the total diversity from all the studied sites was implicated with the deteriorating air quality in the study area. Reductions in the number of phorophytes (any plant on which an epiphyte grows) and anthropogenic factors (higher NOx and HMs by vehicles) were identified as the main causes behind the observed pattern at the urban site, which also showed the dominance of nitrophilous lichen (Shukla and Upreti 2011).

In seven cities of the subtropical region of Brazil, Koch et al. (2016) observed a significant effect of air pollution on lichen community composition and vitality. Fine PM was identified as a major air pollutant, which negatively influenced lichen community with NOx, Cu, and Mn. Apart from air pollutants, land uses and higher population density were also identified as major factors affecting lichen diversity as increases in surrounding urban areas were negatively correlated with lichen vitality (Table 7.4). Among the different traits used for assessment of disturbance in environmental gradient on lichen diversity, photobiont vitality (percentage of photobiont cells) was found to be more sensitive trait compared to species richness or cover (Koch et al. 2016).

Singh et al. (1994) conducted a field study around two coal-fired thermal power plants (TPP) to analyze the impact of emissions on the structure of herbaceous

Table 7.4 Summary of studies related to air pollution and plant responses

Location	Study site	Plant	Parameter	Outcome	References
Aligarh, India	Thermal power plant	*R. tuberosa*	Photosynthesis and stomatal conductance	Reduction in photosynthetic and stomatal conductance at higher pollution load	Nighat et al. (2000)
Aligarh, India	Thermal power plant	*R. tuberosa*	Stomatal characteristics	Reduction in length and width of stomata, length of the stomatal pore, stomatal density	Nighat et al. (2000)
Birbhum district, India	Forest	*S. robusta*	Transpiration index	Negative association between dust fall and transpiration index	Saha and Padhy (2012)
Hyderabad, India	Chamber experiment	*F. infectoria* *P. pinnata*	RuBP carboxylase activity and net photosynthesis	Negative association between O_3 exposure and RuBP carboxylase activity and net photosynthesis	Chapla and Kamalakar (2004)
Lucknow City, India	Urban	*C. procera*	Foliar sulfate accumulation	*C. procera* as a sulfate accumulator plant	Singh et al. (1995)
Lucknow, India	Urban	*T. nerifolia*	Stomatal size stomatal frequency	Reduction in stomatal size and increase in stomatal frequency at higher pollution load	Verma and Singh (2006)
Lucknow, India	Urban	*T. nerifolia*	Leaf area	Reduction in leaf area around traffic emission	Verma and Singh (2006)
Nagpur, India	Urban	*A. indica*	Ascorbic acid	Positive association between ascorbic acid content and pollution load	Ninave et al. (2001)
Pauri City, India	Urban	*P. hispidula*	Metal accumulation	Higher bioaccumulation of Fe, Cr, Cu, Zn, Ni, and Pb	Shukla and Upreti (2007)
Sambalpur, India	Highway	*I. carnea*	Dust accumulation and tree characteristics	The roughness of leaf surface, the small size of petiole, and smaller height of plants favor dust accumulation	Prusty et al. (2005)
Varanasi, India	Urban	*P. longifolia* *P. guajava* *M. indica* *F. benghalensis*	Carotenoids Ascorbic acid LDMC	Positive association between carotenoids Ascorbic acid LDMC with increasing pollution load	Mukherjee and Agrawal (2016)

(continued)

Table 7.4 (continued)

Location	Study site	Plant	Parameter	Outcome	References
Varanasi, India	Urban	13 tropical tree species	Leaf functional traits	PM and O_3 maximally influenced leaf functional traits in tree species	Mukherjee and Agrawal (2018)
Philippines	Urban	*P. indicus* *E. orientalis*	Photosynthesis and stomatal conductance	Increase in photosynthetic and stomatal conductance at higher pollution load	Baek and Woo (2010)
Sarawak, Malaysia	Forest	*D. lanceolata* *Macaranga* spp.	Foliar accumulation of 21 heavy metals	Higher metal accumulation in *D. lanceolata* compared to *Macaranga* spp.	Breulmann et al. (2002)
Cubatão City, Brazil	Atlantic rainforest	*T. pulchra*	Photosynthesis	Reduction in photosynthesis at higher pollution load	Moraes et al. (2002)
Cubatão, Brazil	Atlantic rainforest	*T. pulchra*	Carbon assimilation	Negative association between carbon assimilation and pollution load	Moraes et al. (2002)
Cubatão City, Brazil	Atlantic rainforest	*T. pulchra*	Chlorophyll a, Chl a/b ratio, and foliar F, N, and S concentrations	Positive association between chlorophyll a, Chl a/b ratio, and foliar F, N, and S concentrations with pollution load	Moraes et al. (2002)
Cubatão, Brazil	Atlantic forest	*T. pulchra*	Growth parameters (height, diameter, leaf and root biomass, leaf area, and whole plant biomass)	Reduction in all growth parameters at higher pollution load	Moraes et al. (2003)
Manaus, Brazil	Rainforest	10 species	PAHs in leaves, bark, twigs, stem, and wood	PAHs accumulation leave > bark > twigs > stem > wood	Krauss et al. (2005)
Port Harcourt, Nigeria	Urban	*A. cordifolia*, *M. paradisiaca*, *M. esculenta*	Stem, leaf, and petiole anatomy	Increase in number of vascular bundles at higher pollution load	Ajuru and Friday (2014)

communities. The importance value index (IVI) estimates indicated the dominance of a few plant species like *Cassia tora*, *Cynodon dactylon*, and *Dichanthium annulatum* at the sites receiving higher pollution load. On the other hand, *Paspalidium flavidum*, *Phyllanthus simplex*, and *Rungia repens* were more dominant at less polluted sites. However, some plant species like *Alysicarpus monilifer*, *Convolvulus pluricaulis*, and *Desmodium triflorum* were more or less uniformly distributed. Singh et al. (1994) also recorded significant negative correlation between SO_2 concentrations in air and species diversity of herbaceous communities in the dry tropical environment. According to Narayan et al. (1994), the successful survival of a species in the polluted area may be due to adequate biomass formation and their suitable structure, the ability to survive the lasting impact of pollutants, and the ability to reproduce under the pollution stress. Thus, the species having higher IVI values at the heavily polluted sites showed tolerance to the existing pollution load and were classified as "resistant species." Pandey et al. (2014) reported *Achyranthes aspera*, *Convolvulus alsinoides*, *D. annulatum*, *Eclipta alba*, and *Solanum nigrum* as sensitive species under the pollution stress due to coal mining activities. Species such as *Eragrostis cynosuroides* and *Setaria glauca* were identified as polluphilic species as these were only present at the polluted sites.

Pandey et al. (2014) reported changes in IVI values for sensitive and tolerant species due to air pollution around Jharia and Raniganj coalfields in India. Authors also recorded reductions in species richness of both herbaceous and woody species with increasing pollution load. Pandey et al. (2014) also observed the negative effect of SO_2 on woody vegetation around coal mine areas in Jharia and Raniganj coalfields in India. Distribution of *Ficus religiosa*, *Ficus benghalensis*, *Psidium guajava*, and *Butea monosperma* around coal mine areas was directly related to atmospheric air pollutants such as TSP, SO_2, and NO_2. Based on plant density around coal mining areas, Pandey et al. (2014) identified *F. religiosa*, *F. benghalensis*, *P. guajava*, *B. monosperma*, *Corymbia citriodora*, *M. indica*, and *Terminalia arjuna* as the most resistant species. Narayan et al. (1994) assessed the vegetation characteristics downwind of an aluminum factory in Renukoot, India, and found that long-term emissions of fluoride, SO_2, and PM have altered soil characteristics, which significantly impacted the local vegetation. Woody vegetation around the area was more influenced compared to herbaceous vegetation. *E. alba*, *C. tora*, *S. nigrum*, and *Bothriochloa pertusa* were identified as the most sensitive species, whereas *Zornia diphylla*, *Digitaria sanguinalis*, *Dactyloctenium aegyptium*, *S. glauca*, and *Eragrostis tenella* were only recorded at the most polluted sites around the factory (Narayan et al. 1994).

Zvereva et al. (2008) found differences between effects of air pollution on plant species richness among the four terrestrial biomes (boreal and temperate forests, deserts, and tropical savanna). Lack of data from the tropical region is a major limitation to compare differences between the effects of air pollution on biodiversity from different regions of the world (Zvereva et al. 2008). Studies related to effects of N deposition on biodiversity in the tropical and subtropical regions having several biodiversity hotspots are yet to be investigated.

7.6 Air Pollution Impacts on Plant Performance

7.6.1 Growth and Morphology

Both biotic and abiotic stress tolerance in plants depend upon the ability of the plant species to regulate between growth and defense response. Higher allocation of energy in defense response may lead to lower growth, which results in changes in morphological characteristics. In saplings of *Tibouchina pulchra*, one of the most common tree species in the tropical Atlantic forest, reductions in growth parameters such as height, diameter, leaf and root biomass, leaf area, and whole plant biomass were recorded at the sites with higher air pollution load in Cubatão, Brazil (Moraes et al. 2003). Reductions in plant height, basal diameter, canopy area, root-shoot ratio, and plant biomass were also recorded by Pandey and Agrawal (1994) in a tropical urban environment with maximum reduction in *Delonix regia* followed by *C. fistula* and *Carissa carandas*. Ajuru and Friday (2014) studied the effects of PM on the anatomy of three tropical plants, *Alchornea cordifolia*, *Musa paradisiaca*, and *Manihot esculenta*, in Port Harcourt, Nigeria, and found significant effects of pollution load with an increase in the number of vascular bundles although no effect was observed in cortex, epidermis, and pith tissues (Table 7.4).

To avoid air pollution effects from traffic, plants employ different strategies such as reductions in leaf area, specific leaf area, leaf dry matter content, and leaf water content (Mukherjee and Agrawal 2016; Verma and Singh 2006). Verma and Singh (2006) recorded reduction in leaf area by 15–40% in *Thevetia nerifolia* and *F. religiosa* under higher automobile exhaust emission. Prusty et al. (2005) investigated the dust accumulation potential in vegetation near the national highway at Sambalpur, Orissa, India, and found higher dust accumulation in the leaves of *Ipomoea carnea*, *Tabernaemontana divaricata*, and *Pongamia pinnata* compared to *F. religiosa* and *Quisqualis indica*. The observed outcomes were correlated with the roughness of leaf surface, the small size of petiole, and the smaller height of plants favoring higher dust accumulation.

Leaf structure, geometry, leaf epidermal length, cuticle thickness, phyllotaxy of leaf, height of the plant, canopy size, nature of the tree, and leaf forms determine plant response to higher dust pollution (Nighat et al. 2000; Prusty et al. 2005). Prusty et al. (2005) found that dust accumulation was more in plants with smaller height compared to taller plants. Nighat et al. (2000) recorded reductions in leaf area, length and width of stomata, length of the stomatal pore, and stomatal density in *Ruellia tuberosa* due to thermal power plant emission in Aligarh, India, whereas stomatal index was unaffected (Table 7.4).

7.6.2 Plant Physiology

For proper growth and development, plants have to regulate their function according to variations in environmental conditions. Regulating physiological activities in accordance with environmental conditions provides a plant to maximize the resource utilization and maintain the growth process. Variations in physiological activities such as photosynthesis, stomatal conductance, respiration, transpiration, and photosynthetic efficiency are known under air pollution stress conditions (Baek and Woo 2010; Moraes et al. 2002).

Moraes et al. (2002) found a reduction in photosynthesis rate in *T. pulchra* at polluted site compared to control in Cubatão, Brazil, indicating the influence of air pollution in damaging stomata and pathways related to light and CO_2 fixation. Similarly, reductions in photosynthetic rate and stomatal conductance were observed in *R. tuberosa* plants receiving thermal power plant emission in Aligarh, India (Nighat et al. 2000). In contrast, *Pterocarpus indicus* and *Erythrina orientalis* showed enhancement in photosynthetic rate as well as stomatal conductance at sites with higher pollution load (Baek and Woo 2010). The contradictory response showed compensation ability by increasing photosynthetic rate in response to pigment degradation and high carbon utilization of the test plant at higher pollution load (Baek and Woo 2010).

Saha and Padhy (2012) compared the effects of stone dust deposition on the rate of transpiration in *Shorea robusta* plant in the polluted and non-polluted forest of Birbhum district, India. Transpiration rate was declined at polluted forest due to damage to leaf and deposition of dust particles at the stomatal surface. Higher SPM concentrations and dust fall also lowered the transpiration index (Table 7.4). Carbon assimilation declined at higher pollution load, whereas stomatal conductance remained unchanged in *T. pulchra* at Cubatão, Brazil (Moraes et al. 2002). Traffic-related pollutants in high concentrations modified plant physiology through altering the stomatal response. Verma and Singh (2006) recorded 70% decline in stomatal size, whereas stomatal frequency increased up to 50% in *T. nerifolia* at sites with higher pollution load. Alterations in foliar surface configuration were also recorded in plants exposed to traffic-related pollutants. Chapla and Kamalakar (2004) found a significant negative effect of O_3 fumigation (40–120 ppb) on RuBP carboxylase activity with percent reductions varying from 10 to 32 in *Bauhinia variegata* followed by 10–23 in *Ficus infectoria* and 9–15 in *P. pinnata*. Percent reduction in photosynthesis was maximum in *F. infectoria* (16–39) followed by *P. pinnata* (7–31) and *B. variegata* (6–26) due to O_3 fumigation.

7.6.3 Leaf Functional Traits

Leaf functional traits are most widely used in the assessment of plant response to biotic and abiotic stresses. Leaf functional traits such as leaf mass per area or specific leaf area, tissue thickness, leaf area, foliar C and N contents, photosynthetic pigments, antioxidants, and relative water content vary in relation to the environmental conditions (Table 7.4). Therefore, these characteristics provide the opportunity to correlate the variations in relation to changing environment under natural as well as in modified systems. The increases in antioxidants and defense-related metabolites are the primary response of plants against air pollution stress. Verma and Singh (2006) recorded increases in foliar cysteine content at higher traffic pollution load with maximum percent increases of 44% and 30%, respectively, in *F. religiosa* and *T. nerifolia*. Ninave et al. (2001) also observed an increase in ascorbic acid content in plants at the polluted environment.

Verma and Singh (2006) recorded reductions in Chl a, Chl b, and carotenoid contents by 6–64% at higher pollution load in both *T. nerifolia* and *F. religiosa* in Lucknow, India, indicating a decrease in productivity at a higher load of air pollutants. A similar trend was also observed for foliar protein content in both plants indicating inhibition in protein synthesis or damage to existing protein although percent reduction in protein content was lower than the photosynthetic pigments indicating more sensitivity of pigments to air pollutants (Verma and Singh 2006). In contrast, Ninave et al. (2001) found an increase in Chl content in all the studied plants except for *Polyalthia longifolia* at polluted environments. Based on responses of plants, *Azadirachta indica* was identified as tolerant species in the urban environment of Nagpur, India. Due to foliar dust deposition, declines in both chlorophyll and carotenoid contents were observed in most of the plants. Chl a as well as total chlorophyll showed maximum reductions of 46.5% and 45.4% in *F. benghalensis* due to dust deposition (Prusty et al. 2005). Dust contains many organic and inorganic compounds which on entry into plant tissues cause increases in reactive oxygen species (ROS) and primarily damage to the membrane and pigments. Moraes et al. (2003) found a decline in ascorbate concentration in *T. pulchra* while increases in chlorophyll a, Chl a/b ratio, and foliar F, N, and S concentrations at polluted site compared to control in Cubatão, Brazil. The increase in pigment concentration is attributed to increasing leaf N content, indicating the utilization of atmospheric NOx and NH_4^+ in N assimilation. Pandey and Agrawal (1994), however, recorded reductions in chlorophyll, ascorbic acid, and nitrogen contents in leaves, whereas sulfate content showed increment with increasing pollution load in three tropical tree species. Baek and Woo (2010) assessed the responses of *P. indicus* and *E. orientalis* growing in four sites within the Philippines with differential pollution load. In both the test plants, chlorophyll content and Chl a/b ratio declined at sites with higher pollution load, whereas antioxidative enzymes showed increments. A decrease in the size of starch grains in plant leaves indicated consumption of stored carbon at the expense of defense response (Baek and Woo 2010).

Moraes et al. (2002) found higher variations in water-soluble thiols and peroxidase activity at higher pollution load compared to ascorbic acid in *P. guajava* and *P. cattleyanum* around the industrial complex of Cubatão, SE Brazil. Mukherjee and Agrawal (2016) also observed an increase in ascorbic acid content in four tropical tree species (*M. indica*, *P. longifolia*, *F. benghalensis*, and *P. guajava*) growing at higher pollution load at different parks in Varanasi City, India. Baek and Woo (2010) observed an increase in thickness of cell wall in response to pollution load in *P. indicus*, which may be due to increase in defense-related metabolites to protect membrane damage (Fig. 7.4).

In the urban area of Varanasi City, Mukherjee and Agrawal (2016) found an increase in photosynthetic pigments with an increase in air pollution load in *M. indica* and *P. longifolia*, whereas *F. benghalensis* and *P. guajava* showed marked reductions. Carotene content showed a significant increase with higher pollution

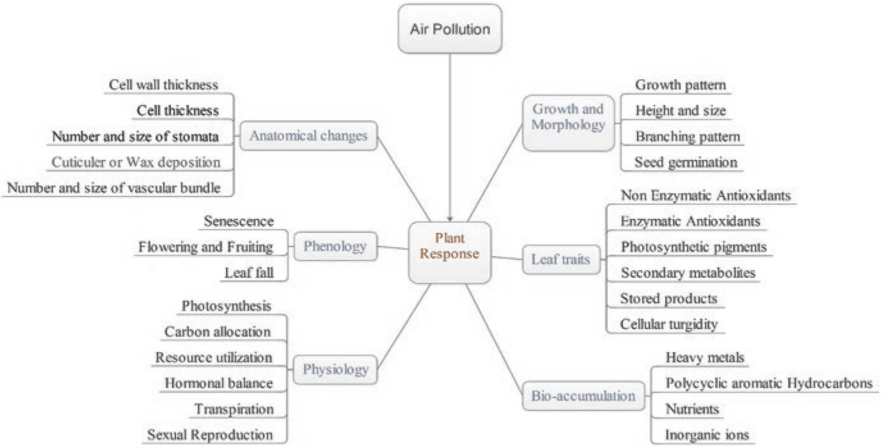

Fig. 7.4 Different responses of plants in relation to air pollution tolerance

load in all the studied plant species with a maximum increase of 88% in *P. longifolia*. Chl a/b ratio as well as TChl/Car ratio showed reductions in most plants with increasing pollution load. Leaf area, leaf dry matter content (LDMC), relative water content (RWC), and specific leaf area (SLA) were suggested to be important markers for stress tolerance under air pollution stress (Mukherjee and Agrawal 2016; Pandey and Agrawal 1994). Most studies have identified increases or maintenance of leaf biomass allocation and cellular turgidity at higher pollution load in tolerant plant species (Fig. 7.4) (Mukherjee and Agrawal 2016).

7.6.4 Foliar Bioaccumulation

Plants have the ability to accumulate, detoxify, and concentrate pollutants (De França et al. 2004; Mukherjee et al. 2016) that enable them to tolerate harsh environments. PM constituents like HMs, PAHs, and ions after deposition can directly enter the leaf through stomata or via absorption through root after getting deposited in the soil. Nakazato et al. (2016) assessed foliar accumulation and enrichment factor for 36 elements in *P. guajava* and *T. pulchra* trees of the Atlantic rainforest and found *P. guajava* as a better accumulator of metals with distinct spatial-temporal variations with pollution load. Authors have also suggested the use of *P. guajava* for biomonitoring of toxic elements. De França et al. (2004) assessed the foliar metal accumulation in native trees of the tropical Atlantic forest located at the southwest portion of São Paulo State, Brazil, and found the lowest bioaccumulation of metals in *Euterpe edulis*, with exception of Cs and Zn, whereas the highest concentration of Se was found in the leaves of *Tetrastylidium grandifolium* and *Eugenia cuprea*. *Hyeronima alchorneoides* tree was identified as potential Co hyperaccumulator compared to other trees with 80–300 times higher accumulation of Co (Table 7.4).

Moraes et al. (2002) reported higher bioaccumulation of F, S, and N in *P. guajava* compared to *P. cattleyanum* and *M. indica* around the industrial complex of Cubatão, Brazil (Moraes et al. 2002). Authors also found 2–2.6 times higher accumulation of F in *P. guajava* compared to other plants. Breulmann et al. (2002) analyzed 21 chemical elements in emergent (*Dryobalanops lanceolata*) and pioneer species (*Macaranga* spp.) in a tropical forest in Sarawak, Malaysia. Higher bioaccumulation was recorded in pioneer species compared to emergent species, which may be due to higher physiological activities in the former species. Higher bioaccumulation was also found due to extensive forest fire raging in Borneo and other parts of Southeast Asia, which increased the atmospheric load of certain elements. Singh et al. (1995) identified *Calotropis procera* as the best accumulator plant for Pb and sulfate around roadside receiving auto exhaust pollution in Lucknow City, India. *C. procera* followed by *T. nerifolia*, *D. sissoo*, and *Eucalyptus* sp. showed significant positive correlation between sulfate in leaves and SO_2 in the ambient air, whereas correlations were insignificant for *P. longifolia* and *P. glabra*. Agrawal and Singh (2000) recorded increases in total S content by 41%, 25%, 35%, 27%, 33%, and 24%, respectively, in *M. indica*, *P. guajava*, *C. siamea*, *D. regia*, *Eucalyptus hybrid*, and *Bougainvillea spectabilis* around two thermal power plants in the Sonbhadra district of Uttar Pradesh, India, and found direct correlation with average SO_2 concentrations in the ambient air. Plants growing near the power plants also showed higher bioaccumulation of Ca, K, Mn, Fe, Cd, Pb, and Ni, whereas the total N content was reduced.

High metal bioaccumulation was also reported in lichens (Shukla and Upreti 2007). At five different sites of Pauri City in Garhwal Himalaya, India, metal accumulation in *Phaeophyscia hispidula*, a common foliose lichen, was found higher for Fe, Cr, Cu, Zn, Ni, and Pb, which correlated well with traffic load at different sites. PAHs in leaves, bark, twigs, stem, and wood were estimated in a tropical rainforest near Manaus, Brazil, by Krauss et al. (2005). The order of PAHs accumulation was maximum in leaves followed by bark, twigs, stem, and wood. The study also identified that atmosphere is the major source of PAHs accumulation in plants.

Most of the studies analyzed in this review article showed a negative effect of air pollutants on growth and morphology with reductions in leaf area, height, biomass, size of stomata, and overall canopy size. Physiological parameters also showed a significant negative effect of air pollutants with variable responses in different plant species. Most of the tolerant species showed increases in enzymatic and nonenzymatic antioxidants with an increase in pollution load to cope up with the oxidative stress. The sensitive species, however, showed marked reductions in most of the studied leaf functional traits. Significant variability in bioaccumulation pattern among different plant groups and species was also ascertained. Higher accumulation in most of the plant species under highly polluted environment indicates their tolerance which depends upon the nature of the plant, leaf characteristics, biochemistry, and physiology.

7.7 Conclusions

The tropical and subtropical regions harbor highly diverse ecosystems of the world. Overexploitation of resources and increasing demands for natural resources have drastically altered the dynamics of the tropical ecosystems. Air pollution is one of the major root causes of such changes in the tropics. Air pollution in the form of particulate matter, O_3, nitrogen deposition, and hydrocarbons has deteriorated the air quality in the tropics. Both coarse and fine PM are identified as major air pollutants in Southeast Asia, in South America, and in West and East Africa. NO_2 and O_3 are the major gaseous air pollutants in the tropics, whereas SO_2 concentrations are significantly lowered down in recent time. In most of the tropical region, air pollutants are identified as a major stress factor for vegetation. The effects are more significant in urban areas where air pollutant sources such as industrial, traffic, and combustion activities are higher. Apart from anthropogenic emissions, forest fires and prevailing meteorological conditions in the tropics also contribute to air pollution load. Air pollutants influence plants diversity by altering soil quality, litterfall, and decomposition rate. Vegetation responds to air pollution load by modifying the growth and morphological pattern, physiological activities, varying leaf functional traits, and bioaccumulation of pollutants. Lack of air pollution response studies in the tropical regions and different vegetation types limited the scope of this chapter to identify a broad range of air pollution effects in the tropics, although current evidence clearly highlights the negative influence of air pollution in different tropical and subtropical regions of the world. To estimate future effects of air pollution in the tropics, it is necessary to estimate projections of future emission patterns, air quality, deposition pattern, biodiversity, and pollution response assessment in the tropics.

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References

Agrawal M, Agrawal SB (1999) Effects of air pollution on plant diversity. In: Agrawal SB, Agrawal M (eds) Environmental pollution and plant responses. CRC/Lewis Publishers, New York, pp 154–176

Agrawal M, Singh J (2000) Impact of coal power plant emission on the foliar elemental concentrations in plants in a low rainfall tropical region. Environ Monit Assess 60:261–282. https://doi.org/10.1023/A:1006135317896

Agrawal M, Nandi PK, Rao DN (1983) Ozone and sulphur dioxide effects on Panicum miliaceum plants. Bull Torrey Bot Club 110:435–441

Ajuru MG, Friday U (2014) Effects of particulate matter on the anatomy of some tropical plants (Alchonea cordifolia, Musa paradisiaca, and Manihot esculenta). Int J Sci Technol Res 3:304–308

Alejo D, Morales MC, De La Torre JB et al (2013) Seasonal trends of atmospheric nitrogen dioxide and sulfur dioxide over North Santa Clara, Cuba. Environ Monit Assess 185:6023–6033. https://doi.org/10.1007/s10661-012-3003-4

Awang NR, Elbayoumi M, Ramli NA, Yahaya AS (2016) Diurnal variations of ground-level ozone in three port cities in Malaysia. Air Qual Atmos Health 9:25–39. https://doi.org/10.1007/s11869-015-0334-7

Ayi Fanou L, Mobio TA, Creppy EE et al (2006) Survey of air pollution in Cotonou, Benin – Air monitoring and biomarkers. Sci Total Environ 358:85–96. https://doi.org/10.1016/j.scitotenv.2005.03.025

Azmi SZ, Latif MT, Ismail AS et al (2010) Trend and status of air quality at three different monitoring stations in the Klang Valley, Malaysia. Air Qual Atmos Health 3:53–64. https://doi.org/10.1007/s11869-009-0051-1

Badarinath KVS, Kharol SK, Reddy RR et al (2009) Black carbon aerosol mass concentration variation in urban and rural environments of India—a case study. Atmos Sci Lett 10:29–33. https://doi.org/10.1002/asl.206

Baek SG, Woo SY (2010) Physiological and biochemical responses of two tree species in urban areas to different air pollution levels. Photosynthetica 48:23–29. https://doi.org/10.1007/s11099-010-0005-8

Baumbach G, Vogt U, Hein KRG et al (1995) Air pollution in a large tropical city with a high traffic density—results of measurements in Lagos, Nigeria. Sci Total Environ 169:25–31. https://doi.org/10.1016/0048-9697(95)04629-F

Baxla SP, Roy AA, Gupta T et al (2009) Analysis of diurnal and seasonal variation of submicron outdoor aerosol mass and size distribution in a Northern Indian city and its correlation to black carbon. Aerosol Air Qual Res 9:458–469. https://doi.org/10.4209/aaqr.2009.03.0017

Begum BA, Hossain A, Nahar N et al (2012) Organic and black carbon in PM2.5 at an urban site at Dhaka, Bangladesh. Aerosol Air Qual Res 12:1062–1072. https://doi.org/10.4209/aaqr.2012.05.0138

Beig G, Singh V (2007) Trends in tropical tropospheric column ozone from satellite data and MOZART model. Geophys Res Lett 34:1–5. https://doi.org/10.1029/2007GL030460

Breulmann G, Markert B, Weckert V et al (2002) Heavy metals in emergent trees and pioneers from tropical forest with special reference to forest fires and local pollution sources in Sarawak, Malaysia. Sci Total Environ 285:107–115. https://doi.org/10.1016/S0048-9697(01)00899-3

Cavalcante RM, Rocha CA, De Santiago ÍS et al (2017) Influence of urbanization on air quality based on the occurrence of particle-associated polycyclic aromatic hydrocarbons in a tropical semiarid area (Fortaleza-CE, Brazil). Air Qual Atmos Health 10:437–445. https://doi.org/10.1007/s11869-016-0434-z

Chapla J, Kamalakar JA (2004) Metabolic responses of tropical trees to ozone pollution. J Environ Biol 25(3):287–290

Chaudhary N, Agrawal SB (2014) Role of gamma radiation in changing phytotoxic effect of elevated level of ozone in Trifolium alexandrinum L. (Clover). Atmos Pollut Res 5:104–112. https://doi.org/10.5094/APR.2014.013

Cho K, Tiwari S, Agrawal SB et al (2011) Tropospheric ozone and plants: absorption, responses, and consequences. In: Whitacre DM (ed) Reviews of environmental contamination and toxicology, vol 212. Springer New York, New York, pp 61–111

Cooper OR, Parrish DD, Ziemke J et al (2014) Global distribution and trends of tropospheric ozone: an observation-based review. Elem Sci Anthr 2:29. https://doi.org/10.12952/journal.elementa.000029

CPCB (2009). Central Pollution Control Board. National ambient air quality standards. http://cpcb.nic.in/National_Ambient_Air_Quality_Standards.php. Accessed 1 May 2017

Crang RE, McQuattie CJ (1986) Qualitative and quantitative effects of acid misting and two air pollutants on foliar structures of Liriodendron tulipifera. Can J Bot 64:1237–1243. https://doi.org/10.1139/b86-170

Crutzen PJ, Andreae MO (2016) Biomass burning in the tropics: impact on atmospheric chemistry and biogeochemical cycles. In: Crutzen PJ, Brauch HG (eds) Paul J. Crutzen: a pioneer on atmospheric chemistry and climate change in the anthropocene. Springer, Cham, pp 165–188

Crutzen PJ, Giedel LT (1983) A two-dimensional photochemical model of the atmosphere. 2: the tropospheric budgets of anthropogenic chlorocarbons {CO}, {CH_4}, {CH_3Cl} and the effect of various {NO_x} sources on tropospheric ozone. J Geophys Res 88:6641–6661. https://doi.org/10.1029/JC088iC11p06641

Dallarosa JB, Mônego JG, Teixeira EC et al (2005) Polycyclic aromatic hydrocarbons in atmospheric particles in the metropolitan area of Porto Alegre, Brazil. Atmos Environ 39:1609–1625. https://doi.org/10.1016/j.atmosenv.2004.10.045

De França EJ, De Nadai Fernades EA, Bacchi MA, Saiki M (2004) Native trees as biomonitors of chemical elements in the biodiversity conservation of the Atlantic Forest. J Atmos Chem 49:579–592. https://doi.org/10.1007/s10874-004-1267-y

Dey S, Di Girolamo L, van Donkelaar A et al (2012) Variability of outdoor fine particulate (PM2.5) concentration in the Indian Subcontinent: a remote sensing approach. Remote Sens Environ 127:153–161. https://doi.org/10.1016/j.rse.2012.08.021

Dionisio KL, Arku RE, Hughes AF et al (2010) Air pollution in Accra neighborhoods: spatial, socioeconomic, and temporal patterns. Environ Sci Technol 44:2270–2276. https://doi.org/10.1021/es903276s

Doumbia EHT, Liousse C, Galy-Lacaux C et al (2012) Real time black carbon measurements in West and Central Africa urban sites. Atmos Environ 54:529–537. https://doi.org/10.1016/j.atmosenv.2012.02.005

Ferreira ML, Portella Ribeiro A, Rodrigues Albuquerque C et al (2017) Air contaminants and litter fall decomposition in urban forest areas: the case of São Paulo – SP, Brazil. Environ Res 155:314–320. https://doi.org/10.1016/j.envres.2017.02.023

Gupta GP, Kumar B, Kulshrestha UC (2016) Impact and pollution indices of urban dust on selected plant species for green belt development: mitigation of the air pollution in NCR Delhi, India. Arab J Geosci 9:1–15. https://doi.org/10.1007/s12517-015-2226-4

Gurjar BR, Ravindra K, Nagpure AS (2016) Air pollution trends over Indian megacities and their local-to-global implications. Atmos Environ 142:475–495. https://doi.org/10.1016/j.atmosenv.2016.06.030

Guttikunda SK, Goel R, Mohan D et al (2015) Particulate and gaseous emissions in two coastal cities—Chennai and Visakhapatnam, India. Air Qual Atmos Health 8:559–572. https://doi.org/10.1007/s11869-014-0303-6

Hayasaka H, Noguchi I, Putra EI et al (2014) Peat-fire-related air pollution in Central Kalimantan, Indonesia. Environ Pollut 195:257–266. https://doi.org/10.1016/j.envpol.2014.06.031

Heil A, Langmann B, Aldrian E (2007) Indonesian peat and vegetation fire emissions: study on factors influencing large-scale smoke haze pollution using a regional atmospheric chemistry model. Mitig Adapt Strateg Glob Chang 12:113–133. https://doi.org/10.1007/s11027-006-9045-6

Herawati H, Santoso H (2011) Tropical forest susceptibility to and risk of fire under changing climate: a review of fire nature, policy and institutions in Indonesia. For Policy Econ 13:227–233. https://doi.org/10.1016/j.forpol.2011.02.006

Heue KP, Coldewey-Egbers M, Delcloo A et al (2016) Trends of tropical tropospheric ozone from 20 years of European satellite measurements and perspectives for the Sentinel-5 Precursor. Atmos Meas Tech 9:5037–5051. https://doi.org/10.5194/amt-9-5037-2016

Hung NTQ, Lee SB, Hang NT et al (2014) Characterization of black carbon at roadside sites and along vehicle roadways in the Bangkok Metropolitan Region. Atmos Environ 92:231–239. https://doi.org/10.1016/j.atmosenv.2014.04.011

Johnston CP, Chrysochoou M (2016) Mechanisms of chromate, selenate, and sulfate adsorption on al-substituted ferrihydrite: implications for ferrihydrite surface structure and reactivity. Environ Sci Technol 50:3589–3596. https://doi.org/10.1021/acs.est.5b05529

Kaur S, Senthilkumar K, Verma VK et al (2013) Preliminary analysis of polycyclic aromatic hydrocarbons in air particles (PM10) in Amritsar, India: sources, apportionment, and possible risk implications to humans. Arch Environ Contam Toxicol 65:382–395. https://doi.org/10.1007/s00244-013-9912-6

Klumpp A, Domingos M, Klumpp G (1996) Assessment of the vegetation risk by fluoride emissions from fertiliser industries at Cubatão. Brazil Sci Total Environ 192:219–228. https://doi.org/10.1016/S0048-9697(96)05298-9

Koch NM, Branquinho C, Matos P et al (2016) The application of lichens as ecological surrogates of air pollution in the subtropics: a case study in South Brazil. Environ Sci Pollut Res 23:20819–20834. https://doi.org/10.1007/s11356-016-7256-2

Krauss M, Wilcke W, Martius C et al (2005) Atmospheric versus biological sources of polycyclic aromatic hydrocarbons (PAHs) in a tropical rain forest environment. Environ Pollut 135:143–154. https://doi.org/10.1016/j.envpol.2004.09.012

Kumar R, Gupta A, Kumari K, Srivastava S (2004) Simultaneous measurements of SO2, NO2, HNO3 and NH3: seasonal and spatial variations. Curr Sci 87:1108–1115

Kumar A, Singh D, Singh BP et al (2015) Spatial and temporal variability of surface ozone and nitrogen oxides in urban and rural ambient air of Delhi-NCR, India. Air Qual Atmos Health 8:391–399. https://doi.org/10.1007/s11869-014-0309-0

Kunhikrishnan A, Thangarajan R, Bolan NS et al (2016) Functional relationships of soil acidification, liming, and greenhouse gas flux. Adv Agron 139:1–71. https://doi.org/10.1016/bs.agron.2016.05.001

Laurance WF, Carolina Useche D, Rendeiro J et al (2012) Averting biodiversity collapse in tropical forest protected areas. Nature 489:290–294

Le TH, Nguyen TNT, Lasko K et al (2014) Vegetation fires and air pollution in Vietnam. Environ Pollut 195:267–275. https://doi.org/10.1016/j.envpol.2014.07.023

Mallik C, Lal S (2014) Seasonal characteristics of SO2, NO2, and CO emissions in and around the Indo-Gangetic Plain. Environ Monit Assess 186:1295–1310. https://doi.org/10.1007/s10661-013-3458-y

Marinoni A, Cristofanelli P, Laj P et al (2010) Aerosol mass and black carbon concentrations, a two year record at NCO-P (5079 m, Southern Himalayas). Atmos Chem Phys 10:8551–8562. https://doi.org/10.5194/acp-10-8551-2010

Masih J, Singhvi R, Kumar K et al (2012) Seasonal variation and sources of polycyclic aromatic hydrocarbons (PAHs) in indoor and outdoor air in a semi arid tract of Northern India. Aerosol Air Qual Res 12:515–525. https://doi.org/10.4209/aaqr.2011.11.0192

Mmari AG, Potgieter-Vermaak SS, Bencs L et al (2013) Elemental and ionic components of atmospheric aerosols and associated gaseous pollutants in and near Dar es Salaam, Tanzania. Atmos Environ 77:51–61. https://doi.org/10.1016/j.atmosenv.2013.04.061

Moraes RM, Klumpp A, Furlan CM et al (2002) Tropical fruit trees as bioindicators of industrial air pollution in southeast Brazil. Environ Int 28:367–374. https://doi.org/10.1016/S0160-4120(02)00060-0

Moraes RM, Delitti WBC, Moraes JAPV (2003) Gas exchange, growth, and chemical parameters in a native Atlantic forest tree species in polluted areas of Cubatão, Brazil. Ecotoxicol Environ Saf 54:339–345. https://doi.org/10.1016/S0147-6513(02)00067-2

Moreira-Nordemann LM, Forti MC, Lascio VALD et al (1988) Acidification in Southeastern Brazil. In: Henning R, Herrera R (eds) Acidification in tropical countries. Wiley, New York, pp 257–296

Mudd JB (ed) (2012) Responses of plants to air pollution. Elsevier

Mukherjee A, Agrawal M (2016) Pollution response score of tree species in relation to ambient air quality in an urban area. Bull Environ Contam Toxicol 96:1–6. https://doi.org/10.1007/s00128-015-1679-1

Mukherjee A, Agrawal M (2017a) A global perspective of fine particulate matter pollution and its health effects. In: Gunther F.A., de Voogt P. (eds) Reviews of environmental contamination and toxicology (continuation of residue reviews), 244:5–51 Springer, Cham.

Mukherjee A, Agrawal M (2017b) World air particulate matter: sources, distribution and health effects. Environ Chem Lett 15:283–309. https://doi.org/10.1007/s10311-017-0611-9

Mukherjee A, Agrawal M (2018) Use of GLM approach to assess the responses of tropical trees to urban air pollution in relation to leaf functional traits and tree characteristics. Ecotoxicol Environ Saf 152:42–54. https://doi.org/10.1016/j.ecoenv.2018.01.038

Mukherjee A, Agrawal SB, Agrawal M (2016) Heavy metal accumulation potential and tolerance in tree and grass species. In: Singh A, Prasad SM, Singh RP (eds) Plant responses to Xenobiotics. Springer Singapore, Singapore, pp 177–210

Nakazato RK, Rinaldi MCS, Domingos M (2016) Tropical trees: are they good alternatives for biomonitoring the atmospheric level of potential toxic elements near to the Brazilian Atlantic Rainforest? Ecotoxicol Environ Saf 134:72–79. https://doi.org/10.1016/j.ecoenv.2016.08.013

Narayan D, Agrawal M, Pandey J, Singh J (1994) Changes in vegetation characteristics downwind of an aluminium factory in India. Ann Bot 73:557–565. https://doi.org/10.1006/anbo.1994.1069

Nighat F, Mahmooduzzafar, Iqbal M (2000) Stomatal conductance, photosynthetic rate, and pigment content in Ruellia tuberosa leaves as affected by coal-smoke pollution. Biol Plant 43:263–267. https://doi.org/10.1023/A:1002712528893

Ninave SY, Chaudhari PR, Gajghate DG, Tarar JL (2001) Foliar biochemical features of plants as indicators of air pollution. Bull Environ Contam Toxicol 67:133–140. https://doi.org/10.1007/s001280101

Nishanth T, Satheesh Kumar MK, Valsaraj KT (2012) Variations in surface ozone and NOx at Kannur: a tropical, coastal site in India. J Atmos Chem 69:101–126. https://doi.org/10.1007/s10874-012-9234-5

Oka E, Tagami Y, Oohashi T, Kondo N (2004) A physiological and morphological study on the injury caused by exposure to the air pollutant, peroxyacetyl nitrate (PAN), based on the quantitative assessment of the injury. J Plant Res 117:27–36. https://doi.org/10.1007/s10265-003-0127-1

Pandey J, Agrawal M (1994) Evaluation of air pollution phytotoxicity in a seasonally dry tropical urban environment using three woody perennials. New Phytol 126:53–61. https://doi.org/10.1111/j.1469-8137.1994.tb07529.x

Pandey J, Agrawal M, Khanam N et al (1992) Air pollutant concentrations in Varanasi, India. Atmos Environ Part B Urban Atmos 26:91–98. https://doi.org/10.1016/0957-1272(92)90040-Y

Pandey B, Agrawal M, Singh S (2014) Coal mining activities change plant community structure due to air pollution and soil degradation. Ecotoxicology 23:1474–1483. https://doi.org/10.1007/s10646-014-1289-4

Panicker AS, Pandithurai G, Safai PD et al (2010) On the contribution of black carbon to the composite aerosol radiative forcing over an urban environment. Atmos Environ 44:3066–3070. https://doi.org/10.1016/j.atmosenv.2010.04.047

Pereira EG, Oliva MA, Kuki KN, Cambraia J (2009) Photosynthetic changes and oxidative stress caused by iron ore dust deposition in the tropical CAM tree Clusia hilariana. Trees - Struct Funct 23:277–285. https://doi.org/10.1007/s00468-008-0275-y

Pereira TS, Beltrami LS, Rocha JAV et al (2013) Toxicogenetic monitoring in urban cities exposed to different airborne contaminants. Ecotoxicol Environ Saf 90:174–182. https://doi.org/10.1016/j.ecoenv.2012.12.029

Powers JS, Salute S (2011) Macro- and micronutrient effects on decomposition of leaf litter from two tropical tree species: inferences from a short-term laboratory incubation. Plant Soil 346:245–257. https://doi.org/10.1007/s11104-011-0815-x

Pozo K, Harner T, Rudolph A et al (2012) Survey of persistent organic pollutants (POPs) and polycyclic aromatic hydrocarbons (PAHs) in the atmosphere of rural, urban and industrial areas of Concepción, Chile, using passive air samplers. Atmos Pollut Res 3:426–434. https://doi.org/10.5094/APR.2012.049

Provençal S, Kishcha P, da Silva AM et al (2017) AOD distributions and trends of major aerosol species over a selection of the world's most populated cities based on the 1st version of NASA's MERRA Aerosol Reanalysis. Urban Clim 20:168–191. https://doi.org/10.1016/j.uclim.2017.04.001

Prusty BAK, Mishra PC, Azeez PA (2005) Dust accumulation and leaf pigment content in vegetation near the national highway at Sambalpur, Orissa, India. Ecotoxicol Environ Saf 60:228–235. https://doi.org/10.1016/j.ecoenv.2003.12.013

Qu Y, An J, He Y et al (2016) An overview of emissions of SO 2 and NO x and the long-range transport of oxidized sulfur and nitrogen pollutants in East Asia. J Environ Sci 44:13–25. https://doi.org/10.1016/j.jes.2015.08.028

Ramachandran S, Rajesh TA (2007) Black carbon aerosol mass concentrations over Ahmedabad, an urban location in western India: comparison with urban sites in Asia, Europe, Canada, and the United States. J Geophys Res Atmos 112:1–19. https://doi.org/10.1029/2006JD007488

Raysoni AU, Sarnat JA, Sarnat SE et al (2011) Binational school-based monitoring of traffic-related air pollutants in El Paso, Texas (USA) and Ciudad Juárez, Chihuahua (Mexico). Environ Pollut 159:2476–2486. https://doi.org/10.1016/j.envpol.2011.06.024

Reddy KK, Naja M, Ojha N et al (2012a) Influences of the boundary layer evolution on surface ozone variations at a tropical rural site in India. J Earth Syst Sci 121:911–922. https://doi.org/10.1007/s12040-012-0200-z

Reddy SKB, Raghavendra Kumar K, Balakrishnaiah G et al (2012b) Potential source regions contributing to seasonal variations of black carbon aerosols over Anantapur in southeast India. Aerosol Air Qual Res 12:340–354. https://doi.org/10.4209/aaqr.2011.10.0159

Rodrigues-Silva F, Santos U de P, Saldiva PHN et al (2012) Health risks and economic costs of absenteeism due to air pollution in São Paulo, Brazil. Aerosol Air Qual Res 12:826–833. https://doi.org/10.4209/aaqr.2011.12.0235

Romero R, Sienra R, Richter P (2002) Efficient screening method for determination of polycyclic aromatic hydrocarbons (PAHs) in airborne particles. Application in real samples of Santiago–Chile metropolitan urban area. Atmos Environ 36:2375–2381. https://doi.org/10.1016/S1352-2310(02)00161-9

Saha DC, Padhy PK (2012) Effect of particulate pollution on rate of transpiration in Shorea robusta at Lalpahari forest. Trees 26:1215–1223. https://doi.org/10.1007/s00468-012-0697-4

Sahu SK, Pandit GG, Puranik VD (2008) Dry deposition of polycyclic aromatic hydrocarbons associated with atmospheric participate matters in an urban site, Mumbai, India. Aerosol Air Qual Res 8:437–446

Salako GO, Hopke PK, Cohen DD et al (2012) Exploring the variation between EC and BC in a variety of locations. Aerosol Air Qual Res 12:1–7. https://doi.org/10.4209/aaqr.2011.09.0150

Salam A, Hossain T, Siddique MNA, Alam AMS (2008) Characteristics of atmospheric trace gases, particulate matter, and heavy metal pollution in Dhaka, Bangladesh. Air Qual Atmos Health 1:101–109. https://doi.org/10.1007/s11869-008-0017-8

Sanders GE, Robinson AD, Geissler PA, Colls JJ (1992) Yield stimulation of a commonly grown cultivar of Phaseolus vulgaris L. at near-ambient ozone concentrations. New Phytol 122:63–70. https://doi.org/10.1111/j.1469-8137.1992.tb00053.x

Sanhueza E, Cuenca G, Gomez MJ et al (1988) Characterization of the Venezuelan environment and its potential for acidification. In: Henning R, Herrera R (eds) Acidification in tropical countries. Wiley, New York, pp 197–255

Seinfeld JH, Pandis SN (2016) Atmospheric chemistry and physics: from air pollution to climate change. Wiley, Hoboken

Sharma RK, Bhattarai BK, Sapkota BK et al (2012) Black carbon aerosols variation in Kathmandu valley, Nepal. Atmos Environ 63:282–288. https://doi.org/10.1016/j.atmosenv.2012.09.023

Shaw PJA, Holland MR, Darrall NM, Mcleod AR (1993) The occurrence of SO2-related foliar symptoms on Scots pine (Pinus sylvestris L.) in an open-air forest fumigation experiment. New Phytol 123:143–152. https://doi.org/10.1111/j.1469-8137.1993.tb04540.x

Shores CA, Klapmeyer ME, Quadros ME, Marr LC (2013) Sources and transport of black carbon at the California-Mexico border. Atmos Environ 70:490–499. https://doi.org/10.1016/j.atmosenv.2012.04.031

Shukla V, Upreti DK (2007) Heavy metal accumulation in Phaeophyscia hispidula en route to Badrinath, Uttaranchal, India. Environ Monit Assess 131:365–369. https://doi.org/10.1007/s10661-006-9481-5

Shukla V, Upreti DK (2011) Changing lichen diversity in and around urban settlements of Garhwal Himalayas due to increasing anthropogenic activities. Environ Monit Assess 174:439–444. https://doi.org/10.1007/s10661-010-1468-6

Sicard P, Augustaitis A, Belyazid S et al (2016) Global topics and novel approaches in the study of air pollution, climate change and forest ecosystems. Environ Pollut 213:977–987. https://doi.org/10.1016/j.envpol.2016.01.075

Silva AA, Tomaz LM (2013) Surface ozone concentrations and local cloud cover at an urban, tropical site in the southern hemisphere. J Atmos Sol Terr Phys 105–106:54–60. https://doi.org/10.1016/j.jastp.2013.08.002

Singh RK, Agrawal M (2005) Atmospheric depositions around a heavily industrialized area in a seasonally dry tropical environment of India. Environ Pollut 138:142–152. https://doi.org/10.1016/j.envpol.2005.02.009

Singh J, Agrawal M, Narayan D (1994) Effect of power plant emissions on plant community structure. Ecotoxicology 3:110–122. https://doi.org/10.1007/BF00143409

Singh N, Yunus M, Srivastava K et al (1995) Monitoring of auto exhaust pollution by roadside plants. Environ Monit Assess 34:13–25. https://doi.org/10.1007/BF00546243

Singh RK, Dutta RK, Agrawal M (2004) Litter decomposition and nutrient release in relation to atmospheric deposition of S and N in a dry tropical region. Pedobiologia (Jena) 48:305–311. https://doi.org/10.1016/j.pedobi.2004.03.003

Singh R, Singh MP, Singh AP (2014) Ozone forming potential of tropical plant species of the Vidarbha region of Maharashtra state of India. Urban For Urban Green 13:814–820. https://doi.org/10.1016/j.ufug.2014.05.007

Sukitpaneenit M, Kim Oanh NT (2014) Satellite monitoring for carbon monoxide and particulate matter during forest fire episodes in Northern Thailand. Environ Monit Assess 186:2495–2504. https://doi.org/10.1007/s10661-013-3556-x

Sumit K, Devara PCS, Manoj MG (2012) Multi-site characterization of tropical aerosols: implications for regional radiative forcing. Atmos Res 106:71–85. https://doi.org/10.1016/j.atmosres.2011.11.009

Swamy YV, Venkanna R, Nikhil GN et al (2012) Impact of nitrogen oxides, volatile organic compounds and black carbon on atmospheric ozone levels at a semi Arid Urban Site in Hyderabad. Aerosol Air Qual Res 12:662–671. https://doi.org/10.4209/aaqr.2012.01.0019

Tian D, Niu S (2015) A global analysis of soil acidification caused by nitrogen addition. Environ Res Lett 10:24019. https://doi.org/10.1088/1748-9326/10/2/024019

Tiwari S, Srivastava AK, Bisht DS et al (2009) Black carbon and chemical characteristics of PM10 and PM2.5 at an urban site of North India. J Atmos Chem 62:193–209. https://doi.org/10.1007/s10874-010-9148-z

Trivedi S, Agrawal M (2003) Air quality status in Varanasi City. J Environ Stud Pol 6(2):75–84

Vadrevu KP, Ellicott E, Giglio L et al (2012) Vegetation fires in the himalayan region – Aerosol load, black carbon emissions and smoke plume heights. Atmos Environ 47:241–251. https://doi.org/10.1016/j.atmosenv.2011.11.009

Valdés Manzanilla A, de la Cruz Uc X (2015) Spatial distribution of SO2 concentration and population exposure in the tropical city of Villahermosa, Mexico. Air Qual Atmos Health 8:321–329. https://doi.org/10.1007/s11869-014-0273-8

Vasconcellos PC, Souza DZ, Magalhães D, Da Rocha GO (2011) Seasonal variation of n-alkanes and polycyclic aromatic hydrocarbon concentrations in PM10 samples collected at urban sites of São Paulo State, Brazil. Water Air Soil Pollut 222:325–336. https://doi.org/10.1007/s11270-011-0827-4

Verma A, Singh SN (2006) Biochemical and ultrastructural changes in plant foliage exposed to auto-pollution. Environ Monit Assess 120:585–602. https://doi.org/10.1007/s10661-005-9105-5

WHO (2016). Ambient air pollution database. http://www.who.int/entity/phe/health_topics/outdoorair/databases/who-aap-database-may2016.xlsx?ua=1. Accessed 26 June 2017

Wickramasinghe AP, Karunaratne DGGP, Sivakanesan R (2011) PM10-bound polycyclic aromatic hydrocarbons: concentrations, source characterization and estimating their risk in urban, suburban and rural areas in Kandy, Sri Lanka. Atmos Environ 45:2642–2650. https://doi.org/10.1016/j.atmosenv.2011.02.067

Xiankai L, Jiangming M, Shaofeng D (2008) Effects of nitrogen deposition on forest biodiversity. Acta Ecol Sin 28:5532–5548. https://doi.org/10.1016/S1872-2032(09)60012-3

Yerramsetti VS, Gauravarapu Navlur N, Rapolu V et al (2013) Role of nitrogen oxides, black carbon, and meteorological parameters on the variation of surface ozone levels at a tropical urban site – Hyderabad, India. Clean Soil Air Water 41:215–225. https://doi.org/10.1002/clen.201100635

Zvereva EL, Toivonen E, Kozlov MV (2008) Changes in species richness of vascular plants under the impact of air pollution: a global perspective. Glob Ecol Biogeogr 17:305–319. https://doi.org/10.1111/j.1466-8238.2007.00366.x

Shorea robusta Forest Resources of Mainpat/Phendeling Tibetan Refugee Camp, Chhattisgarh, India

8

Amanda E. Knauf, Peter Z. Fulé, and Emily E. Fulé

Abstract

Many tropical forests of India are highly degraded and sometimes lack adequate regeneration. In the tropical dry deciduous forest, *Shorea robusta* is a highly valued timber species. The Mainpat/Phendeling Tibetan settlement was established in 1962 in Chhattisgarh, east-central India. We assessed the forest surrounding the settlement and interviewed local residents to assess the forest conditions and the uses of forest products. Six sites were chosen to measure forest structure over a gradient of human impact. The forest area surrounding Mainpat showed evidence of degradation and overuse. We found that those sites classified as higher impact had more dead trees ha^{-1} and more dead tree basal area than low-impact sites. The dominant tree species, *S. robusta*, had no regeneration. Survey responses showed that the forest was primarily used for the collection of fuelwood and construction materials. Although the forest is under government protection, there are social needs and ecological issues associated with regeneration that merit management attention. We suggest several considerations for management including environmental education, provision of alternative forest resources, testing of artificial *S. robusta* regeneration, alternative energy sources, and participation in a payment for ecosystem services plan. No matter the approach, the restoration and conservation of the forest is critical for the regeneration of *S. robusta* and the future sustainability of the forest ecosystem.

Keywords

Sal tree · Regeneration · Sustainability · Forest structure

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8.1 Introduction

Approximately 86% of India's forest land is classified as tropical forest (Sagar and Singh 2005). Large areas of these forests throughout India are degraded due to habitat destruction, pollution, overexploitation, and species introduction (UNEP 2001). Tropical forests typically have high regeneration capacities, but 72% of forests in India have no regeneration (Rai and Saxena 1997). With proper protection and management, tropical forests in India have shown recovery, and the return of regeneration is feasible (Behera and Misra 2006). The dry deciduous forest present in many of India's eastern states is dominated by *Shorea robusta* C.F. Gaertn. (Dipterocarpaceae), commonly known as "Sal." *Shorea robusta* is a widespread species in India and Nepal, where it forms forests of high biomass and carbon storage (Thapa-Magar and Shrestha 2015) that are often heavily exploited by people (Chitale et al. 2014). It is a highly valued timber species also heavily used for subsistence needs such as house construction, fuelwood, and leaves for livestock forage, cups and plates, etc. The non-timber forest products (NTFP) obtained from these forests are often crucial to residents´ livelihoods because they provide additional income (Mahapatra and Shackleton 2012). Consequently, the conservation and restoration of these forest ecosystems is critical for sustaining ecosystem function and for the local communities that rely on them.

Non-timber forest products can significantly alleviate poverty for households in forest communities (Fisher 2004; Shackleton and Shackleton 2004; Belcher et al. 2005). The degradation of local forest in poor forest communities is a challenge for resource planners because degradation is closely linked with poverty (Clark and Sunderland 2004). However, harvesting NTFP is thought to create incentives to conserve the forest (Arnold 2002; Cocks and Wiersum 2003). Management strategies that provide local communities with the ability to harvest NTFP could be advantageous in restoring degraded forest areas, recovering regeneration, and providing monetary relief to local residents (Pagdee et al. 2008).

In 1962, the Indian government established the Mainpat (Phendeling) settlement to accommodate Tibetan refugees. The settlement is surrounded by tropical forest dominated by *S. robusta* which shows visible signs of heavy use. Concern over the sustainability of the forest prompted an assessment with three goals: (1) determine the current state of the forest, (2) determine the primary use of the forest by settlement residents, and (3) suggest possible management changes that could be made to enhance the sustainability of the forest.

8.2 Methods

8.2.1 Study Area

This study took place in the forest in and surrounding the Mainpat (Phendeling) Tibetan settlement (22° 49´ N, 83° 15′ E). The settlement is located in Raipur, Surguja District, Chhattisgarh state, in east-central India and is made up of seven camps with a total population of approximately 1825 people. The land area that surrounds the camps is agricultural and forest land, which the residents of Mainpat and surrounding Indian communities use to support their livelihood.

Six study sites were located in the forest surrounding Mainpat's seven camps. Mainpat sits atop a plateau that ranges in elevation from 990 to 1152 m above sea level (Bhagat et al. 2014). Our six sites ranged in elevation from 1020 to 1141 m. The forest area surrounding Mainpat is classified as a tropical dry forest and is dominated by *Shorea robusta* (Sal). Mainpat experiences a typical monsoon climate with three distinctive seasons: summer from March to June, monsoon season from July to October, and winter from November to February. The average temperatures are 11–15 °C in the winter and 22–25 °C in the summer (Bhagat et al. 2014).

8.2.2 Forest Structure

We established six sites representing natural forests dominated by *S. Robusta*, two sites each in the western, central, and eastern Mainpat region. Each site represented either high or low human impact (i.e., tree cutting). Level of impact was determined based on observations and density; specifically if the difference between live and dead trees ha^{-1} was less than 100, impact was considered high. There were two low-impact sites and four high-impact sites. At each site, we systemically located four plots 100 m apart to measure forest overstory and regeneration. Each plot was a circular, fixed-area plot 15 m in radius (707 m^2). At each plot we measured diameter at breast height (DBH, cm) and total height (m) of each tree that exceeded breast height (1.3 m). Cut stumps of trees were recorded as well. Regeneration, defined as trees shorter than breast height, was tallied on each plot. In the lab, we calculated density (trees ha^{-1}) and basal area (m^2 ha^{-1}) of living and cut trees, as well as the distribution of tree heights. Additionally, we calculated regeneration density. Aboveground forest biomass was calculated by applying the allometric equation for *S. robusta* from Rana et al. (1989).

One-way ANOVAs were used to determine if differences in density, basal area, and regeneration were different between high- and low-impact sites. All analyses were done in IBM SPSS Statistics 24. The alpha level was 0.05.

8.2.3 Community Interviews

Residents of different settlements within Mainpat were interviewed to understand how they use forest resources, what access they have to forest resources, and their thoughts on forest health. Bilingual local residents translated the interviews, allowing us to interview non-English-speaking residents.

8.3 Results

8.3.1 Forest Structure and Biomass

The forest structure surrounding Mainpat consisted of living *Shorea robusta* trees in a large range of size classes, including many large trees (≥30 cm DBH) (Fig. 8.1). Trees in the smaller class sizes (≤30 cm DBH) were the most heavily cut, recorded as dead, and observed as cut stumps (Fig. 8.1). There were no standing dead trees. Tree densities among all sites ranged from 95 to 266 trees ha^{-1}, with an overall average of 149 trees ha^{-1} for all six sites (Table 8.1). The distribution of live trees in four of the six sites showed a mostly even-aged distribution, accounting for areas of both high and low impact.

The basal area between the sites ranged from 7.6 to 19.3 m^2 ha^{-1}, and the mean basal area of all the sites was 15.5 m^2 ha^{-1} (Table 8.1). Mean live biomass was 186 Mg ha^{-1} and the mean dead biomass was 60 Mg ha^{-1}. Live biomass ranged from 97 to 235 Mg ha^{-1}. Dead biomass that had been removed from the sites, as calculated based on the cut stumps, ranged from 10 to 107 Mg ha^{-1}, approximately equivalent on average to one-third of live biomass.

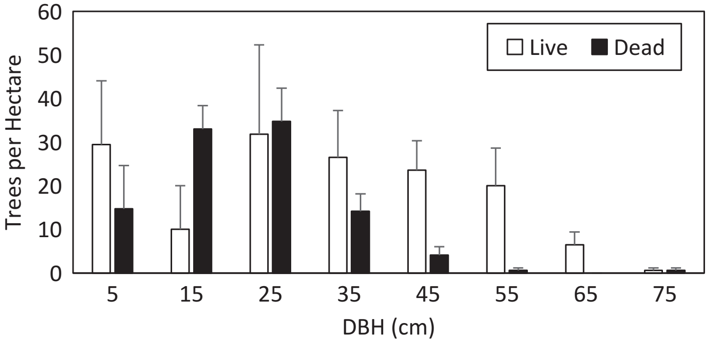

Fig. 8.1 Average forest density for live and dead trees per size class. Error bars are standard errors of the mean

Table 8.1 Basal area, tree density, and live and dead biomass for each site

Site/impact level	Basal area (m^2 ha^{-1})	Tree density (trees ha^{-1})	Live biomass (Mg ha^{-1})	Dead biomass (Mg ha^{-1})
1/high	12.2	191	151	59
2/high	19.3	113	223	107
3/high	19.3	99	235	85
4/low	15.5	266	194	10
5/low	7.6	95	97	25
6/high	18.8	127	215	76
Mean	**15.5**	**149**	**186**	**60**

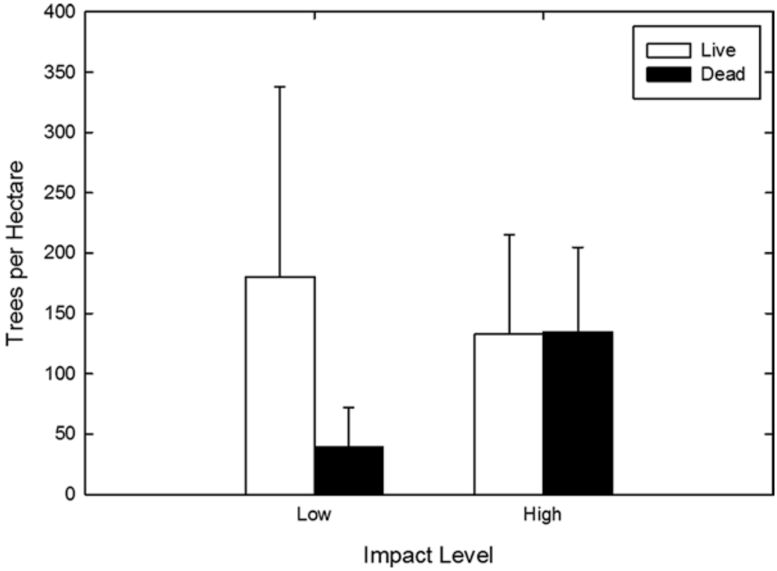

Fig. 8.2 Average live and dead trees per hectare for low and high impact levels. No significant difference in TPH for live trees; however there were significantly less TPH in low-impact areas compared to high-impact areas for dead trees

No significant differences were found between the high- and low-impact sites for live tree density ($P = 0.41, f = 0.93$), but low-impact sites had significantly less dead tree density than high-impact sites ($P < 0.01, f = 6.24$) (Fig. 8.2). Similarly, no differences were found among the high- and low-impact sites for basal area of live trees, but low-impact sites had significantly less basal area for dead trees than high-impact sites ($P < 0.01$, $f = 21.20$) (Fig. 8.3). A few native woody species were encountered on regeneration plots. Regeneration did not differ significantly ($P = 0.09, f = 4.69$), and no regeneration of *S. robusta* was observed.

8.3.2 Community Forest Resource Use

Nine Mainpat residents were interviewed, representing settlements 1, 4, 5, and 6. The residents reported various uses of forest products. All residents stated that the primary use of forest wood throughout the settlement was for construction material and firewood for cooking. Three stated that they continue to use forest wood as their main energy source. The other residents used gas, either LPG or biogas, as their primary energy source. Those residents whose main source of energy was forest wood stated that they collected wood approximately two to three times a month, while others bought it from local Indians who sold it in the settlement.

When asked about the access, use, and health of the forest which surrounds the Mainpat settlements, the residents noted that the government owned the forest and

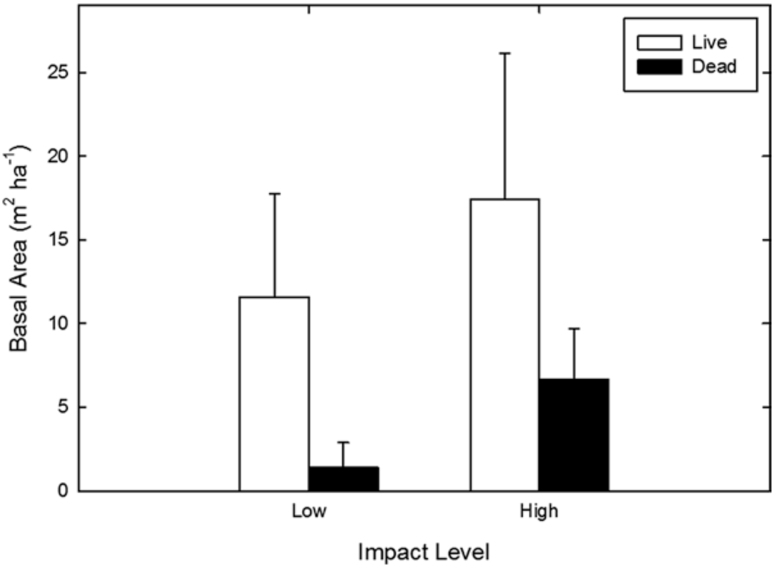

Fig. 8.3 Average live and dead basal area for low and high impact levels. No significant difference in TPH for live trees, but there was significantly less basal area of dead trees in low- vs. high-impact areas for dead trees

no trees could be cut legally. They also reported that there were consequences for being caught cutting down trees within the forest (e.g., fines). In regard to the residents´ judgment of forest health, three felt that there was no damage, and the other six felt that there were multiple causes for damage including heavy deforestation, insects, small fires killing young trees, and lack of tree regeneration.

Apart from the interviews, we also noted numerous women in the forest collecting and carrying *S. robusta* leaves to feed to livestock and fuelwood. The forest floor was generally cleared of all dead wood. Tree canopies close to communities were notably deformed due to pruning of high branches for wood and leaves (Fig. 8.4). Cattle grazing was widespread.

8.4 Discussion

The current state of the *S. robusta* forest surrounding the Mainpat settlement had some characteristics associated with healthy mature forests and others associated with forest degradation. On the positive side, the forest was comprised of many relatively large trees representing an important amount of biomass, averaging 55% higher than the biomass of managed *S. robusta* forests in the mid-hill region of Nepal (Thapa-Magar and Shrestha 2015). However, Singh et al. (1994) reported much higher forest density in the Indian central Himalaya at 300 m elevation where two studied *S. robusta* stands had 443 and 726 trees ha^{-1} and basal areas of 56 and

Fig. 8.4 Canopies of *Shorea robusta* trees close to a Mainpat settlement camp have been affected by pruning of high branches for fodder and fuelwood (upper photo). Forest in a relatively low-impact site is dominated by large trees, but regeneration and dead wood are absent (lower photo)

41 m^2 ha^{-1}. Over 23 stands on an elevational gradient, *S. robusta* ranged from 610 to 1670 trees ha^{-1} and 18 to 92 m^2 ha^{-1} (Singh et al. 1994). In comparison, the *S. robusta* forest surrounding Mainpat had an average of 149 trees ha^{-1} and basal area of 15.5 m^2 ha^{-1}, lower than the minimum values reported by Singh et al. (1994). A study by Bhagat et al. (2014) in a broader region around our study area found that half of the 14 study sites were monospecific, composed exclusively of *S. robusta*, but they did not report absolute values of density or basal area.

Forest use and degradation were evidenced by substantial cutting of trees, approximately equal to a third of the live biomass, and the absence of *S. robusta* regeneration on our plots. Bhagat et al. (2014) did encounter *S. robusta* regeneration on plots in the broader region but did not report quantitative data. Sagar and Singh (2005) reported an average of nearly 4000 seedlings and 150 saplings ha^{-1} of *S. robusta* in Vindhyan dry tropical forests to the northwest of our study region. Chauhan et al. (2008) also reported high seedling density but also high seedling mortality. Regeneration of *S. robusta* forest has been found to be a serious problem in some parts of India due to the combination of high soil water content and poor soil aeration (Gautam and Devoe 2006).

Uses of timber and NTFP noted in survey responses from community members were consistent with our findings in the field. Residents stated their concern about the lack of regeneration and overuse of *S. robusta*. Despite the official policy of forest protection, the forest areas adjacent to the settlement's camps exhibited signs of heavy usage, especially tree cutting. The survey responses supported our field findings, stating that residents obtain fuelwood and building materials from the local forest. Those residents who did not use alternative forms of energy, such as biogas or LPG, either bought wood or collected it themselves multiple times a month.

There are several possible management changes that could be considered to enhance the sustainability of the forest, based on the assessment. First, it is helpful to note that the forest protection policy is helping to conserve the forest. Given the intensive use of wood and leaves, this policy clearly remains important. Nonetheless, the lack of *S. robusta* regeneration in the area is a concern for forest health and resiliency. The following suggestions could be useful:

1. Environmental education can be used to inform the local communities of approaches that may be utilized within the camps to conserve the forest and address the need for forest resources without further degradation of the land. The education should extend to the local Indian people who live in the same area. Approaches based on community forestry principles, where residents are empowered to participate in decision-making, have potential to improve conservation as well as economic benefits from the forest (Mehta and Heinen 2001; Badola et al. 2012).
2. Provide officially sanctioned sources of timber and NFTP. Depending on the circumstances, this could include limited harvest of native trees. Given the *S.*

robusta regeneration failure, however, this species should remain protected. An alternative would be to propagate other tree species, especially sprouting species that could be coppiced. There is an existing nursery facility in the camps that is testing various tree species for fruit, shade, and living fence applications. Expanding this work to address wood and forage resources could be useful.

3. Test artificial regeneration of *S. robusta*. A few limited plantings were carried out in the camps, with low survival. However, considerable research attention to the issue of *S. robusta* regeneration has led to recommendations in the literature for nursery and planting guidelines, such inoculations with mycorrhizae (Tapwal et al. 2015). Tests on propagation and outplanting of *S. robusta* seedlings, along with protection from herbivory, would provide useful information on the possibility of underplanting as a mechanism for maintaining the forest.
4. Support alternatives to wood energy. LPG is an existing alternative but can be limited by cost. The Tibetan communities have developed pilot projects to create biogas from cattle manure collected from paddocks and fields. This technology requires an initial investment to install a fermentation tank but then provides clean energy.
5. A longer-term program of payments for ecosystem services could be beneficial for designing and supporting a sustainable forest conservation program that also provides for timber and NTFP use. The REDD+ (Reducing Emissions from Deforestation and Forest Degradation plus other management activities) program under the United Nations Framework Convention on Climate Change is a complex and challenging endeavor (Bau et al. 2014). It has been tested in a limited way so far in India (Vijge 2015). While it is too early to say whether REDD+ or similar schemes could be helpful in the Mainpat region, they merit further attention as a possibly useful management approach.

The *S. robusta* forest of the Mainpat/Phendeling Tibetan refugee camp region provides important resources. While the forest currently conserves many larger trees, the loss of small trees to cutting and absence of regeneration are key concerns. A thoughtful program of restoration and conservation could contribute to protecting the forest while providing for the needs of local residents.

References

Arnold M (2002) Clarifying the links between forests and poverty reduction. Int For Rev 4:231–233

Badola R, Barthwal S, Hussain SA (2012) Attitudes of local communities towards conservation of mangrove forests: a case study from the east coast of India. Estuar Coast Shelf Sci 96:188–196

Bae JS, Kim Y-S, Fisher L, Moeliono M, DeShazo J (2014) Promises and perils of decentralized forest governance: the case of Indonesia's Forest Management Units in Reducing Emission from Deforestation and Forest Degradation (REDD+). Soc Nat Resour 27(12):1346–1354. https://www.cifor.org/library/5380/

Behera SK, Misra MK (2006) Aboveground tree biomass in a recovering tropical sal (Shorea robusta Gaertn. f.) forest of Eastern Ghats, India. Biomass Bioenergy 30:509–521

Belcher B, Ruiz-Perez M, Achdiawan R (2005) Global patterns and trends in the use and management of commercial NTFPs: implications for livelihoods and conservation. World Dev 33:1435–1452

Bhagat M, Sahu PK, Shah RP, Soni M (2014) Plant species diversity and phytosociological analysis in forest of Jaljali Mainpat, Ambikapur, (CG). Int J Pharm Life Sci 5(6):3627–3630

Chauhan DS, Dhanai CS, Singh B, Chauhan S, Todaria NP, Khalid MA (2008) Regeneration and tree diversity in natural and planted forests in a Terai-Bhabhar forest in Katarniaghat wildlife sanctuary, India. Trop Ecol 49(1):53–67

Chitale VS, Behera MD, Matin S, Roy PS, Sinha VK (2014) Characterizing *Shorea robusta* communities in the part of Indian Terailandscape. J For Res 25(1):121–128

Clark LE, Sunderland TCH (eds) (2004) The key non-timber forest products of Central Africa: a state of the knowledge. Technical paper no. 122. USAID. Washington, DC, USA, p 185

Cocks ML, Wiersum KF (2003) The significance of plant diversity to rural households in Eastern Cape province of South Africa. For Trees Livelihoods 13:39–58

Fisher M (2004) Household welfare and forest dependence in Southern Malawi. Environ Dev Econ 9:135–154

Gautam KH, Devoe NN (2006) Ecological and anthropogenic niches of sal (Shorea robusta Gaertn. f.) forest and prospects for multiple-product forest management–a review. Forestry 79:81–101

Mahapatra A, Shackleton C (2012) Exploring the relationships between trade in natural products, cash income and livelihoods in tropical forest regions of Eastern India. Int For Rev 14:62–73

Mehta JN, Heinen JT (2001) Does community-based conservation shape favorable attitudes among locals? An empirical study from Nepal. Environ Manag 28:165–177

Pagdee A, Homchuen S, Sang-arome P, Sasaki Y (2008) Community forest: a local attempt in natural resource management, economic value of ecosystem services, and contribution to local livelihoods. 12th International conference on integrated diffuse pollution management (IWA DIPCON 2008), Khon Kaen University, Thailand, 25–29 August 2008

Rai S, Saxena A (1997) The extent of forest fire, grazing and regeneration status in inventoried forest areas of India. Indian Forester 123:689–702

Rana BS, Singh SP, Singh R (1989) Biomass and net primary productivity in Central Himalayan forests along an altitudinal gradient. For Ecol Manag 27:199–218

Sagar R, Singh J (2005) Structure, diversity, and regeneration of tropical dry deciduous forest of northern India. Biodivers Conserv 14:935–959

Shackleton C, Shackleton S (2004) The importance of non-timber forest products in rural livelihood security and as safety nets: a review of evidence from South Africa. S Afr J Sci 100:658–664

Singh SP, Adhikari BS, Zobel DB (1994) Biomass, productivity, leaf longevity, and forest structure in the central Himalaya. Ecol Monogr 64:401–421

Tapwal A, Kumar R, Borah D (2015) Effect of mycorrhizal inoculations on the growth of *Shorea robusta* seedlings. Nusant Biosci 7(1):1–5

Thapa-Magar KB, Shrestha BB (2015) Carbon stock in community managed hill sal (*Shorea robusta*) forests of central Nepal. J Sustain For 34:483–501

UNEP (2001) India: state of the environment-2001. United Nationals Environmental Programme

Vijge MJ (2015) Competing discourses on REDD+: global debates versus the first Indian REDD+ project. Forest Policy Econ 56:38–47

9

Ecology of Treeline Vegetation in Western Himalaya: Anthropogenic and Climatic Influences

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and G. S. Rawat

Abstract

A landscape level survey of vegetation structure and composition was conducted along the treeline ecotone in Western Himalaya in the states of Uttarakhand and Himachal Pradesh. We used a combination of field surveys and remote sensing data (Landsat MSS and TM images) to quantify the extent and distribution of various classes of forests, forest cover, patterns of vegetation structure and composition and recent changes in forest cover along the treeline ecotone. Total area under the treeline ecotone (3000–4000 m) in these states comes to ca. 11587 km^2, of which Himachal Pradesh has higher representation (6818.4 km^2). In terms of extent, conifers (*Abies* and *Pinus*) occupy the largest cover followed by brown or Kharsu oak (*Quercus semecarpifolia*) and birch (*Betula utilis*). Mean tree density was 553 individuals ha^{-1} which ranged between 440 and 690 individuals ha^{-1}. Average elevation of the treeline was approximately 3615 m above mean sea level, and the maximum altitude of tree growth was 4200 m. Time series analysis of remote sensing data reveals that there has been no altitudinal shift in the upper treeline during the last 40 years, although forest canopy cover has increased or decreased significantly within a protected area and outside, respectively.

Keywords

Anthropogenic pressures · Change analysis · Protected area · Treeline · Western Himalaya

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9.1 Introduction

Natural barriers to the spread of forests on the earth's surface are represented by the extremes of environmental conditions (Ellenberg 1966). Mountain environments, in particular, exhibit prominent biophysical limits to the distribution of various forest formations. One of the most prominent ecological boundaries in the high elevations is the alpine treeline or timberline. Treeline represents the most striking vegetation boundary on the earth's surface wherever mountains rise to sufficient heights. In reality, the transition from the uppermost closed montane forests to the treeless alpine vegetation is commonly not a line but a steep gradient of increasing stand fragmentation and stuntedness, often called the treeline ecotone (Körner 1998a, b; Körner and Paulsen 2004). In other words, the treeline is defined as the altitude above which trees do not attain a height more than 3 m and the timberline ecotone refers to the broad area well below the treeline bounding the closed forest. The forest may end as a closed stand with an abrupt and sharp line with treeless zone above, or it may gradually dissolve with scattered trees with low stunted vegetation or *krummholz* and end with the alpine zone as a broad transition zone. The gradual opening of the stand results from the sharp climatic gradient that prohibits tree growth (Tranquillini 1979; Körner 2012). Besides climate, many other factors such as rocky slopes, slope debris and avalanche chutes also prevent the forests from reaching its upper limit resulting in orographic timberline (Holtmeier 2003).

Worldwide, seasonal mean air temperature between 5 and 7 °C is the most consistent predictor of the altitudinal position of treelines (Körner 1998a, b, 2012). It has been established that tree growth is limited by the isotherm of 10 °C for the warmest months but sometimes other factors also limit their growth. For example, in southwestern Alaska, the isotherm stands ca. 400 km beyond the edge of the forest (Griggs 1937) and stunted and isolated forest stands can be formed at higher elevation in favourable microclimatic conditions. The treeline ecotones are of considerable ecological interest as they serve as indicators of response of plant species to environmental gradients, as loci for species diversity and as transition between two ecosystems reflected by spatial flux of species, matter and energy (Malanson 1997). Thus the alpine treelines are ideally suited to study the impacts of climate change on vegetation and, conversely, recent changes in climate using climate-sensitive species and indicators.

Several authors have dealt with global distribution of high-altitude forests and factors that limit the tree growth above natural treelines (Troll 1973; Arno 1984; Wardle 1974; Tranquillini 1979; Körner 2003; Körner and Paulsen 2004; Holtmeier and Broll 2007). There is a long history of studies on the ecophysiology, positions and causes of treeline formations (Däniker 1923; Tranquillini 1979; Wardle 1993; Miehe and Miehe 1994; Kullman 1998; Ohsawa 1990; Körner 1998a, b). However, most of these studies have been done on the treelines of the Alps, Andes and mountains of New Zealand. Treeline shift, increased growth and increased tree establishment near treelines have been observed in the recent years in many parts of the world (Gehrig-Fasel et al. 2007; Paulsen et al. 2000; Motta and Nola 2001).There is a common thinking that altitudinal treelines are currently advancing towards higher altitudes, and along latitudinal gradient, they are perceived to be moving further north in the northern hemisphere (Arno 1984; Shugart et al. 2001; Parmesan 2006; Holtmeier and Broll 2007). However, this trend has many anomalies due to its dependency upon local and regional conditions such as edaphic factors, orographic influences, herbivory, fire, disease and anthropogenic influences (Cairns and Moen 2004; Weiser and Tuasz 2007). Grace et al. (2002) reviewed the impacts of climate change on the treeline including patterns of the growth of the treeline species. Impacts of climatic fluctuation on treeline vegetation have drawn attention of several workers across the globe especially on the increased growth, regeneration and shifting of treelines to higher elevation (Kullman 2001; Gehrig-Fasel et al. 2007; Paulsen et al. 2000; Motta and Nola 2001; Körner 2003).

Although several authors have worked on the ecology of high-altitude forests along the elevation gradients touching the treeline zone (Singh and Singh 1992; Garkoti and Singh 1994; Adhikari et al. 1995; Rawal and Dhar 1997), in-depth studies on the ecology of Himalayan treelines are very few (Schickhoff 2005). Recently, Bharti et al. (2012), Rai et al. (2012) and Rai (2012) studied ecology, extent and changes along the timberline ecotone in the Western Himalaya with rigorous field data and observations. These studies indicated either increase or decrease of vegetation cover along timberline zone (3000–4000 m), but none of them report a clear shift in the upper limit of forests. Based on remote sensing tools, Singh et al. (2012) and Panigrahy et al. (2010) attempted to detect changes in terms of treeline shift in the Western Himalaya, but due to lack of sufficient ground observation and methodological error, these studies were not widely accepted (Bharti et al. 2011). Regeneration of tree species towards higher alpine slopes adjacent to treeline has been reported from several localities in the Western Himalaya (Dubey et al. 2003; Rai et al. 2012) mainly in the protected areas. Shrestha et al. (2007) also reported improved regeneration of birch at the treeline in a trans-Himalayan valley of Nepal. Based on dendrochronological studies, Yadava et al. (2016) and Gaire et al. (2017) have reported elevational movements of the subalpine tree species in the Western Himalaya mainly under microclimatic conditions favourable for the growth. However, these trends are not consistent across the region (Chhetri and Cairns 2016).

The foregoing review indicates that very few studies have been conducted on the extent, community structure, composition and recent changes in the forest cover at a landscape level in the Himalayan region. Identification of sensitive treeline communities as well as sites that can ideally serve as long-term monitoring to observe changes in the region is limited. This paper attempts to address some of these issues. The study was conducted with the following objectives: (i) to study the broad patterns and the extent of forest vegetation along the treeline ecotone in Western Himalaya, (ii) to assess the patterns of vegetation structure across climatic and anthropogenic pressure gradients and (iii) to detect recent changes in vegetation cover along the treeline within and outside a few protected area.

9.2 Material and Methods

9.2.1 Survey of Treeline Ecotone

Treeline of the Western Himalaya was surveyed intensively in two of the Indian states, viz. Uttarakhand (UA) and Himachal Pradesh (HP). Representative sites in UA were various parts of Kedarnath Wildlife Sanctuary (WS), Nanda Devi Biosphere Reserve, Valley of Flowers National Park (NP), Gangotri NP and Askot WS and the vicinity outside these protected areas. In HP, survey localities include Great Himalayan NP, Churdhar WS and Kothi Forest near Rohtang Pass. Ground-truth points for mapping were collected from >500 localities based on various field visits. Vegetation sampling was also done following Misra (1968) and Kershaw (1973). We used 100 m^2 area (10 × 10 m square plots) for sampling the tree layer which has been found adequate for sampling forest vegetation in the Himalayan region (Singh and Singh 1992; Adhikari et al. 1995; Rai 2012). At each site, ten random plots were laid out covering the treeline ecotone at a distance of 10 m between subsequent plots. Isolated trees much beyond the ecotone were not considered. Primary data was collected following standard phytosociological approach (Muller-Dombois and Ellenberg (1974). Saplings (<20 cm circumference at breast height, cbh) and seedlings of tree species were recorded within10 × 10 m quadrats to observe the regeneration status. The structural aspect of vegetation such as density, frequency, abundance and dominance of constituent species was determined following Misra (1968). The Importance Value Index (IVI) was computed for all the tree species by adding the relative values of frequency, density and dominance (basal area) following Curtis and McIntosh (1950). Each site was categorized into high, moderate, low anthropogenic impact on the basis of the presence (number) of seasonal livestock camps and temporary shops which affect the structure of treeline vegetation (Singh et al. 2010).

9.2.2 Satellite and Ancillary Data

Landsat data was used for the study due to its high temporal resolution and continuous availability since 1972. Acquisition dates of satellite images have a strong impact on the total area available for change detection due to the fact that a large sun angle causes the increase in areas with deep shadow that cannot be included in change detection analysis. Images of peak growing season (July to August) were not considered due to poor discrimination of alpine meadow from deciduous forest, while the use of images before the month of April is not justifiable due to the fact that most of the deciduous species attain full maturity only after this month. Visual interpretation indicates that May to June is the best period to acquire shadow-free images because of low sun angle. Images acquired during this period also reduce error due to phenological differences as leaves flushing in *Betula* and other deciduous tree species starts only after March, and low reflectance of alpine meadow does not create conflict with other vegetation types. Therefore, in the absence of

Table 9.1 Landsat dataset used for vegetation classification and change analysis

Acquisition period	Sensor	Path/Row	
For vegetation classification			
October, November 2016	OLI and TIRS	145/039, 146/038, 146/039, 144/039, 147/037, 147/038, 148/037	
For change analysis			
Acquisition period	Sensor	Path/row (Gangotri)	Path/row (Chhipla Kedar)
31 October and 01 November 1976	MSS	156/039	155/039
02, 06 November 1996	TM	146/039	145/039
13, 22 November 2016	OLI and TIRS	146/039	145/039

cloud-free images from the post-monsoon period, images from October and early November were used in the present study (Table 9.1).

Satellite imagery of Landsat MSS, TM, OLI and TIRS were obtained from USGS Earth Explorer imagery archive for the years of 1976, 1996 and 2016, respectively. The images were further geometrically corrected using ortho-rectified Landsat series imagery as reference and were geo-referenced to the UTM coordinate system (Zone 44) with the datum of WGS84. Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER) Global Digital Elevation Model Version 2 (GDEM Ver2) was used as elevation data. This 30 m spatial resolution elevation data served as the digital elevation model (DEM) in this study. All images were clipped within the elevational zone of 3000 to 4000 m elevational zone for Himachal Pradesh and 3000 to 4200 m for Uttarakhand area using AOI tool in ERDAS IMAGINE 9.3 Software.

9.2.3 Image Classification

To attain a good discrimination among different vegetation classes, Landsat data were corrected for radiometric distortions by converting raw digital number (DN) to satellite reflectance. Since high mountain ranges of Himalaya cause very high spatial variation in plant species assemblages through the impact of elevation, slope and aspect, an ASTER global digital elevation model (AGDEM) was used along with multispectral bands. To avoid the impact of diffuse sunlight on steep slopes, normalized difference vegetation index (NDVI) calculated from reflectance band along with tasseled cap-derived brightness and wetness index were used as additional layers with four multispectral bands. Finally, eight bands composed of four multispectral bands, tasseled cap brightness and wetness index, NDVI and resampled AGDEM at float scale were stacked together for vegetation classification.

An unsupervised classification approach through isodata clustering was adopted in the present study. However, derivation of snow and glaciers and the rocks/barren class is based on supervised classification for which training samples were collected from the image itself. First non-vegetated surfaces were removed from the eight-band stacked image with the help of a mask generated from NDVI. Threshold value for vegetation was selected based on visual inspection of pseudo-colour NDVI image overlaid over colour composite of the same area using the swipe function. Data collected during field work were used to sort out the clusters into eight land cover classes. Image classification and accuracy assessment of thematic maps were done based on ground-truth points collected using Global Positioning System (GPS) covering various vegetation types. Only pure vegetation patches larger than a hectare were considered for validation. Since it was not possible to collect minimum number of GPS point locations for each class due to highly undulating terrain, some points were generated from image itself based on field knowledge.

9.2.4 Assessment of Change in Forest Cover

Image differencing method was used for change analysis as it has less computational requirements and has better performance as compared to visual interpretation. The difference images were derived by simple subtraction of the previous image from the recent image. Before subtracting the image, snow and cloud pixels were masked, and minimum equal area available for both the image was used to derive the difference image. Masks were generated from tasseled cap wetness index by selecting a threshold based on visual interpretation. The rescaled eight-bit image was used to create a difference image. The difference image contains negative, zero and positive values indicating decreased, no-change and increased vegetation, respectively. Conversion of difference image into binary image (change no-change) is mostly based on threshold value decided from the standard deviation of the change image. Generally threshold value is decided based on accuracy assessment of the different binary images derived from different multiples of standard deviation. Since this study focused on changes only above timberline, a higher multiple (2.5) of standard deviation was selected to avoid changes due to low difference that may be the result of error at various processing stages.

Changes in the timberline ecotone in part of Gangotri National Park (protected area) and upper part of Gori valley (Chhipla Kedar) (high anthropogenic pressure) were studied over a period of 40 years (1976–2016) to understand the treeline vegetation pattern in the sites. Vegetation index-based change detection method is used, which is comparatively easier yet the most effective method. Normalized difference vegetation index (NDVI) derived from Landsat TM and MSS data shows positive correlation with canopy cover, and its performance was found better than that of single band or combined use of all six bands (Larsson 1993; Xu et al. 2003). NDVI has widely been used for detecting changes in vegetation parameters and in tundra landscape (Myneni et al. 1997; Masek 2001; Fraser et al. 2011) and also to study the advancement of treeline vegetation (Jeffrey 2001; Zhang et al. 2009).

9.3 Results and Discussion

9.3.1 Broad Patterns and Land Cover Along Treeline Ecotone

Based on extensive survey, we recognize three categories of treeline in the Western Himalaya, viz. (i) anthropogenic treeline, (ii) natural treeline and (iii) orographic treeline (Plate 9.1). The anthropogenic treelines are generally located below 3300 m closer to age-old temporary settlements reflecting long history of human use especially by migratory pastoral communities, who camp around timberline for the collection of fuelwood and use alpine meadows to graze their livestock. The natural treelines are found mostly in least-disturbed mountain slopes especially in the inner valleys of Gangotri National Park and Nanda Devi National Parks and inner regions in Byans valley (Eastern UA). The orographic treelines are characterized by abrupt termination of subalpine forests below the potential elevations due to steeper rocky slopes.

The dominant tree species at the treeline of Western Himalaya are kharsu or brown oak (*Quercus semecarpifolia*), *Abies spectabilis* and *Betula utilis* found in various associations with other species also, while *Pinus wallichiana* and *Juniperus polycarpos* also form pure forest stands in the inner dry valleys. *Rhododendron campanulatum* forms *krummholz* vegetation and dominates shrub layer with one or more species of *Salix*, *Rubus*, *Sorbus*, *Rosa*, *Viburnum*, *Lonicera* and *Spiraea*. All natural treelines ranged between 3500 and 4000 m as discontinuous patches, while in some areas it is extended up to 4200 m. It is higher in the north-facing slopes as compared to south-facing dry slopes where most forests terminated abruptly. *Abies spectabilis* forms pure community or sometimes mixed with birch in the northwest- to north-facing slopes. The treelines on south-west facing slopes are formed mainly by *Q. semecarpifolia* in association with *Rhododendron arboreum*. *Betula utilis* is a typical species forming treeline in the moist slopes in association with various species of *Salix* and *Rhododendron*. *Abies spectabilis* forms pure community or sometimes mixed with birch in the northwest- to north-facing slopes. Timberlines of southwest-facing slopes are formed mainly by *Q. semecarpifolia* where *R. arboreum* also associated with it sometimes forms pure patches in steep rocky slopes near treeline where oak is unable to grow.

The anthropogenic timberlines are generally devoid of regenerating individuals (seedlings, saplings) and represented by large trees only. In many places due to the cumulative impacts of livestock grazing, fuelwood and fodder extraction and lack of regeneration, the treeline is depressed to lower elevation, and large forest gaps are being formed (Thakur et al. 2011). The depressed timberlines are mostly on the gentle warm slopes favourable for grazing of livestock and camping. These timberlines are highly prone to disturbances as lack of regeneration may cause large forest gaps and depress timberline in lower elevations with the mortality of trees due to natural or anthropogenic causes. Similar observations have been made in other mountains of the world (Holtmeier 2003).

Plate 9.1 Types of treelines in the Western Himalaya
Anthropogenic treeline: Left: *Q. semecarpifolia* timberline at Bedini-Aali depressed due to high grazing pressure and camping sites in vicinity. Right: *A. spectabilis* timberline in Kedarnath WS, also under heavy anthropogenic pressure
Natural treelines: Left: *Abies spectabilis* timberline (3600 m) in the Great Himalayan NP showing a transition with the *krummholz* formation towards higher slopes and mixing of young individuals. Right: *Betula utilis* treeline near Bhojbasa, Gangotri NP (4000 m), showing gradual change in tree physiognomy and stature
Orographic treeline: Left: *Quercus semecarpifolia* timberline near Sokharak, Kedarnath WS (3350 m), wherein upper slopes are replaced by *R. arboreum* which has ability to colonize steep rocky habitats. Right: *Pinus wallichiana* treeline reaching up to 3800 m at Gunji, Byans valley (UA), where forest terminates due to rocky slope

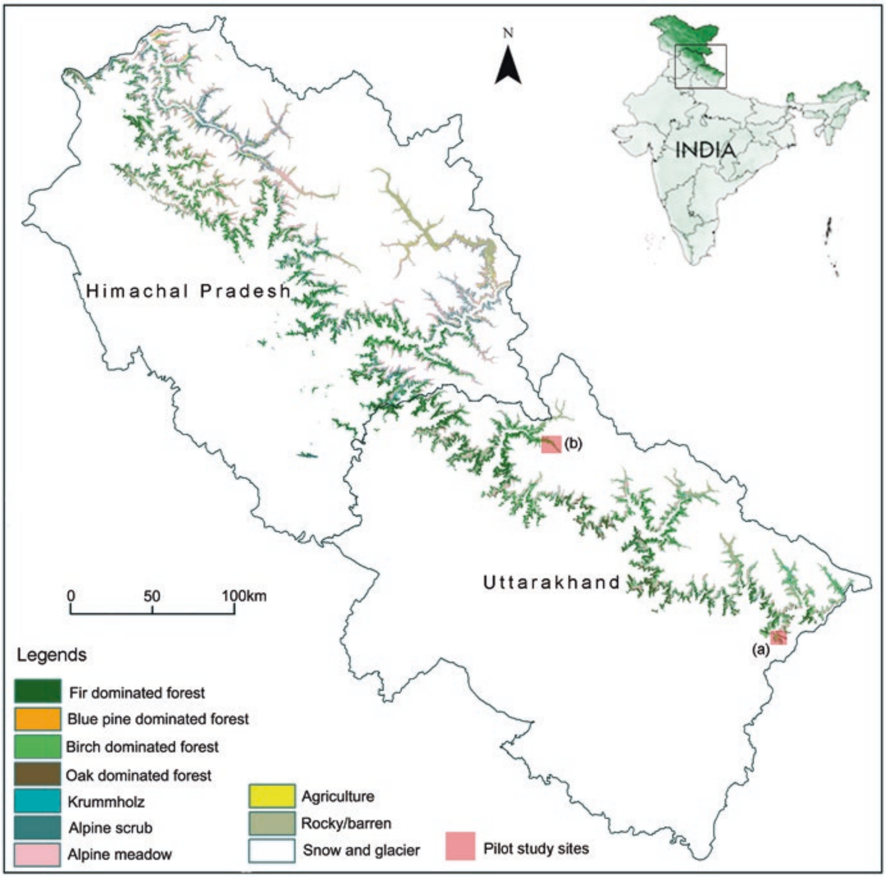

Fig. 9.1 Vegetation map of treeline ecotone of Western Himalaya with two pilot sites (**a**) and (**b**)

Total geographical area between (3000–4000 m) the two states comes to about 11,587 km^2, of which Himachal Pradesh has the higher representation (6818.37 km^2). However, Himachal Pradesh has a considerable area under cultivation and orchards and forest cover is only about 2519 km^2. Uttarakhand has a larger proportion of timberline vegetation under forest (2743.54 km^2).

Thematic map based on vegetation classification reveals that conifers (*Abies* and *Pinus wallichiana*) form the prominent forest cover around treeline (Fig. 9.1). Major cover classes include fir (1719.3 km^2), brown oak (1461.7 km^2), birch (34.6 km^2) and *Krummholz* (316.9 km^2) (Fig. 9.2). The extent of conifers increases towards northwest. Himachal Pradesh has colder arid region in the inner valleys of Lahaul-Spiti where *J. polycarpos* also forms isolated stands generally above 4000 m areas. These formations are also predominant in parts of Gangotri NP and interior areas of Nanda Devi Biosphere Reserve, Uttarakhand. Fir-bearing forests were more prominent in HP (1173.4 km^2), while brown oak (983.3 km^2) has higher proportion in Uttarakhand. Alpine scrub class is mostly represented by *Rhododendron anthopogon* and *Juniperus indica*.

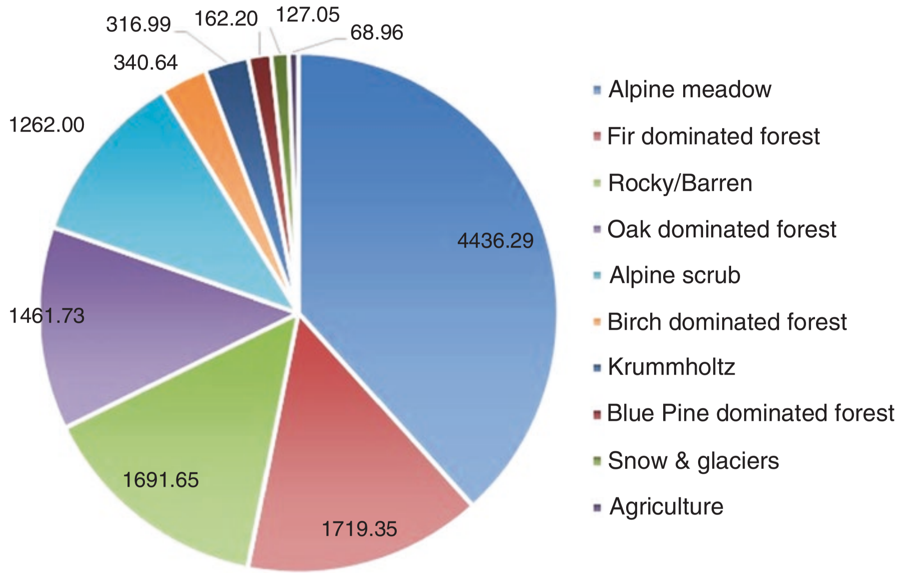

Fig. 9.2 Area statistics (km^2) of different vegetation and land cover classes of treeline ecotone in Western Himalaya

Fragmentation analysis of the treeline vegetation reveals that overall average height of timberline is approximately 3615 m and the maximum height is 4200 m in Western Himalaya. However, the actual maximum height recorded by authors during the survey is 4175 m at Kutti Yangti (Byans valley) in Uttarakhand. In Himachal Pradesh, upper treeline was recorded up to 3800–4000 m as observed in inner Sutlej valley where *P. wallichiana* and *Q. semecarpifolia* form the treeline. This was also reported by Schweinfurth 1957).

9.3.2 Community Characteristics

Five species of trees dominated the woody vegetation along treeline ecotone, viz. *Abies spectabilis*, *Betula utilis*, *Pinus wallichiana*, *Rhododendron arboreum* and *Quercus semecarpifolia*. Degree of slope, aspect and altitudinal preference by these species are shown in Table 9.2. *B. utilis* and *A. spectabilis* were restricted to the moist northeast- to northwest-facing slopes. One or more species of *Rhododendron*, *Viburnum*, *Sorbus* and *Salix* formed stunted forests or *krummholz* that is replaced by open alpine meadows at higher altitudes. In dry southwest- to southeast-facing slopes *Q. semecarpifolia* and *R. arboretum* forests form timberline. However, *R. arboreum* forms treeline only when *Q. semecarpifolia* is unable to reach up to potential treeline elevations due to steep rocky slopes which favours growth of *Rhododendron*. In the inner dry valleys along transition zone of greater and trans-Himalaya, *P. wallichiana* and *Juniperus polycarpos* also reach the higher dry slopes and form orographic treelines.

Betula utilis formed extensive stands on northern shady slopes, ravines and glacial-fluvial deposits. A total of six species were found to be associated with *B. utilis* at treeline, viz. *A. spectabilis*, *Sorbus foliolosa*, *Prunus cornuta* and

Table 9.2 Dominant tree species along the timberline ecotone in Western Himalaya with the topographic features

Species	Slope (°)	Aspect	Timberline (m., asl.)	Characteristic biophysical features
Betula utilis	30–65	NE-N-NW	3300–4200	Early successional tree species in shady slopes forms broad ecotone with *krummholz* in undisturbed areas
Quercus semecarpifolia	25–40	NW-W-SW-S	3200–3700	Strong sunny aspects with gentle slopes, thick soil layer, more along the exposed ridges, sharp timberlines
Abies spectabilis	25–45	NW-N	3200–3500	In the shady aspects, gentle to steep slope, sharp timberline
Pinus wallichiana	30–60	W-SW-S	3500–3800	Dry slopes, inner dry valleys, generally orographic timberlines
Rhododendron arboreum	40–70	SW-S-SE	3300–3500	Steep rocky areas, beyond the limits of *Q. semecarpifolia*, stunted individuals towards alpine area forms ecotone

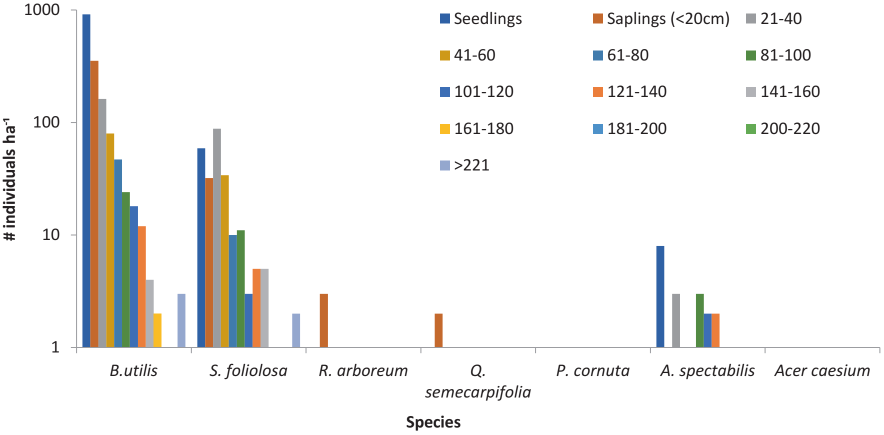

Fig. 9.3 Population structure of *B. utilis* communities

occasionally *Q. semecarpifolia*, *Acer caesium* and *R. arboreum*. Mean tree density was 553 individuals ha^{-1} which ranged between 440 and 690 individuals ha^{-1}. Density of the *B. utilis* ranged between 200 and 540 individuals ha^{-1}. Average TBA of *B. utilis*-dominated community in the study area was 17.8 ± 9.8 m^2 ha^{-1} along the treeline ranging between 6.4 and 33.3 m^2 ha^{-1}, which was higher in the old growth stands in anthropogenic and orographic treelines. Population structure of the community reveals that in the many protected areas, there is high seedling and sapling density of *B. utilis* along the treeline ecotone which indicates ameliorated conditions for recruitments and growth of saplings in the recent past (Fig. 9.3). This condition might be attributed to the change in land use after law enforcement, whereas very poor regeneration and no regeneration were observed in the disturbed treeline. Establishment of seedlings beyond the canopy forest at treeline towards alpine meadows was also evident. Earlier studies have indicated that *B. utilis* has higher freezing tolerance and is well suited to grow around alpine treelines (Sakai and Larcher 1987; Cairns and Moen 2004).

Silver fir (*A. spectabilis*) was more abundant in HP. Tree density in fir patches ranged between 420 and 780 (mean 660 ± 207.8) individuals ha^{-1}. TBA of the community was between 22.7 and 55.6 (mean 39.9 ± 16.3) m^2 ha^{-1}. TBA was lower in steep slope and protected forests. Regeneration status was very poor in *A. spectabilis* communities. The conversion rate of seedlings to saplings was very low, and treeline was generally depressed in the absence of natural regeneration (Fig. 9.4). Treelines formed by *Q. semecarpifolia* were mostly sharp and influenced by anthropogenic pressures. Mean density of *Q. semecarpifolia*-dominated timberline was 516 ± 158.2 individuals ha^{-1} which ranged between 340 and 750 individuals ha^{-1}. The total basal area varied between 9.6 and 47.3 (mean 28.8 ± 14.5) m^2 ha^{-1}. A good regeneration was observed in undisturbed, steep slopes/inaccessible sites and protected areas. Establishment of seedlings and saplings beyond the canopy and towards the alpine meadows was observed in several localities (Fig. 9.5). Thus, Q. *semecarpifolia* has the potential to form prominent a treeline ecotone on

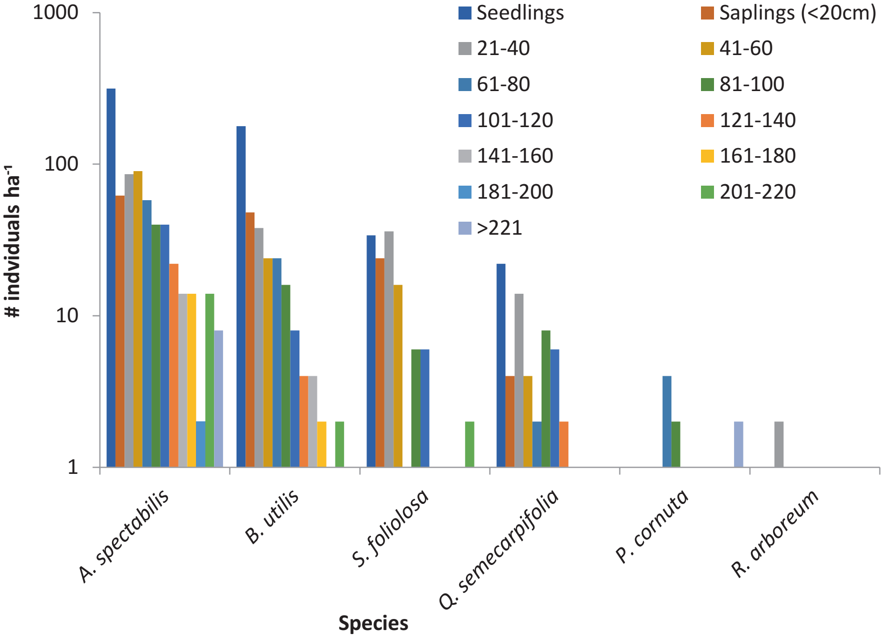

Fig. 9.4 Population structure of *A. spectabilis* communities

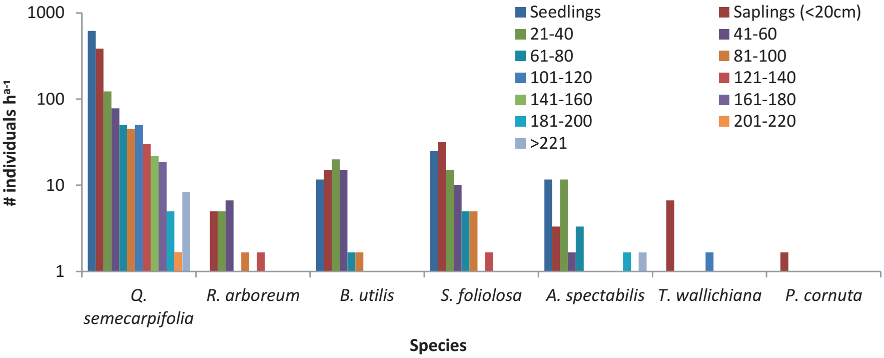

Fig. 9.5 Population structure of *Q. semecarpifolia* communities

south-facing slopes of Western Himalaya. Interestingly, no other oak species forms a natural treeline in the northern hemisphere. The only other *Fagaceae* species known to form continuous treeline is *Nothofagus pumilio*, in high ranges of Australia and New Zealand (Mark et al. 2008). Long-term monitoring of populations of this species especially along natural treeline would be desirable. Presently, its regeneration is severely affected due to chronic stress (Singh 1998; Rai et al. 2012).

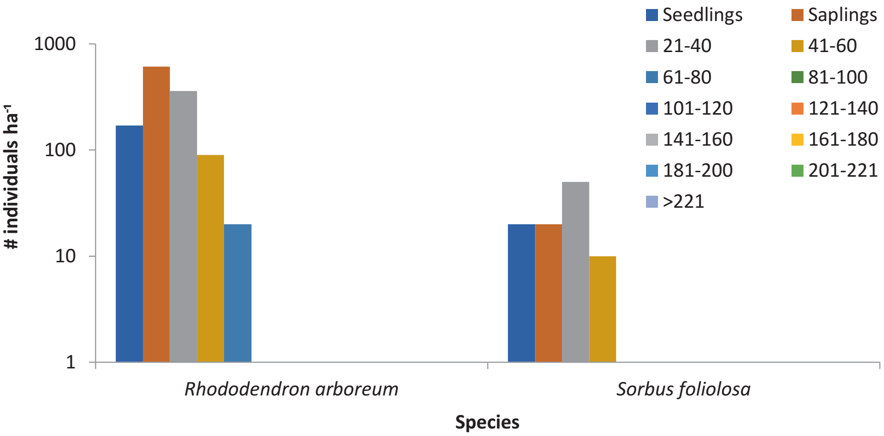

Fig. 9.6 Population structure of *Rhododendron arboreum* community

R. arboreum formed treeline in very steep, warm, south- to southwest-facing rocky slopes. Due to sparse distribution of trees and short statures in stressed conditions, the TBA was very low (4.6 m^2 ha^{-1}). Though the tree density in the community was 530 individuals ha^{-1}, the sapling density (610 individuals ha^{-1}) indicates good regeneration of *R. arboreum* along the ecotone (Fig. 9.6) and can be visualized in some upward movement and more growth along the timberline than other species.

9.3.3 Regeneration Status of Tree Species

The forest patches surveyed for regeneration status of treeline tree species (viz. *B. utilis*, *A. spectabilis* and *Q. semecarpifolia*) revealed the lack of natural regeneration in most of the survey localities in the Western Himalaya except at the protected sites. High seedling (1450–1860 individuals ha^{-1}) and sapling densities (930–400 individuals ha^{-1}) were observed in case of *B. utilis* at the sites with negligible anthropogenic pressures. Such sites exhibited near natural population structure (reverse ‘J-‘shaped curve). The ratios of sapling/seedling and tree/sapling were 0.4 and 0.6, respectively, indicating a steady rate of conversion from the young to mature individuals. Treelines along high anthropogenic pressures were characterized by low (0.26) sapling/seedling ratio and high (2.0) tree/sapling ratio. Sites having high anthropogenic pressures were also characterized by the absence of *krummholz* vegetation and very poor regeneration of timberline tree species especially *A. spectabilis*. Undisturbed *Q. semecarpifolia*-dominated treelines were rather difficult to find (except on steep slopes >60^0). Edges of Kharsu oak timberlines at disturbed sites did have high abundance of seedlings, but there was low conversion to sapling. Only on steeper and inaccessible terrain there was higher recruitment to saplings and pole size crop.

9.3.4 Recent Changes in Treeline Ecotone

Ecology of timberline has gained greater attention of ecologists during the past few decades owing to their high sensitivity to changing climate. Several studies indicate that geographical shift of timberlines towards higher altitude is less frequent than expected (Crawford, 1997; Lloyd and Graumlich 1997; Peterson 1998; Cullen et al. 2001; Camarero and Gutiérrez 2004). Response of timberline vegetation to changing environment can be seen in terms of changes in growth, growth forms and regeneration and changes in spatial heterogeneity. However, the observed results depend on the scale of study (Holtmeier and Broll 2005). Based on remote sensing analysis of Landsat MSS and TM images, it is evident that there has not been any significant altitudinal shift in West Himalayan treeline since the 1970s. However, canopy cover of forests has increased substantially at the sites with negligible anthropogenic pressure and inaccessible areas, e.g. inside Gangotri NP. In parts of Askot WS (Chhipla Kedar), treeline vegetation has gone under rapid changes during the last four decades (Table 9.3). Mostly these changes have occurred due to camping by livestock and humans. Similar observations have been made by other workers in Nanda Devi Biosphere Reserve (Adhikari et al. 2012) and Kedarnath WS (Singh et al. 2010). Increase in homogeneous reflectance pattern near upper boundary of timberline ecotone (above 4000 m asl) suggests more homogenous growth in the grasslands.

Comparison of mean and standard deviation of NDVI differencing images suggests that most of the changes have occurred below 3900 m elevation. The nature of change does not clearly indicate any upward shift of timberline species, as most of the changes are associated with forests of lower elevation (Fig. 9.7). The sites with negligible anthropogenic pressures do exhibit increased vegetation cover below 3900 m elevation. Field observations and binary change image suggest that in both cases (negligible and high anthropogenic pressure) most changes in the last four decades are related to *Betula utilis*. Results of the study are similar to the findings of Zhang et al. (2009) that the impact of changing climate during the last few decades has not caused the geographical shift of timberline vegetation but increase in density of trees especially in the case of *Betula*.

The high-altitude fir (*A. spectabilis*) forms gregarious patches, especially on shady moist and rocky slopes, along alpine treeline in the Western Himalaya, and intergrades into silver fir (*A. pindrow*). Owing to spectral similarity with *A. pindrow*, it was not possible to conduct time series analysis of this species in the present

Table 9.3 Increased/decreased area at protected (Gangotri NP) and high anthropogenic sites (Chhipla Kedar)

	Gangotri NP		Chhipla Kedar	
Duration (years)	Increased area (sq. km)	Decreased area (sq. km)	Increased area (sq. km)	Decreased area (sq. km)
1976–1996	1.97	1.43	2.64	3.23
1996–2016	5.74	0.41	8.7	2.76
1976–2016	6.61	0.74	6.04	4.49

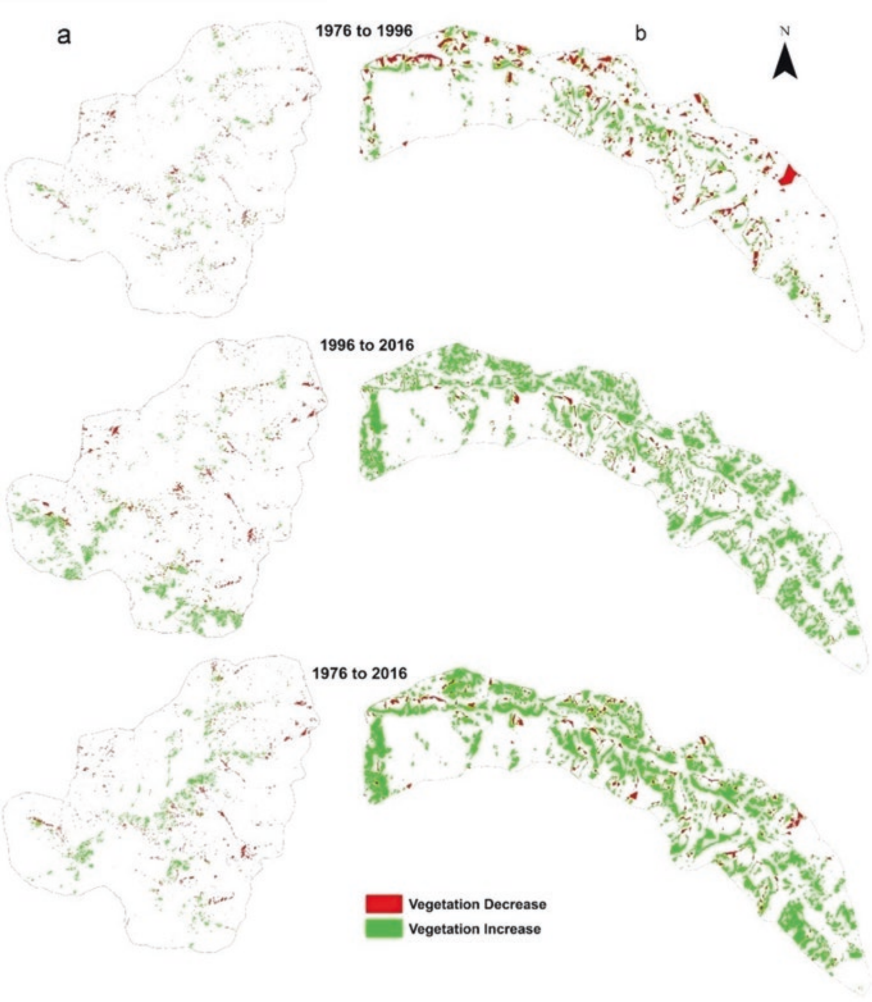

Fig. 9.7 Temporal change in forest cover in subalpine forest (increase/decrease) between 3000 and 4000 m: (**a**) high anthropogenic site in Chhipla Kedar (fringes of Askot WS) and (**b**) protected site in Gangotri NP

study. Moreover, owing to its occurrence on steeper shady slopes, detection of change in its areal extent was not feasible. This implies that the long-term ecological monitoring of this species using remote sensing would require more training sites and rigorous ground-truth data (Bharti et al. 2012). Other conifers touching the timberline in this region include *P. wallichiana*, *P. gerardiana* and *Juniperus polycarpos* (Singh et al. 2002). However, their status and response to anthropogenic and climatic drivers are yet to be ascertained. It is believed that in recent decades blue pine has shown rapid spread towards higher altitudes (e.g. Padma 2014; Dubey et al. 2003). We observed that within Gangotri National Park, *P. wallichiana* has successfully been established towards treeline in recent decades. Thus strictly protected sites do reflect an overall increase in the vegetation cover along timberline ecotone in the last four decades (1976–2016).

9.4 Conclusion

From the preceding analysis, it is evident that the average elevation of treeline in the Western Himalaya is around 3615 m asl, but trees are recorded up to 4200 m. Within the subalpine zone, high-altitude fir occupies the maximum area (1719.3 km^2) followed by kharsu oak (1461.7 km^2) and birch (340.6 km^2). The sites exhibiting the natural (climatic) treelines in the Western Himalaya are confined to a few strictly protected and inaccessible areas. The single mountain slopes providing adequate representation of timberline ecotone, alpine scrub and meadows covering wide altitudinal range free from chronic anthropogenic pressures are rather sparse in the Western Himalaya.

Most of the extensive patches of birch were seen in protected areas or on steeper inaccessible slopes. Within the Greater Himalayan range of Uttarakhand and Himachal Pradesh, we estimate that about 340.6 km^2 is occupied by birch forest. Regeneration of birch in most of the gentle slopes was virtually absent primarily owing to heavy browsing by domestic sheep and goats during summer season. It was evident from the occurrence of birch patches on flat areas in strictly protected places such as Gangotri NP. Though birch is one of the most widespread and well-adapted species for treeline environment in the Himalaya, its ecological status and vulnerability to climate change are yet to be studied at length. The presence of scattered and small patches of birch in few pockets of trans-Himalaya, e.g. Lahaul and Spiti, Nelang valley in Uttarakhand and parts of western Ladakh, calls for a detailed mapping and tracing back of ecological history of this species. Considering the immense ecological role of birch in the Himalayan region in terms of environmental stability and habitat for several endangered species of fauna, this species deserves participatory monitoring using citizen science approach and restoration of degraded sites.

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References

Adhikari BS, Rawat YS, Singh SP (1995) Structure and function of high altitude forests of central Himalaya I. Dry matter dynamics. Ann Bot 72:237–248

Adhikari BS, Rawat GS, Rai ID, Bhattacharyya S, Bharti RR (2012) Ecological assessment of timberline ecotone in Western Himalaya with special reference to climate change and anthropogenic pressures. Unpublished Report, Wildlife Institute of India, Dehradun

Arno SF (1984) Timberline. Mountain and Arctic forest frontiers. Mountaineers Books, Seattle

Bharti RR, Adhikari BS, Rawat GS (2012) Assessing vegetation changes in timberline ecotone of Nanda Devi National Park, Uttarakhand. Int J Appl Earth Obs 18:472–479

Bharti RR, Rai ID, Adhikari BS, Rawat GS (2011) Timberline change detection using topographic map and satellite imagery: a critique. Trop Ecol 52(1):133–137

Cairns D, Moen J (2004) Herbivory influence tree lines. J Ecol 92:1019–1024

Camarero JJ, Gutiérrez E (2004) Pace and pattern of recent treeline dynamics: response of ecotones to climatic variability in the Spanish Pyrenees. Climate Change 63:181–200

Chhetri PK, Cairns DM (2016) Dendroclimatic response of Abies spectabilis from the treeline ecotone of Barun valley, eastern Nepal Himalaya. J For Res 27:1163–1170

Crawford RMM (1997) Consequences of climatic warming for plants of the northern and polar regions of Europe. Flora Colon 5(6):65–78

Cullen LE, Stewart GH, Duncan RP, Palmer JG (2001) Disturbance and climate warming influences in New Zealand *Nothofagus* tree-line population dynamics. J Ecol 8:1061–1071

Curtis JT, McIntosh RP (1950) The interrelation of certain analytical and synthetic phytosociological character. Ecology 31:434–455

Däniker A (1923) Biologische Studien über Baum- und Waldgrenzen, insbesondere über die klimatischen Ursachen und deren Zusammenhänge. Vierteljahresschr Naturforschenden Ges Zürich 68:1–102

Ellenberg H (1966) Leben und Kampf an den Baumgrenzen der Erde. Naturwiss Rundsch 19:133–139

Dubey B, Yadav RR, Singh J, Chaturvedi R (2003) Upward shift of Himalayan pine in Western Himalaya, India. Curr Sci 85:1135–1136

Fraser RH, Olthof I, Carriere M, Deschamps A, Pouliot D (2011) Detecting long-term changes to vegetation in Northern Canada using the Landsat satellite image archive. Environ Res Lett 6. https://doi.org/10.1088/1748-9326/6/4/045502

Gaire NP, Koirala M, Bhuju DR, Carrer M (2017) Site- and species-specific treeline responses to climatic variability in eastern Nepal Himalaya. Dendrochronologia 41:44–56

Garkoti SC, Singh SP (1994) Nutrient cycling in the three central Himalayan forests ranging from close canopied to open canopied treeline forests, India. Arct Alp Res 26:339–348

Gehrig-Fasel J, Guisan A, Zimmermann NE (2007) Tree line shifts in the Swiss Alps: climate change or land abandonment. J Veg Sci 18:571–582

Grace J, Berninger F, Nagy L (2002) Impacts of climate change on the tree line. Ann Bot 90:537–544

Griggs RF (1937) Timberlines as indicator of climatic trends. Science 85(2202):251–255

Holtmeier FK (2003) Mountain timberlines– ecology, patchiness, and dynamics advances in global change research, vol 14. Kluwer Academic, Dordrecht

Holtmeier FK, Broll G (2005) Sensitivity and response of northern hemisphere altitudinal and polar treelines to environmental change at landscape and local scales. Glob Ecol Biogeogr 14:395–410

Holtmeier FK, Broll G (2007) Treeline advance driving processes and adverse factors. Landscape Online 1:1–33. https://doi.org/10.3097/L0.200701

Jeffrey GM (2001) Stability of boreal forest stands during recent climate change: evidence from Landsat satellite imagery. J Biogeogr 28:967–976

Kershaw KA (1973) Quantitative and dynamic plant ecology. Edward Arnold Ltd, London, p 308

Körner C (1998a) A re-assessment of high elevation treeline position and their explanation. Oecologia 115:445–459

Körner C (1998b) World-wide positions of alpine treelines and their causes. In: Beniston M (ed) The impacts of climate variability on forests. Springer, Heidelberg, pp 221–229

Körner C (2003) Carbon limitation in trees. J Ecol 91:4–7

Körner C (2012) Alpine treelines. Functional ecology of the global high elevation tree limits. Springer, Basel. 220p

Körner C, Paulsen J (2004) A world-wide study of high altitude treeline temperatures. J Biogeogr 31:713–732

Kullman L (1998) Tree-limits and montane forests in the Swedish Scandes: sensitive biomonitors of climate change and variability. Ambio 27:312–321

Kullman L (2001) 20th-century climate warming and tree limit rise in the Southern Scandes of Sweden. Ambio 30:72–80

Larsson H (1993) Linear regressions for canopy cover estimation in acacia woodlands using Landsat-TM, Landsat-MSS and SPOT HRV XS data. Int J Remote Sens 14:2129–2136

Lloyd AH, Graumlich LJ (1997) Holocene dynamics of treeline forests in the Sierra Nevada. Ecology 78(4):1199–1210

Mark AF, Proter S, Piggot JJ, Michel P, Maegli T, Dickinson KJM (2008) Altitudinal pattern of vegetation, flora, life forms, and environment in the alpine zone of the fiord ecological region, New Zealand. New Zeal J Bot 46:205–237

Masek JG (2001) Stability of boreal forest stands during recent climate change: evidence from Landsat satellite imagery. J Biogeogr 28:967–976

Miehe G, Miehe S (1994) Zuroberen Waldgrenze in tropischen Gebirgen. Phytocoenologia 24:53–110

Misra R (1968) Ecology work book. Oxford and IBH Publ, New Delhi

Malanson GP (1997) Effects of feedbacks and seed rain on ecotone patterns. Landsc Ecol 12:27–38

Motta R, Nola P (2001) Growth trends and dynamics in subalpine forest stands in the Varaita Valley (Piedmont, Italy) and their relationships with human activities and global change. J Veg Sci 12:219–230

Muller-Dombois D, Ellenberg H (1974) Aims and methods of vegetation ecology. Wiley, New York

Myneni RB, Keeling CD, Tucker CJ, Asrar G, Nemani RR (1997) Increased plant growth in the northern high latitudes from 1981 to 1991. Nature 386:698–702

Ohsawa M (1990) An interpretation of latitudinal patterns of forest limits in South and East Asian mountains. J Ecol 78:326–339

Padma TV (2014) Himalayan plants seek cooler climes. Nat Lett 512:359

Panigrahy S, Anitha D, Kimothi MM, Singh SP (2010) Timberline change detection using topographic map and satellite imagery. Trop Ecol 51(1):87–91

Parmesan C (2006) Ecological and evolutionary responses to recent climate change. Annu Rev Ecol Evol Syst 37:637–669

Paulsen J, Weber UM, Körner C (2000) Tree growth near treeline: abrupt or gradual reduction with altitude? Arct Antarct Alp Res 32:14–20

Peterson DL (1998) Climate limiting factors and environmental change in high altitude forests of Western North America. In: The impact of climate variability in forests. Springer, Heidelberg, pp 198–208

Rai ID (2012) Ecological Attributes of timberline vegetation with special reference to climatic variability in Kedarnath Wildlife Sanctuary, Western Himalaya, Ph.D Thesis, Kumaun University, Nainital

Rai ID, Adhikari BS, Rawat GS, Bargali K (2012) Community structure along timberline ecotone in relation to micro-topography and disturbances in Western Himalaya. Not Sci Biol 4(2):41–52

Rawal RS, Dhar U (1997) Sensitivity of timberline flora in Kumaon Himalaya, India: conservation implications. Arct Alp Res 29:112–121

Sakai A, Larcher W (1987) Frost survival of plants. Responses and adaptation to freezing stress. Ecological studies 62. Springer, Berlin

Schickhoff U (2005) The upper timberline in the Himalayas, Hindu Kush and Karakorum: a review of geographical and ecological aspects. In: (Broll, G. and B. Keplin eds.) mountain ecosystems. Studies in treeline ecology. Springer, Berlin/Heidelberg/New York, pp 275–354

Schweinfurth U (1957) Die horizontale und vertikale Verbreitung der Vegetation im Himalaya. Bonner Geogr Abh 20

Shrestha BB, Shimire B, Lekhak HD, Jha PK (2007) Regeneration of treeline Birch (*Betula utilis* D.Don) forest in a Trans-Himalayan Dry Valley in Central Nepal. Mt Res Dev 27(3):259–267

Shugart HH, French NHF, Kasischke ES, Slawski JJ, Dull CW, Shuchman RA, Mwangi J (2001) Detection of vegetation change using reconnaissance imagery. Glob Chang Biol 7:247–252

Singh G, Rawat GS, Verma D (2010) Comparative study of fuelwood consumption by villagers and seasonal "Dhaba owners" in the tourist affected regions of Garhwal Himalaya, India. Energy Policy 38:1895–1899

Singh JS, Singh SP (1992) Forests of Himalaya. Structure, functioning and impact of man. Gyanodaya Prakashan, Nainital

Singh SP (1998) Chronic disturbance, a principal cause of environmental degradation in developing countries. Environ Conserv 25:1–2

Singh NP, Singh DK, Uniyal BP (2002) Flora of Jammu & Kashmir, vol 1. Botanical Survey of India, Dehra Dun

Singh CP, Panigrahy S, Thapliyal A, Kimothi MM, Soni P, Parihar JS (2012) Monitoring the alpine treeline shift in parts of the Indian Himalayas using remote sensing. Curr Sci 102(4):559–562

Thakur AK, Singh G, Singh S, Rawat GS (2011) Impact of pastoral practices on forest cover and regeneration in the outer fringes of Kedarnath Wildlife Sanctuary, Western Himalaya. J Indian Soc Remote Sens 39(1):127–134

Tranquillini W (1979) Physiological ecology of the alpine timberline. In: Tree existence at high altitudes with special references to the European Alps, ecological studies, vol 31. Springer, Berlin

Troll C (1973) The upper timberlines in different climatic zones. Arct Alp Res 5(3):3–18

Wardle P (1993) Causes of alpine timberline: a review of the hypotheses. In: Alden J, Mastrantonio L, Odum S (eds) Forest development in cold climates. Plenum Press, New York, pp 89–103

Wardle P (1974) Alpine timberlines. In: Ives JD, Barry RG (eds) Arctic and alpine environments. Methuen, London, pp 371–402

Weiser J, Tuasz M (2007) Current concepts for tree life limitation at the upper timberline. In: Wieser G, Tauz M (eds) Tree at their upper limit. Springer, New York, pp 1–18

Xu B, Gong P, Pu R (2003) Crown closure estimation of oak savannah in a dry season with Landsat TM imagery: comparison of various indices through correlation analysis. Int J Remote Sens 24:1811–1822

Yadava AK, Sharma YK, Dubey B, Singh J, Singh V, Bhutiyani MR, Yadav RR, Misra KG (2016) Altitudinal treeline dynamics of Himalayan Pine in Western Himalaya, India. Quat Int. https://doi.org/10.1016/j.quaint.2016.07.032

Zhang Y, Xu M, Adams J, Wang X (2009) Can Landsat imagery detect tree line dynamics? Int J Remote Sens 30:1327–1340

Ideas and Approaches for Studying Treelines in the Himalayas: Expanding the Concept from a Landmark to Community and Ecosystem Levels

10

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Abstract

Here, we report research design, methodological framework and preliminary findings of a treeline research, based on several sites covering much of the Indian Himalayas. Apart from investigating tree populations, this study expands the treeline concept to community and ecosystem levels. A treeline ecotone consists not only of trees but also several species of herbs and other growth forms. The paper sheds light on methods to delineate treelines and timberlines and to characterize species distribution patterns. Treeline landscapes have complex mosaics of ecosystems, diverse in physiognomy, such as forests, juniper mats and rhododendron krummholz, isolated trees and trees in clusters and rows. Several ecosystem characters change rapidly as forests approach treelines, including a drastic drop in biomass accumulation, which leads to treeline formation. A compilation of treeline elevation studies at the regional level indicates that its elevation can vary widely, by 1800 m across the Himalayan Arc. A similar elevational range is indicated by mapping with the help of remote sensing. The common treeline genera are *Betula*, *Abies*, *Picea* and *Juniperus*, forming almost all treeline forms described in the literature. We found a relatively lower Temperature Lapse Rate (TLR) value, partly because of elevation-dependent warming (EDW) which needs to be analysed. The upward movement of treelines due to climatic warming could be restricted by pre-monsoon droughts as the studies based on tree ring width chronology suggest. During winters, trees have access to water for only a limited part of the day as the soil remains frozen most of the time. Response to early snowmelt seems to vary from species to species.

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Keywords
Climate change · Dependent warming · Elevation · Phenology · Pre-monsoon drought · Temperature lapse rate · Treeline-timberline mapping · Tree-water relation

10.1 Introduction

Named variously, treeline, timberline and forest line, the high altitude limit of forests and trees (Körner 1998; Holtmeier 2009) is a dramatic physiognomic discontinuum along a temperature gradient. Heat deficiency with increase in altitude is considered to be the main cause of a woody plant not attaining tree form beyond an elevation in mountains. Globally, treelines are linked to a growing season mean temperature of 6.7 °C ± 0.85 °C in mountains (Körner and Paulsen 2004). Thus, global warming is expected to result in an upward movement of treelines in mountains. That is why the dramatic drop in the size of vegetation from trees to herbs and other short forms in high mountains has been drawing widespread attention from researchers in recent years (e.g. Gaire et al. 2014; Liang et al. 2011; Liu et al. 2013; Schickhoff et al. 2015).

Though the highest treeline of the Northern Hemisphere occurs in the Himalayas (*Juniperus tibetica* at 4900 m in the Tibet Plateau; Miehe et al. 2007), the terms, treelines and timberlines, have been missing in the literature on vegetation in the Himalayas, until recently. Being higher than in most other regions, treelines and timberlines in the Himalayas are remotely located, rendering scientific measurements and monitoring difficult and expensive.

In general, a timberline in the mountains is considered the upper elevational limit of forests with crown cover, at least between 20% and 40%, while a treeline is regarded as an imaginary line connecting the uppermost trees at a site (Holtmeier 2009). Here, a tree is generally an upright individual of tree species with a minimum height of 2 m or more (Körner 2012). When a forest limit is abrupt, the timberline and treeline are the same. However, often trees do not stop to grow abruptly as elevation increases; instead, they become gradually sparser and shorter on high mountains. In that case, treelines are higher than timberline, generally by 50–200 m in the Himalayas (Singh and Rawal 2017). Individual trees may grow still higher, but their form is no longer tree-like; they become stunted and contorted, with several stems and with twisted and gnarled branches. This is called tree species line.

In the context of climate change, treeline responses generally have been investigated in terms of upslope advancement of trees, densification of existing tree stands and tree ring width chronologies, with focus on their relationship with temperature and precipitation (Schickhoff et al. 2015). Globally, in about 52% studies, an upward movement of trees has been reported, while in most of the remaining studies, it has remained stationary (Harsch et al. 2009). However, not all advances of treeline species are ascribed to climatic warming; many are believed to be due to the decrease in pastoralism (Schickhoff et al. 2015). Trees being long-lived have a long time lag to track warm temperatures, so tree sensitivity to climate change in terms of upslope migration may not be expressed immediately in all situations. In all such studies on high mountains, a treeline is treated merely as a kind of landmark.

Treeline-timberline areas are also important from the standpoint of studies of ecosystem attributes and ecosystem services, such as hydrologic regulation and carbon storage. Under the influence of warming both gross ecosystem productivity and net ecosystem, CO_2 exchange is reported to increase significantly in Tibet Plateau (Ganjurjav et al. 2015). Speed et al. (2015) have found that total ecosystem carbon shows a discontinuum pattern with elevation; it increases with elevation above the treeline, but decreases with elevation below forest line. Therefore, total carbon is lowest between forest line and treeline. The changes in cryosphere are likely to affect treeline areas and alpine meadows at community and ecosystem levels (Greenwood and Jump 2014). For example, on concave surfaces the longer the snowbed remains on the ground, the less time the tree has for growth. However, the loss of snow due to climatic warming may result in tree establishment in such places.

In this paper, we have briefly discussed the possible methods which could be applied to study treeline-timberline vegetation with the view of expanding treeline studies from a simple tree population level to community and ecosystem levels. This approach could improve our understanding and management of treeline areas in the Himalayas, being increasingly affected by climate change impact. Theoretically, a line is one-dimensional, but trees occupy some ground surface with some widths, and like other communities they grow with and support several other species of varied forms, like shrubs, herbs, lichens and mosses.

While describing treelines, we have considered Temperature Lapse Rate, tree ring width changes in relation to climate, tree-water relations, phenological change, plant species distribution and mapping of populations not only of trees but also of associated herbs and other growth forms more sensitive to climate change than trees in terms of time lag (Ge et al. 2015). These ideas and approaches have emerged from a multidisciplinary research on Himalayan treelines with emphasis on team science. Expanding the treeline from a mere population of trees to include concepts and methods of vegetation science and ecosystem analysis and applying gradient analysis to spatial changes towards treelines surface up prominently in this article.

10.2 The Study Region, Approach and Methods

The Himalayas are not only massive and vast but also highly heterogeneous. Within the region (Himalayas plus Hindu Kush), the latitudinal range is 16° and longitudinal range, above 35°; precipitation generally varies from less than 500 mm in some of the areas north of the main Himalayan ranges to more than 4000 mm in monsoon-drenched southern slope of Eastern Himalayas (Fig. 10.1). Generally, monsoon weakens from the East to West; however, conditions vary in each part because of topographic factors. So, in areas shielded by high mountains, the monsoon months (June to September) account for less than 40% of annual rainfall; Kashmir Valley

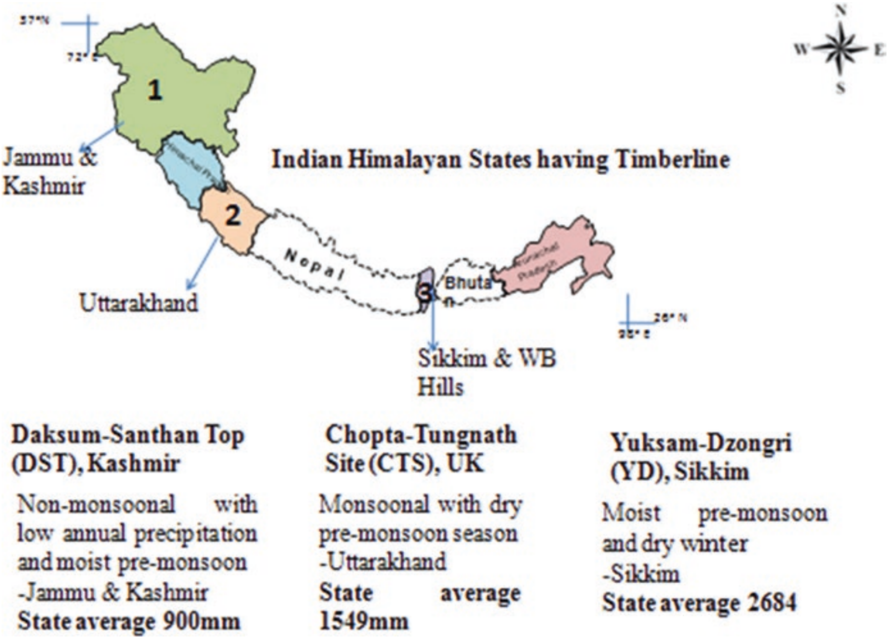

Fig. 10.1 Three timberline study sites. The study sites cover much of the range of variation across the Himalayan Arc and represent three precipitation regimes. (Courtesy: Subrat Sharma, GBPNIHESD, Almora)

surrounded by high Pir Panjal ranges is a typical example of this. However, in much of the Himalayas, monsoon months account for 70–80% of annual rainfall. The Himalayas have been warming for the last two to three decades at rates significantly higher than the global average rate and with the warming rate increasing with altitude (Shrestha et al. 2012; Yao et al. 2012). A rapid glacier shrinkage and snowmelt and glacier lake formation are common features of the high Himalayas in a warming climate. Subalpine forests and treeline vegetation play an important role in regulating hydrology of high mountains.

The average elevation in the Greater Himalaya is above 6100 m, which indicates why it has numerous glaciers of varying dimensions. The Himalayas and its surrounding regions together have more snow than any other parts of the world outside the two poles, so it is referred to as the third pole. As many as 10 river basins originate from the Himalayas, supporting about 1 billion people. A population density over 1000 persons per km^2 is common in the lower half of the Gangetic.

The common treeline-timberline genera are *Betula*, *Juniperus*, *Abies*, *Rhododendron* and *Picea*. Occasionally, blue pine (*Pinus wallichiana*), a deciduous conifer, *Larix*, and evergreen oak (*Quercus semecarpifolia*) also go up to treelines. Treeline areas are affected because of pastoralism, tourism and now by mass collection of *Cordyceps sinensis* (a fungus forming an ascocarp that emerges from the head of a caterpillar).

Learning how to develop a team to conduct research in remote areas was quite challenging. First, we formed a multidisciplinary team of investigators, consisting of (i) biodiversity experts, representing all the three major regions of the Indian part of the Himalayan Arc, Kashmir (western part), Uttarakhand (central part) and Sikkim (eastern part) (Fig. 10.1); (ii) an expert on the application of remote sensing techniques to treeline distribution; (iii) tree-ring width chronologists; (iv) a tree-water relation scientist; (v) a phenology expert; (vi) a climatologist; (vii) an ecologist; and (viii) an expert on working with local communities on livelihoods. The team was drawn from Kashmir University, Srinagar; Kumaun University, Nainital; Wildlife Institute of India, Dehradun; GB Pant National Institute of Himalayan Environment and Sustainable Development, Almora; Birbal Sahni Institute of Paleobotany, Lucknow; and Central Himalayan Environment Association, Nainital. Investigators were encouraged to go to study sites in groups, to exchange their ideas and field knowledge with the coordinator and among themselves. Two brainstorming workshops were conducted to work out methods, time schedule, specific sites and other issues. The second workshop was held at Tungnath, one of the three study sites, located in Uttarakhand with the purpose of examining whether the methods chosen in the first workshop would actually work in field conditions. Necessary modifications were made in sampling design, number of replicates, timing and sample points. The team along with research students met again to refine methods further based on the learnings within the first 6 months. The workshop helped us in finalizing finer details of methods and understanding how to expand the treeline concept to include community and ecosystem study elements. Studies had specific purposes (Table 10.1), and wherever possible complementary approaches were factored in. For example, we examined the critical role of pre-monsoon drought, which has been intensified by climate warming by undertaking studies both on tree-water relations and tree-ring width chronology. The collaborators were in continuous touch with one another, and necessary steps were taken whenever there were sampling problems.

10.3 Results and Discussion

Since our study is still in an early stage, here we are largely describing what we are doing to study Indian Himalayan Treelines, rather than focussing on the findings.

10.3.1 Timberline Distribution Map and Temperature Lapse Rate

We aimed at preparing a regional timberline map, so that generalizations for the Himalayan region could be made. Using Landsat images of 30 m resolution, timberlines were mapped. Though imageries of 1 m resolution are now available, 30 m resolution was used because it allowed to assess changes from the past data, which too had 30 m resolution. For the first time, as far as we know, a regional treeline is being mapped. The elevation range of timberline in the Himalayan region is wide,

Table 10.1 Studying treelines in Indian Himalayan Region (IHR) – study components and purpose

Study components	Purpose and remarks
Mapping the treeline distribution in the IHR, using satellite imageries and comparing with the past	To compare changes in elevation of treeline at the regional level, and over time; and giving information on treeline species, elevations and treeline forms
Temperature lapse rate (TLR) based on observed temperature data along a transect	To develop the first observed data-based TLR; examining how it differs from the west to east along the arc, and seasonally, and analyse their implications to vegetation pattern and treeline elevation
Phenology of trees, namely, fir, kharsu oak and rhododendron	To find out how the timing of phenophases is changing in response to global warming and snow melt/snow depletion
Tree and soil water relations	To understand how tree species adapt to water stress in cold climate? How do tree-water relations vary because of difference in the monsoon contribution to annual rainfall?
Tree ring width chronology in the past	To find out whether warmer temperatures and resultant longer growing period promote tree growth or suppress tree growth by intensifying water stress through increased evapotranspiration
Patterns of community and ecosystem changes along altitudinal gradient centred around treeline; delineating treelines and timberlines	To know the patterns in species composition and biodiversity across treelines; examine also species of other growth forms for their indicator value for climate change. How non-tree species populations are distributed in relation to the physiognomic discontinuum of treeline? Is the rapid decrease in biomass accumulation towards treeline a common feature?
Snow depletion and plant growth and species distribution	To find out how depletion of snow is likely to affect species growth and species distribution

approaching 1800 m across the timberline sites, because apart from temperature, it is also affected by anthropogenic pressure, topography, tree species and other factors, such as aspect and landslides (Schickhoff 2005). In Sikkim, we are documenting timberline and timberline formed by *Abies densa* within 20 km area. Ranging from about 3800 m to 4011 m, in about half of 25 transects sampled, timberline/treeline was abrupt, and in the rest it was diffused type, with the uppermost trees generally occurring 50–155 m elevations above timberline (Fig. 10.2). Usually, in diffuse treelines, trees become sparser and smaller as elevation increases. However, a detailed mapping indicates that a treeline ecotone can be complex. The highest tree can be older than trees next to timberline. In Kashmir, generally treeline is the diffused type (Fig. 10.3), although finger-like and island-like treelines also occur (Fig. 10.4). The treeline oak, *Quercus semecarpifolia*, generally forms abrupt treelines to which grazing pressure also contributes (Fig. 10.5).

The remote sensed maps have shown how little we can know about timberline elevations and their spatial pattern by sampling manually. In fact, field sampling at a few sites disinforms about the regional position.

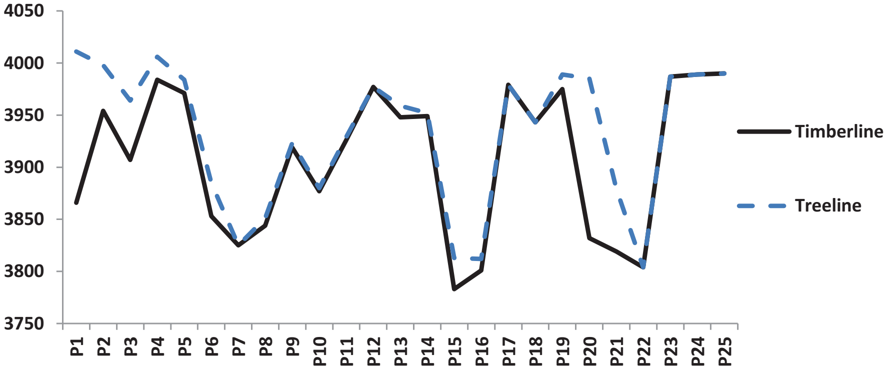

Fig. 10.2 A representation of *Abies densa* (fir) treeline (broken line) and timberline (solid line) in Sikkim. Plots (P1 to P25) were sampled (each of 50 × 20 m) across 9 sites (mostly 3 transects a site) over about 20 km distance. In timberline tree crown cover is at least 30%, treeline connects the highest trees of each transect, and distances between timberline and treeline range from 0 to 155 m. At many sites treeline and timberline are the same, thus distance being 0. Details of location and stem girth of other trees, saplings and seedlings are being developed. None of the seedlings and saplings extend beyond a treeline, so the treeline is unlikely to advance; the treeline and timberline ecotone can get densified. (Data source: GBP, Sikkim Unit of the project)

Fig. 10.3 *Betula utilis* and *Rhododendron campanulatum* forming treeline at Lata-Kharak Nanda Devi Biosphere Reserve 4000 m. (Courtesy: Vikram Negi, GBPNIHESD)

Fig. 10.4 Finger-like treeline of *Betula utilis* and *Abies pindrow*. The snow that accumulates in furrows stays up to late summer; thus snow-free period is insufficient for tree establishment. That is why trees are confined to rows on ridges, forming finger-like treeline (Courtesy: Prof. Zafar Reshi, Kashmir University)

Temperature Lapse Rate (TLR), the rate of decrease in temperature with increasing altitude, is one of the determiners of treeline elevation: the lower the TLR, the higher is the treeline elevation. It is being increasingly realized that TLR varies considerably across mountains and within a mountain across seasons. TLR in the Himalayas is relatively lower during monsoon months and might partly account for the higher treelines in the Himalayas (treelines above 4200 m are quite common). Here, two research questions are important: (i) are TLRs in the Himalayas relatively lower than in other mountain areas, and (ii) is TLR decreasing because of global warming? Elevation Dependent Warming (EDW), an increase in the rate of temperature rise with elevation, is being seen in most of the mountains in the warming climate (EDW Working Group 2015).

As far we know, this is the first TLR estimate based on observed data along an altitudinal transect in the Indian Himalayas. Winter data from Tungnath site of Uttarakhand indicated that TLR varies from one aspect to other. The TLR data and factors which influence them such as season and aspect may shed light on treeline dynamics. Generally, in the Himalayas treelines are higher on Southern aspect than the Northern. However, in extremely xeric Chitral (Pakistan), treelines go higher in the North aspect than the South aspect (Nüsser and Dickoré 2002).

Fig. 10.5 *Quercus semecarpifolia* forming abrupt timberline in Chaudas Valley (3800 m). Here canopy cover in the timberline is above 70%, and this is immediately followed by treeless alpine meadows. Grazing is considered the main contributor to the abrupt timberline, which is a characteristic feature of the kharsu oak timberline. (Courtesy: Vikram Negi, GBPNIHESD)

10.3.2 Tree-Level Responses to Climate

10.3.2.1 Plant Phenology

Being deficient in heat, the treeline habitat is expected to witness many changes in plant phenology because of global warming. Earliness of plant phenological phases, such as bud burst, leaf expansion and flowering, is quite common. Earlier leafing and delayed leaf drop are likely to prolong growth period, particularly in deciduous species, like *Betula utilis* (Negi GCS, Unpubl. Data). Data are being collected on the dates of various phenophases, recruitment and growth of krummholz forming rhododendron. It is possible to find out the age of the rhododendron (*Rhododendron campanulatum*) and the rate of their upslope movement on the basis of internode numbers (Negi GCS, Unpubl. Data). Growth initiation is likely to be considerably affected by the date of snowmelt, and its impact is being observed by comparing adjacent plots covered with snow and without snow for various plant phenophases.

Since plant growth is affected not only by temperature, but also by water availability, the study of tree-water relations was also investigated. In the Himalayas, treelines have multi-year evergreen conifer, fir (*Abies pindrow* and *A. spectabilis*), broadleaved krummholz rhododendron (*R. campanulatum*) and deciduous species, birch (*Betula utilis*). Occasionally, *Quercus semecarpifolia* also reaches timberline.

In this evergreen oak with about 1-year life span, formation of new leaves induces fall of old leaves. Therefore, while leaf exchange in oak may occur earlier in the warming climate, the length of growth period will not be lengthened (Singh and Singh 1992; Negi GCS, Unpubl. Data). To develop an understanding of treeline dynamics, growth phenology of different growth forms needs to be monitored for a long period. At the Tungnath site, permanent plots representing various growth forms have been established and their geographical coordinates recorded. In each plots, individuals and their parts have been marked for data collection for a 3-year period. Efforts are being made to take data for another 3 years.

10.3.2.2 Tree Ring Width Chronology

An increasing number of studies have been conducted in recent years on the relationship between tree ring width and climate. Some studies on *Pinus wallichiana*, *Cedrus deodara* and *Picea smithiana* on treeline sites have shown a marked growth enhancement during recent decades of warming (Singh and Yadav 2000; Borgaonkar et al. 2009, 2011). In contrast, some other studies have shown that warmer temperatures during the pre-monsoon (March to May) period intensify drought stress by increasing evapotranspiration loss, thus resulting in no growth or growth reduction (Gaire et al. 2011). Therefore, warmer temperatures of pre-monsoon periods are expected to have a positive impact on tree growth only in moist regions or when warming is accompanied by increased rainfall.

We are analysing tree ring width (silver fir, birch, deodar, rhododendron) in climates with weak and strong monsoon rainfall. To represent this variation in the timing of rainfall, we have chosen treelines in Kashmir (non-monsoonal) and Uttarakhand (strongly monsoonal). We expect that the difference in timing of rainfall between Kashmir and Uttarakhand should be reflected in different seasonal growth responses. Tree ring studies are also used to understand tree population dynamics and upward advancement of trees in relation to climate. A warm and moist year may lead to the recruitment of numerous individuals. However, not all tree species provide good tree ring samples. For example, in *Betula utilis*, the most common treeline species in much of the Himalayas, the inner part of the stem often gets rotten. Occurring in dry regions, the species is useful for finding out relationship between pre-monsoon water stress and tree ring growth. Dendrochronologists find *Cedrus deodara* very suitable, but it mostly remains below the timberline (Ranhotra et al., BSIP, Lucknow). Fir has been one of the most researched species for tree ring width analysis in relation to climate change.

Ranhotra and his team (BSIP, Lucknow) have collected a total of 60 core samples of *Abies spectabilis* and 256 core samples of *Betula utilis* from Uttarakhand and Kashmir treeline sites, of which the oldest samples are from a 460-year-old fir tree. The cross-dating and chronology development of the dated cores are under progress.

10.3.2.3 Tree and Soil Water Relations

How do evergreen tree species take water from cold soil to sustain wintertime photosynthesis in a treeline? To what extent the pre-monsoon water stress is likely to become a key factor in the warming world? These are some of the questions we are addressing. As far as we know, treeline water relations have not been investigated in the Himalayas before. Our early observations indicate that during winter cold, tree species depend more on subsurface soil water than on surface soil water, and trees encounter water stress more during morning hours of the day because of frozen soils than during daytime when it is thawed.

10.3.2.4 Methods for Studying Treeline as a Community

Treelines in mountains generally have been treated as a landmark, consisting of individuals of tree species, at the edge of alpine grasslands/meadows. The physiognomic discontinuum of treeline is set out by low temperatures, along an altitudinal gradient (Körner 1998). In other words, the physiognomic discontinuum is not due to a temperature discontinuum, but rather due to a certain temperature threshold below which trees cannot grow. Here, we are applying community-level methods, without going into the debate of community nature (discrete vs continuous) to timberlines and treelines. In this treatment, communities consist of many species of different growth forms; a treeline community type consists of a group of similar treeline stands, and communities are classifiable.

A line, theoretically, has no breadth; it has only length. However, trees and other plants occupy some breadth and area on ground and in canopy. This leads to the question: how far back from the upper side edge of a timberline vegetation can be included in a timberline? One simple answer could be as far back as vegetation is homogenous in composition. To study timberline vegetation, we need to sample stands or plots representing a timberline, each having homogenous species composition. Thus, a timberline can be given a thickness starting from the topmost line of trees to downslope vegetation similar in composition. A sampling design for this could be as following:

- Starting from the higher elevational end of timberline, run a 20-m-wide belt transect, divided continuously at 10 m interval. For a site, three to five such belt transects can be used to describe vegetation. The length of transect should be enough to sample the entire range of variation in vegetation: a part of alpine meadow, treeline and timberline.
- Sample within each 20 × 10 m division of belt transect with 100 m^2 quadrats for trees, 5 × 5 m for shrubs and 1 × 1 m for herbs (Fig. 10.6).
- For trees, place one 100 m^2 quadrat in each division; within the tree quadrat, sample shrubs and the saplings with three 5 × 5 m quadrats and herbs and seedlings with five 1 × 1 m quadrats. Out of the five randomly distributed quadrats, three could be distributed within the three shrub quadrats and two outside them randomly. For herbs, a point frame can be used to measure cover or coverage.
- For shorter forms, like bryophytes and lichens, a 10 × 10 cm frame with internal divisions of 1 cm could be used. This can be used to find out the cover percentage. For epiphytic ferns and angiosperms, sampling is limited to 1.5 m height of tree stems to minimize habitat variability and to find out comparable data. To have a homogenous sample, only trees of average sizes could be considered to sample epiphytes. In this approach, a treeline is considered as not only consisting of trees but also other plant species.

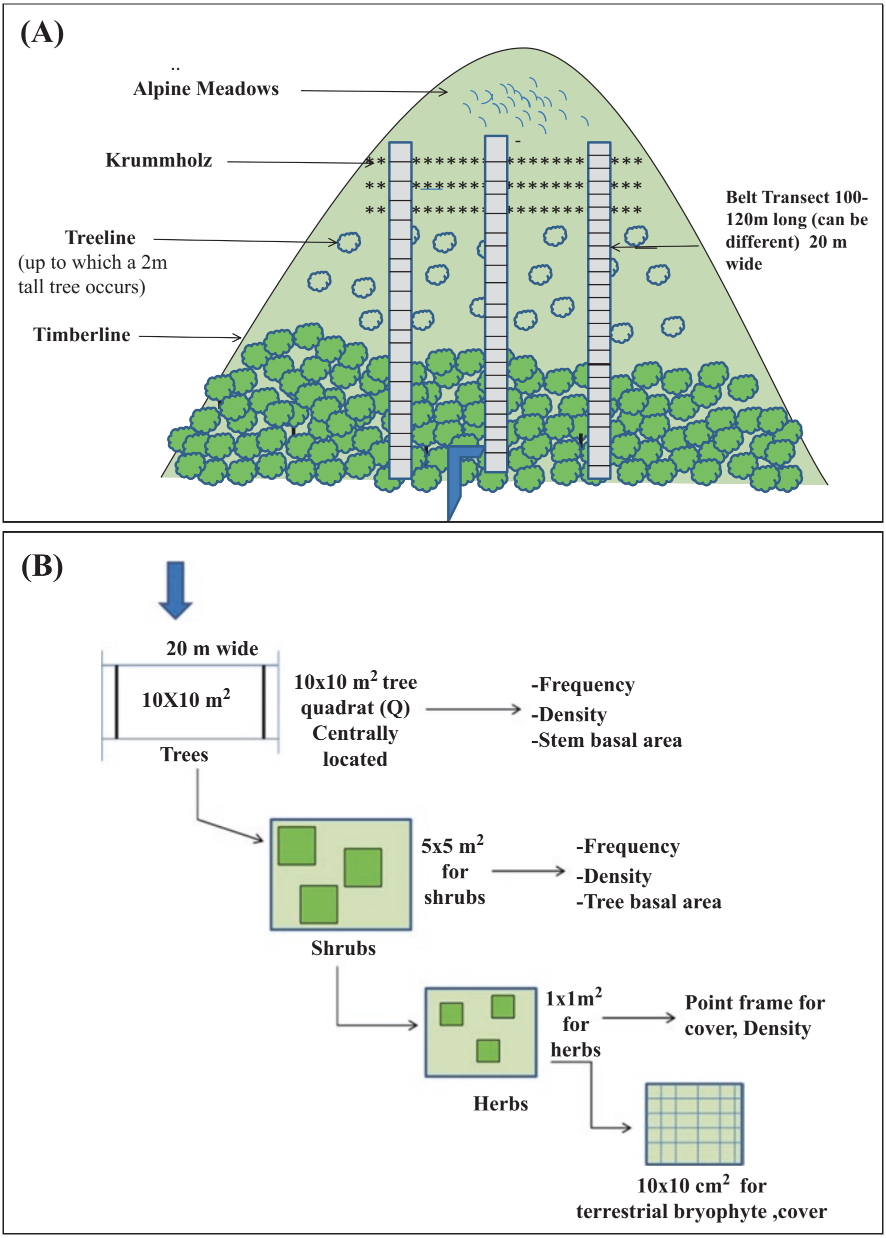

Fig. 10.6 (**a**) Belt transect to show pattern of species population in treeline areas, and (**b**) an outline of sampling for different growth forms in treeline areas to find out the patterns of species populations

This approach is particularly suitable when the purpose of the study also includes finding species with high indicator value for climate change. Temporal changes in species distribution can be analysed to find out their sensitivity to climate change. Being short lived, herbaceous species are likely to be better indicators of climate change than trees.

How are herbs and other species populations distributed along the temperature gradients in treeline areas? In other words, how are they affected by the critical temperature which brings about such a drastic change in physiognomy from tree to herbs?

10.3.2.5 Delineating Boundaries of Timberlines and Treelines

A treeline can be drawn by connecting elevations where the topmost trees occur. By connecting elevations up to which tree crown cover is at least 30% and above which trees become sparse, we can draw a timberline. In the case of an abrupt timberline, trees suddenly give way to a shorter form of vegetation like herbaceous communities. To decide about the lower side of timberline, we can include vegetation up to the point at which it is still similar in species composition to that of upper edge timberline. For this, a cut-off level could be taken, say 90% similarity in species composition. The altitude or distance of the contiguous quadrats up to which similarity in species of all forms with the first contiguous quadrats is 90% can be considered the lower limit of timberline. However, we would need to learn from field experiences to decide about the cut-off point of similarity. Treeline thickness could be used as a treeline community character.

When we accept terms like diffuse and finger-type treelines, we suggest that a treeline is not a line but vegetation occupying an area of land. Thus, the line connecting the highest trees in mountains can be called the upper edge of treeline (Fig. 10.7), while the lower edge of treeline is marked by the upper edge of timberline.

In the Himalayas, generally the ecotone has patches of herbaceous vegetation, small islands of trees, juniper mats and rhododendron krummholz distributed within about 50–150 m elevation range. Such patches of mat and krummholz can also be present outside this area. All these physiognomically different pieces of vegetation/communities together form a kind of complex, which can be referred to as treeline-timberline complex (TTC) (Fig. 10.7).

Advantages of considering treeline as communities and ecosystems are as follows:

1. Population distribution of herbaceous species with shorter life spans may prove to be better indicators of climate change than trees.
2. By clearly delineating boundaries of treeline and timberline, community changes in them can be measured precisely, and data of different sites can be meaningfully compared for species diversity, species composition and ecosystem attributes. Data thus collected could be used to classifying treelines, locally, regionally and globally, using a suitable classification method.
3. It also gives importance to other vegetation types that occur in treeline areas such as juniper mats and treats treeline areas as a landscape.

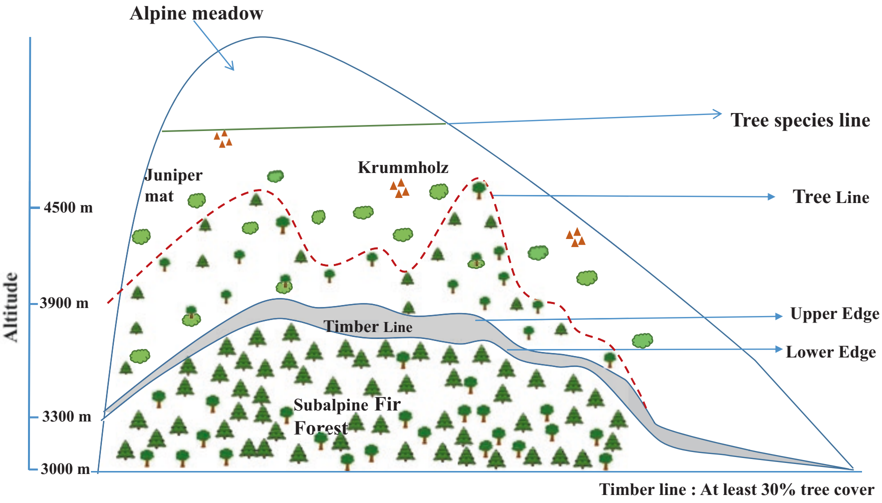

Fig. 10.7 A diagrammatic representation of treeline-timberline complex (TTC). Timberline thickness is determined by how far down the forest interior species composition is similar (90% or more similarity) to the edge of timberline. The upper edge of treeline is determined by a line connecting the uppermost trees (>2 m height). The area marked by the upper edge of treeline, and the upper edge of timberline is called the treeline ecotone, which can have scattered isolated trees, juniper mats and rhododendron (*R. campanulatum*) krummholz. Such patches can also occur above the uppermost trees or uppermost tree species (individuals of tree species, but no more tree-like). Should krummholz determine the upper edge of treeline if it occurs above the highest trees or not is unclear. Here, we consider it as part of TTC. This also applies to juniper mat

10.3.2.6 Distribution of Other Species Than Trees

As referred to earlier, several species other than trees occur at treelines. To what extent herbaceous species respond to climate warming by moving upslope? Though treeline is determined by heat deficiency, it does not represent a discontinuum in temperature. However, the absence of tree cover may result in a sharp increase in light and diurnal temperature range beyond treelines, in alpine meadows. It may be pointed that there are numerous other changes associated with moving away from the treeline: hydrology, microbial communities endo-vs. ectomycorrhizal fungi, quantity and quality of litter, animal activity, etc. Do herbaceous species form sharp boundaries at treeline, particularly when treeline is abrupt? (Fig. 10.8). A well-designed sampling for documenting the distribution of species populations can help answer these questions. We need to examine patterns in species populations of other growth forms too, such as epiphytes, shrubs and others.

Treelines also have mats (junipers) and krummholz (*Rhododendron campanulatum*). In the Kashmir site (Sinthal top), juniper mat seemed to have facilitated the establishment of birch (*Betula utilis*) and fir (*Abies pindrow*) trees (Fig. 10.9) as there were more of them within mats and krummholz than outside.

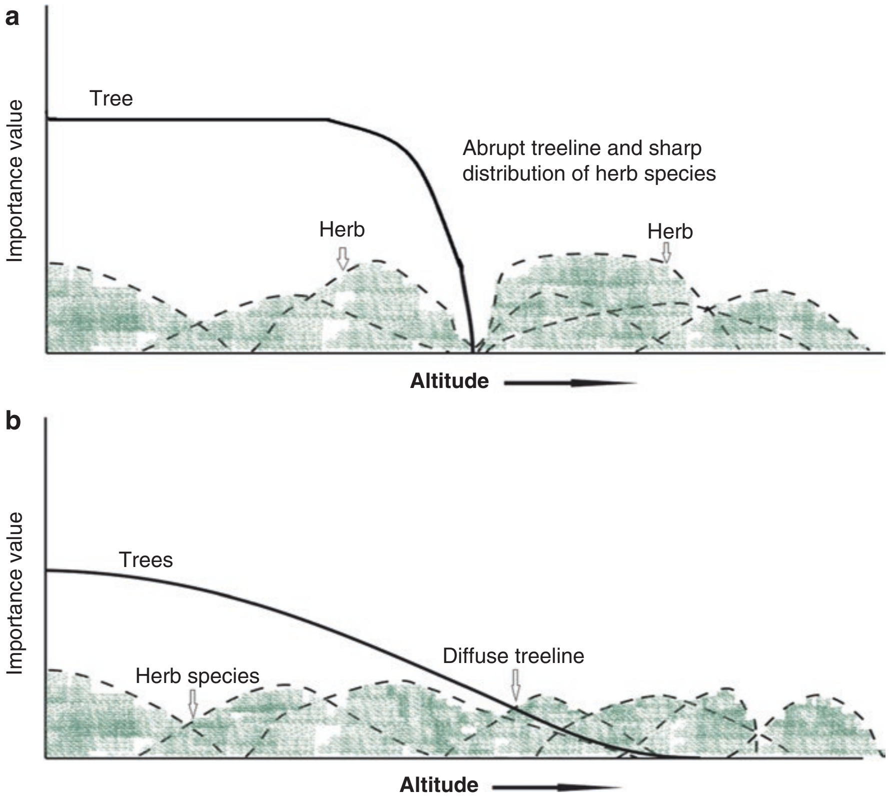

Fig. 10.8 A representation of species population distribution along altitudinal gradient across treeline. In abrupt treeline (**a**), tree population declines sharply. However, we do not know how herb species populations are distributed and respond to climate change. Do herb species populations form sharp boundaries at abrupt treeline because of sudden increase in sunlight or other changes? In diffuse treeline (**b**), tree population declines gradually with increase in altitude. Are herbaceous species populations likely to form continuum in abrupt treeline? We need to examine also whether herbaceous species responses to climate change will also vary between two types of treeline

In many areas south of the main Himalayan ranges, altitudinal transects are covered with woody vegetation up to considerable elevations, leaving limited areas for alpine meadows to occupy dominated by herbaceous vegetation comprising forbs and grasses. These landscapes have a complex mosaic of communities consisting of timberline and treeline communities, juniper mats, krummholz and meadows. Trees and mats are interconnected, the latter facilitating tree establishments, and treelines are often diverse, abrupt, diffuse, island-like and finger-shaped. Depth and residence time of snow which seem to greatly shape landscape features are likely to change under the influence of global climate change. Studies here are required to consider these diverse features to understand ecosystem functioning and services.

Fig. 10.9 Juniper mat and rhododendron krummholz facilitate the establishment of fir and birch trees. (Courtesy: Subrat Sharma, GBPNIHESD, Almora)

10.3.2.7 Changes in Tree Form Through the Treeline Ecotone

Tree form changes towards treeline, but the extent of change varies from one species to another. The pattern of change in the form is being studied as given below:

Seven to ten average-sized trees are identified in each of the following: (i) forest stand adjacent to timberline (just below it), (ii) timberline, (iii and iv) one or two sites between timberline and treeline (TTE) and (v) treeline and, if present, finally (vi) krummholz.

The following measurements are being made: tree height, stem diameter at breast height or a suitable height in case the breast height measurement is difficult (e.g. a krummholz), crown depth and crown breadth (at least in four opposite directions), calculate canopy unevenness ratio using canopy spread in two opposite directions; average of between-tree distances, by measuring distances at least between 8 and 10 tree pairs, height of stem at the point of the first branch, tilt in stems, scars of damages, a rough idea of age of the tree population, etc. A flag tree with no branches and twigs on one side is a good example of extreme unevenness. Consider both within-species variation and variation across species of the gradient.

10.3.3 Ecosystem Approach

In most of the studies over the world, treelines are not studied as ecosystems. While carbon reserve and carbon supply issues have often been debated in relation to treeline formation (Tranquillini 1979; Körner 1998, 1999, 2003), how ecosystem properties change towards treeline has drawn little attention. A few studies (Garkoti and Singh 1992, 1994; Adhikari and Rawat 2012) that have been conducted in high mountain forests indicate a sharp decline in biomass accumulation ratio (calculated as biomass divided by net primary productivity) in high altitude areas towards treelines. In the Pindari area of Uttarakhand, India, biomass declines from birch forest (*Betula utilis* at 3150 m) to rhododendron krummholz (*R. campanulatum* at 3300 m) by 87% in spite of only 50% decline in the net primary productivity. This in combination with winter carbon shortage and resultant injuries in tissues influences the treeline formation (Li et al. 2008).

Since snow and ice are important components of Himalayan treelines, the impact of their early melting on structure and functioning of ecosystem should be investigated. Snow depth affects ecosystem processes, such as runoff initiation date, length of snow-free period, allochthonous deposition of nutrients and plant community composition (Tomback et al. 2016). Many of the Himalayan treelines which lie on the southern side of the main Himalayan ranges are wet and vulnerable to landslips and erosion. Climate change is predicted to result in more days of heavy rainfall. In view of these factors, the regulatory role of forests becomes more important.

In the Himalayas, the interception of rain as the percentage of gross rainfall is generally higher in conifer forests than broadleaved forests (Singh and Singh 1992). In Mount Gongga area of Tibet, interception of the incident rainfall was found to be 75.7% in a conifer forest (dominated by *Abies fabri*) and 45.5% in mixed broadleaved forest having *Populus purdomii*, *Sorbus multijuga*, *Betula utilis* and *Abies fabri* (Liu et al. 2013). On conifer leaves more small droplets are formed than on leaves of broadleaved trees, hence the higher evapotranspiration. At the Tungnath, Uttarakhand at treeline site, we are comparing the hydrology of fir, oak forests and rhododendron krummholz.

Climate change-induced forest advancing into alpine meadows is likely to profoundly affect the ecosystem structure and function in treeline landscapes particularly of invaded ecosystems (Greenwood and Jump 2014). Climate change may drastically modify the mosaics of communities that occur in and around treelines.

Changes in plant species diversity in alpine meadows of the Himalayas due to advancing woody species will affect productivity and stability of communities (Singh et al. 2005). However, tree advancement is expected to stabilize slopes, prevent erosion (Stoffel et al. 2006) and affect nutrient and water cycling (Dirnböck and Grabherr 2000).

While densification and advancement of trees will increase carbon sequestration and storage (Liu et al. 2010), soil carbon will deplete in a warmer condition because of increased soil respiration (Greenwood and Jump 2014).

As for nitrogen availability, it may increase because of increase in decomposition and upslope advancement of trees (Sjogersten and Wookey 2005). How mycorrhizal association of trees is going to respond to altitude and climate change is hardly known, though it may be a key factor in treeline advancement and ecosystem functioning. Because of climate change, it is predicted that both host and ectomycorrhizae will migrate, the relationship between the host and fungus will change, and such changes will affect the carbon cycle (Prickles et al. 2011).

10.4 Conclusion

Because treelines in the Himalayas are high and remote, little investigation has been made about them. The traditional tree population-level studies have contributed significantly to document changes in treeline elevation due to climate change and other anthropogenic factors, such as grazing and burning of trees. However, to understand treeline responses to climate change and manage effectively, community- and ecosystem-level treatments of treelines are also required. To make progress in this direction, we need to develop appropriate methods and study approach, and our paper addresses this need to an extent. Mapping the Himalayan treelines at a regional scale, estimating temperature lapse rate based on observed data, understanding of tree-water relationship during winters and conceptualizing treeline studies at community and ecosystem levels are some of our new efforts.

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References

Adhikari BS, Rawat GS (2012) Structural and functional attributes of forests along an altitudinal gradients in Garhwal Himalaya. Glimpses on Forestry Research in the Indian Himalayan Region, special issue in the International Year of Forests-2011. ENVIS Centre on Himalayan Ecology, Kosi-Katarmal, Almora – 263643, India Chapter 4: 41–53

Borgaonkar HP, Ram S, Sikder AB (2009) Tree-ring analysis of high elevation Cedrus deodara D. Don from Western Himalaya in relation to climate and glacier fluctuations. Dendrochronologia 27:59–69

Borgaonkar HP, Sikder AB, Ram S (2011) High altitude forest sensitivity to the recent warming: a tree-ring analysis of conifers from Western Himalaya, India. Quat Int 236(1–2):158–166

Dirnböck T, Grabherr G (2000) GIS assessment of vegetation and hydrological change in a high mountain catchment of the Northern Limestone Alps. Mt Res Dev 20(2):172–179

EDW Working Group (2015) Elevation-dependent warming in mountains of the world. Nat Clim Chang 5:424–430

Gaire NP, Dhakal YR, Lekhak HC, Bhuju DR, Shah SK (2011) Dynamics of *Abies spectabilis* in relation to climate change at the treeline ecotone in Langtang National Park. Nepal J Sci Technol 12:220–229

Gaire NP, Koirala M, Bhuju DR, Borgaonkar HP (2014) Treeline dynamics with climate change at the central Nepal Himalaya. Clim Past 10(4):1277–1290

Ganjurjav H, Gao Q, Zhang W, Lang Y, Li Y, Cao X, Wan Y, Li Y, Danjiu L (2015) Effects of warming on CO2 fluxes in an alpine meadow ecosystem on the Central Qinghai-Tibetan Plateau. PLoS One 10(7):e0132044

Garkoti SC, Singh SP (1992) Biomass productivity and nutrient cycling in alpine Rhododendron community of Central Himalaya. Oecologica Mountana 2:21–32

Garkoti SC, Singh SP (1994) Nutrient cycling in three Central Himalayan forests ranging from close-canopied to open-canopied tree-line forests. Arct Alp Res 26:339–348

Ge O, Wang H, Rutishauser T, Dai J (2015) Phenological responses to climate change: a met-analysis. Glob Chang Biol 21:265–274

Greenwood S, Jump A (2014) Consequences of treeline shifts for the diversity and function of high altitude ecosystems. Arct Antarct Alp Res 46(4):829–840

Harsch MA, Hulme PE, McGlone MS, Dunca RP (2009) Are treelines advancing? A global meta-analysis of treeline response to climate warming. Ecol Lett 12:1040–1049

Holtmeier FK (2009) Mountain timberlines. Ecology, patchiness, and dynamics. Springer, Berlin

Körner C (1998) A re-assessment of high elevation treeline positions and their explanation. Oecologia 115:445–459

Körner C (1999) Alpine plant life. Functional plant ecology of high mountain ecosystems. Springer, Berlin/Heidelberg

Körner C (2003) Carbon limitation in trees. J Ecol 91:4–17

Körner C (2012) Treelines will be understood once the functional difference between a tree and a shrub is. Ambio 41(3):197–206

Körner C, Paulsen J (2004) A world-wide study of high altitude treeline temperatures. J Biogeogr 31:713–732

Li M, Xiao W, Shi P, Wang S, Zhong Y, Liu X, Wang XD, Cai X, Shi Z (2008) Nitrogen and carbon source–sink relationships in trees at the Himalayan treelines compared with lower elevations. Plant Cell Environ 31:1377–1387

Liang E, Wang Y, Eckstein D, Luo T (2011) Little change in the fir tree-line position on the southeastern Tibetan Plateau after 200 years of warming. New Phytol 190:760–769

Liu Y, Stanturf J, Goodrick S (2010) Trends in global wild-fire potential in a changing climate. For Ecol Manag 259:685–697

Liu G, Du S, Peng S, Wang G (2013) Rainfall interception in two contrasting forest types in the Mount Gongga area of Eastern Tibet, China. Hydrol Current Res 4:161. https://doi.org/10.4172/2157-7587.1000161

Miehe G, Miehe S, Vogel J, Co S, La D (2007) Highest treeline in the northern hemisphere found in southern Tibet. Mt Res Dev 27:169–173

Nüsser M, Dickoré WB (2002) A tangle in the triangle: vegetation map of the eastern Hindukush (Chitral, northern Pakistan). Erdkunde 56:37–59

Prickles BJ, Egger KN, Massicotte HB, Green DS (2011) Ectomycorrhizas and climate change. Fungal Ecol 5(1):73–84. https://doi.org/10.1016/j.funeco.2011.08.009

Schickhoff U (2005) The upper timberline in the Himalayas, Hindu Kush and Karakoram: a review of geographical and ecological aspects. In: Broll G, Keplin B (eds) Mountain ecosystems. Studies in treeline ecology. Springer, Berlin/Heidelberg/New York, pp 275–354

Schickhoff U, Bobrowski M, Böhner J, Bürzle B, Chaudhary RP, Gerlitz L, Heyken H, Lange J, Müller M, Scholten T, Schwab N, Wedegärtner R (2015) Do Himalayan treelines respond to recent climate change? An evaluation of sensitivity indicators. Earth Syst Dynam 6:245–265

Shrestha UB, Gautam S, Bawa KS (2012) Widespread climate change in the Himalayas and associated changes in local ecosystems. PLoS One 7(5):e36741. https://doi.org/10.1371/journal.pone.0036741

Singh SP, Rawal RS (2017) Manuals of field methods. Central Himalayan Environment Association, Nainital

Singh JS, Singh SP (1992) Forests of Himalaya. Gyanodaya Prakashan, Nainital

Singh J, Yadav RR (2000) Tree-ring indications of recent glacier fluctuations in Gangotri, Western Himalaya, India. Curr Sci 79(11):1598–1601

Singh SP, Tyagi V, Sah P, Jina B (2005) Species diversity contributes to productivity – evidence from natural plant communities. Curr Sci 89:548–552

Sjögersten S, Wookey PA (2005) The role of soil organic matter quality and physical environment for nitrogen mineralization at the forest-tundra ecotone in Fennoscandia. Arct Antarct Alp Res 37:118–126

Speed JDM, Martinsen V, Hester AJ, Holand Ǿ, Mulder J, Mysterud A, Austrheim G (2015) Continuous and discontinuous variation in ecosystem carbon stocks with elevation across a treeline ecotone. Biogeosciences 12:1615–1627

Stoffel M, Wehrli A, Kühne R, Dorren LKA, Perret S, Kienholz H (2006) Assessing the protective effect of mountain forests against rockfall using a 3D simulation model. For Ecol Manag 225(1–3):113–122

Tomback DF, Resler LM, Keane RE, Pansing ER, Andrade AJ, Wagner AC (2016) Community structure, biodiversity, and ecosystem services in treeline whitebark pine communities: potential impacts from a non-native pathogen. Forests 7:21. https://doi.org/10.3390/f7010021

Tranquillini W (1979) Physiological ecology of the alpine timberline- tree existence at high altitude with special reference to the European Alps, Ecological studies 31. Springer, Berlin/Heidelberg

Yao T, Thompson L, Yang W, Yu W, Gao Y, Guo X, Yang X, Duan K, Zhao H, Xu B, Pu J, Lu A, Xiang Y, Kattel DB, Joswiak D (2012) Different glacier status with atmospheric circulations in Tibetan Plateau and surroundings. Nat Clim Chang 2:663–667. https://doi.org/10.1038/NCLIMATE1580

Community Response and Adaptation to Climate Change in Central Himalaya, Uttarakhand, India

11

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Abstract

Mountains across the globe are unique areas for assessing climate change impacts on biodiversity, glaciers and fresh water supply, food production and overall human well-being. The livelihood and food security of the people inhabiting the mountainous parts of the state of Uttarakhand falling in the central Himalayan region of India largely depend on closely interlinked and climate-sensitive sectors, viz. agriculture, livestock and forestry. Understanding peoples' perceptions of climate change impacts could help designing and implementation of adaptation strategies at the local level. Traditional knowledge of the farming communities for sustainable use of natural resources is gaining credence in recent times. Here we present an analysis of farmers' perceptions and traditional knowledge in adapting to climate change impacts on agroecosystems in central Himalaya and suggest mechanism for integration of these in developing appropriate locally nuanced adaptation strategies.

Keywords

Community · Adaptation strategies · Sustainability · Bioresources · Himalaya

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11.1 Introduction

The Intergovernmental Panel on Climate Change (IPCC 2001, 2007) reports unequivocally state that global climate change is real and will present multiple challenges to sustainable management of local ecosystems (Houet et al. 2010). In the wake of the projected climate change challenges, among the variables, conservation of natural resources, agricultural and livestock production systems, food security, societal health and stability are of particular interest (Schilling et al. 2012). Worldwide agriculture supports the livelihoods of the largest number of people worldwide and is vital to rural development and poverty alleviation. Himalayan ecosystem is distinguished globally as one of the biodiversity hotspots and for its role in regulating the climate of the entire South Asia (Myers et al. 2000; Saxena et al. 2005). Though agriculture is minor land use in terms of spatial extent, the subsistence economy of the Himalayan villages revolves around it and animal husbandry which are in turn directly dependent on forestry sector for resources (Maikhuri and Ramakrishnan 1990; Maikhuri et al. 2010, 2013). Climate change directly affects agricultural systems in terms of influencing crop yields and choice of crops that can be grown or indirectly by affecting the forest resources and ecosystem services on which agriculture depends. Central Himalaya which includes the Indian state of Uttarakhand has a long heritage of subsistence economy, with mixed (crop and animal husbandry) farming being the core component involving over 70% of its population (Maikhuri et al. 1997, 2001; Semwal et al. 2004; Maikhuri and Rawat 2013). Traditional farmers maintain a high level of crop diversity in this part of the Himalaya. Across agro-climatic zones, over 40 different species of crops comprising cereals, pseudo-cereals, millets, pulses, oilseeds, tubers and condiments and hundreds of their local cultivars are cultivated by the local farmers in their marginal and scattered land holdings through a variety of crop composition and rotations (Maikhuri et al. 1996, 1997; Rao and Saxena 1994; Nautiyal et al. 2002 Negi and Maikhuri 2012). However, during the recent past, due to changes mainly in socio-economic spheres and land use, the agrobiodiversity has been declining in traditional agricultural systems of the region (Ravera et al. 2016).

Changes in climate may further accentuate the rate of erosion of agrobiodiversity and also greatly affect the quality as well as quantity of yields of traditional farming systems in the region (Maikhuri et al. 2009; Rao et al. 2012). People inhabiting the mountainous parts of Uttarakhand (constituting about 88% of the total geographical area of the state in the central Himalayan region) need to adapt and develop coping strategies in multiple sectors of economy especially in agriculture (Maikhuri et al. 2003; Saxena et al. 2004, 2005). Though local knowledge regarding climate change is often weather-related knowledge rather than fully evolved knowledge system on climate change per se, historically, the local farmers were able to adapt to changes in weather with different amounts of success (Saxena and Rao 2009). Fortunately, with the renewed global interest in traditional agricultural systems, a positive trend in this direction has already set in where many of its features such as maintaining high level of genetic diversity, mixed cropping, organic inputs and weed management have already found scientific support from sustainability standpoint (Pradhan et al. 2012). For example, high genetic diversity in traditional agriculture not only helps protect crops from disease and pest attack but often ensures higher yield including enhancing its climate resilience (Zhu et al. 2000). However, because of poor communication through formal institutional arrangements, the lessons learned and many of the recent innovations are not getting disseminated among the farmers to the desired extent. The situation demands a better understanding of the farmers' perceptions on climate change impacts and their adaptation measures. Here we present an analysis of such perceptions and adaptation measures of the farmers inhabiting the mountainous parts of Uttarakhand state falling in the Indian central Himalaya.

11.2 Study Area

The Indian state of Uttarakhand is located in the central Himalayan region covering an area of 53,483 km^2. The region has contrasting climatic zones along the altitudinal gradient from sub-tropical in lower altitude in the south to the alpine and arctic type in the high altitude in the extreme north. The total population of the state is 10.1 million (Census 2011) with a sex ratio of 963. Although the net cultivated area is only 14.07%, agriculture remains to be the major occupation of the local people of the state. The land holding size in the state can be classified as marginal since 49% of holdings are less than 0.5 ha and 21.51% are between 0.5 and 1 ha. Thus, over 70% holdings are marginal in nature with an average size of about 0.37 ha/household. The small and scattered land holdings coupled with rugged terrain pose challenge to economic viablity of traditional agriculture in present times.

The present study was conducted in three different agro-climatic zones, i.e. lower (300–1000 m asl), middle (1100–1800 m asl) and high (1900–3000 m asl), in Uttarakhand. Climate varies from sub-tropical in the lower-altitude valleys to temperate on the higher slopes with summer monsoon contributing over 70% of the total precipitation. The higher Himalayan ranges and glaciers form most of the northern parts of the state while the lower reaches are forested providing habitat to a range of wildlife and flagship species. Two of India's mightiest and sacred rivers, the Ganga and the Yamuna originate in the glaciers of Uttarakhand and provide water to nearly half of the population of the country (Fig. 11.1).

11.3 Methodological Approaches

11.3.1 Information on Climate Change Assessment

Qualitative and quantitative methodologies were followed to document the peoples' perceptions and traditional ecological knowledge related to changes in weather related parameters i.e. temperature and precipitations. Local perceptions and knowledge systems differ across the study sites under diverse climatic conditions and environmental resources. Information on the key sectors of the rural economy viz.

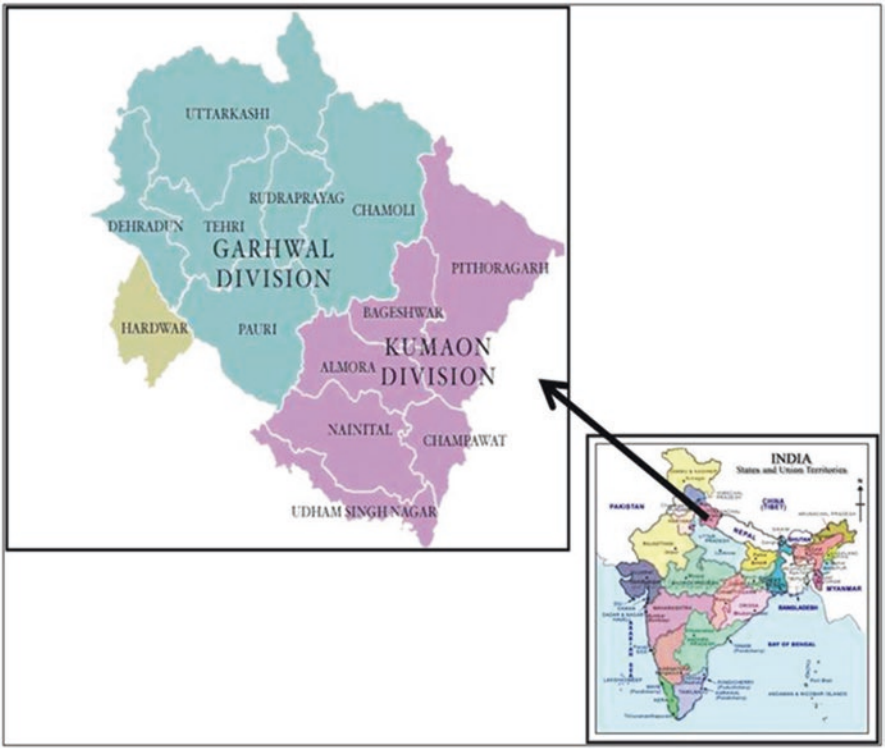

Fig. 11.1 Location map of the study area in central Himalaya, Uttarakhand, India.

agriculture, animal husbandry and forest resources, was collected through household surveys, focus group discussions and other PRA tools such as community mapping, seasonal calendars and historic information. In addition, relevant data were also secured from available literature and other secondary sources.

11.3.2 Sampling Approach

A total of 54 villages, six villages in each of the nine mountainous districts, viz. Rudraprayag, Chamoli, Tehri Garhwal, Uttarkashi, Pauri Garhwal, Almora, Bageshwar, Champawat and Pithoragarh were selected for the study. Twenty households in a village with a total of 1080 households across the identified villages were selected randomly for documenting peoples' perceptions and understanding their adaptation actions to minimize impacts of climate change induced weather variability on local ecosystems and livelihoods. Both knowledgeable men and women were involved in group discussions and personal interviews in order to collect information on climate variability and its impact on natural resources, agroecosystems, livestock husbandry and their adaptation strategies at different time intervals. Data on perceptions, responses and local knowledge of climatic variability were collected through individual interviews with local elderly and knowledgeable persons based on questionnaires and a checklist of location specific issues adopted according to interview situations. The respondents were asked 'whether they have perceived, detected or witnessed the given climate change-related indicator' and requested to choose one of the following three options: 'Yes, have experienced', 'No, haven't experienced' and 'Don't know about it' (Negi et al. 2017).

11.3.3 Qualitative Appraisal

Information on local perceptions was recorded on different aspects such as climate induced weather variability, natural resources and water availability, status of plant diversity, agriculture and livestock productivity, problems related to pest and diseases in crops, livestocks and human health and over all implications for the sustainability of mountain agroecosystems. A detailed rural appraisal survey was undertaken in selected villages between 2014 and 2016 in order to collect the information on climate change impacts on the aforementioned aspects over the last three decades in the region. Group discussions and meetings were organized at village level to validate the general perceptions regarding climate change impacts and verified through field surveys and personal interviews.

11.3.4 Quantitative Assessments

People's perceptions and knowledge on climate change derived not from any direct measurements of variation in weather parameters but from the way they affect their immediate surroundings and livelihood (Saxena et al. 2004). For example, in agricultural systems people pointed out moisture stress in crops as the condition stemming from delayed onset and/or lower amounts of monsoon rainfall, nutrient deficiency due to lower rates of farmyard manure inputs because of lesser availability of leaf litter on forest floor used for making it as a result of frequent forest fire over the last several years as well as impact of climate variability in traditional agriculture system. The questionnaire survey was used to collect household data concerning farmers' perceptions and awareness towards climate variability and its impacts, types and preparedness and adaptation measures adopted by people were further rationalized and verified with the data generated through the qualitative methods. Data on include natural resource availability, animal husbandry practices and changes in crops grown over a period of the last three decades and reasons behind the changes and indicators of climate change were also collected following questionnaire based approach. The survey was conducted in local language for better communication.

11.4 Results and Discussion

11.4.1 Community Perceptions on Climate Change/Weather Variability

Data gathered through the questionnaire survey showed that more than 80% of the farmers of the study area were aware of changes occurring in weather parameters (temperature and precipitation) over the past three decades. At local level, community perceptions and knowledge on the weather parameters were intimately associated to comparisons between the past and present patterns of their local indicators. Any divergence observed in such patterns was expressed as variability in normal weather or change in climate. For example, for people inhabiting the middle and higher agro-climatic zones of the central Himalaya, a normal or benign climate meant sporadic low rainfall during March–May with mild temperature, peak rainfall during July–August and absence of cloudbursts and moderate rainfall/heavy snowfall during December/January with low temperature and without hailstorm events (Saxena et al. 2004). Majority of respondents concurred that during recent decades, there had been many changes in the climate and cited multiple examples (Table 11.1). People also highlighted that now there is a higher incidence of dry spells particularly post monsoon resulting in water scarcity and low agriculture productivity. The most significant piece of evidence was decreased in water resources used by livestock, particularly in alpine pastures, forests and grazing areas over a period of 15–20 years as experienced by the pastoralist communities of high-altitude villages in Chamoli and Pithoragarh districts. People of all the villages across agro-climatic zones indicated that low rainfall or shift in rainfall resulted in crop failure; reduced the yield of food grains, fodder resources, horticultural crops and livestock production; and has weakened the traditional livelihood and economic earning avenues of the local households. Further, they also indicated that the frequency of disease and insect/pest attack on crops has increased manyfolds particularly in agri-horticultural crops (*Amaranthus* spp., *Phaseolus vulgaries*, citrus fruits, bananas, apples, etc.). Traditional legumenous crops such as *Vigna unguiculata*, *Vigna angularis* grown between 1000 and 2000 m asl are facing problems of fruit setting because of the shift in the patterns of peak rainfall in recent years (Negi et al. 2017). The knowledge holders reported early flowering, leafing and fruiting (20–25 days before the timing of their normal phenophases) in medicinal plants (*Allium stracheyi*, *Berginia ligulata*) and wild edibles (*Rhododendron arboreum*, *Prunus cerasoides*, *Bombax cieba* and *Bauhinia variegata*) (Fig. 11.2).

11.4.2 Community Perceptions on Climate Change Impacts

People experienced and admitted that over the last several years, there have been irregular rainfall (uncertainty of monsoon), drastic changes in the frequency and volume of precipitation patterns, low snowfall and rising temperatures resulting in low yield of agri-horticultural crops and vegetables in the region. In the study

Table 11.1 Climate change as perceived by local farmers, their responses and adaptation strategies in central Himalaya, India

Climate change drivers	Perceived changes	Response/adaptation measures
Higher temperature with low rainfall/precipitation	Vertical species migration of and extinction of some crop varieties increased	Replacement of some crops with vegetables like cauliflower, tomato and cabbage in middle altitudes
	Land use change that increases soil degradation/soil fertility and high species mortality	Using higher amount of seeds during sowing resulted in increasing density of plants to cope with drought condition
	Less infiltration affecting groundwater recharge and drying up of natural springs and streams	Cultivation of vegetables like pea, cauliflower and cabbage under kitchen gardens at higher altitudes
	Decline in the moisture retention and water holding capacity of the soils has been linked to resource degradation, abandonment of land resulting in loss of agro-biodiversity	Adoption of water thripty requiring crops to prevent crop failure due to drought
	Fruits like papaya, banana, mango and litchi shifted to higher-altitude areas	Encouraging crop-livestock integration to increase soil organic matter, thereby increasing water retention capacity of soil for longer times
	Decline in yield of apple and citrus spp. as it needs adequate chilling during winters for proper fruit yield	Adoption of alternate crop cultivation, viz. *Zingiber officinale* and *Curcuma longa*, floriculture (*Gladiolus* spp. and *Lilium* spp.) and fodder crops (*Pennisetum purpureum*, *Thysanolaena maxima*, etc.) as option of livelihood
	Low land legumes shifted to higher altitude due to decreased yields	Conversion of irrigated to rainfed farming due to reduced flow of water in the streams as a result of warming and conversion of rainfed land at middle altitude to irrigated if water is available
	Decline in the yield (80%) of *Perilla frutescens*, one of the important oil-yielding crops	
	Increased grain yield of *Eleusine coracana*, *Amaranthus* spp. and *Sesamum indicum* between 1500 and 2200 masl	
	Decline in the grain yield of cash crops like *Phaseolus vulgaris* and *Solanum tuberosum*	
	Farmers in these districts also indicated that inadequate draught power also inhibits their capacity to maximize the crop yields and their ability to prepare larger pieces of land	

(continued)

Table 11.1 (continued)

Climate change drivers	Perceived changes	Response/adaptation measures
Higher temperature and precipitation	High species mortality, loss of species and reduced agro-biodiversity at lower and middle altitude	Human migration due to extreme hydrological events and seasonal displacement
	Unpredictable rain with impacts on harvesting and cropping patterns	Cultivation of some medicinal plants, viz. *Picrorhiza kurrooa*, *Arnebia benthamii*, *Saussurea costus*, *Allium stracheyi*, *Allium humile*, *Angelica glauca*, *Carum carvi*, etc., at high-altitude villages
	Increase in population of invasive, exotic weeds (e.g. *Lantana camara*, *Eupatorium* spp. and *Parthenium* spp.)	
Decline in rainfall during March–May	Adverse impact in terms of decline in yield of Kharif crops due to large scale mortality and/or poor growth in the initial stage of crop growth	Change in cropping pattern, for example, cultivation of *Phaseolus vulgaris* is replaced with *Macrotyloma uniflorum*; *Vigna unguiculata* with *Cajanus cajan*; *and Phaseolus vulgaris* with *Glycine max* in the villages of middle altitude (1000–1800 m asl)
	Abandonment of crop, e.g. *Panicum miliaceum*, which used to be sown in March. This crop matures over a period of 3 months. The crop is badly affected if rainfall is delayed	Replacement of *Amaranthus paniculatus* by cauliflower, cabbage and potato
High rainfall during August and September instead of the normal peak in July/August	Damage to rainy season crop when they are close to maturity	Adoption of alternate crop cultivation, viz. *Zingiber officinale* and *Curcuma longa*, floriculture (*Gladiolus* spp. and *Lilium* spp.) and fodder crops (*Pennisetum purpureum, Thysanolaena maxima*, etc.) as option of livelihood
	Shattering of crops before harvesting	
	Decline in the grain yield due to premature and post harvesting	
	Decline in the biomass of crop by-products negatively impacts livestock production system	
Winter precipitation in January and February instead of December and January and decline in intensity of snowfall	Delayed ploughing/sowing of wheat, barley and mustard resulted in decreased yield as earlier it is used to be done in November but now in December	Replacement of traditional cultivars of wheat by high- yielding shorter duration varieties
	Decline in grain yield of barley, brassica and wheat	Recycling of weed into soil for manuring and retaining moisture
	High-yielding varieties of wheat and green pea which did not perform well earlier can perform well at higher altitude	Farmers growing higher fodder-yielding varieties with low grain production

(continued)

Table 11.1 (continued)

Climate change drivers	Perceived changes	Response/adaptation measures
	Declining fodder availability which indirectly affects farmyard manure (FYM) production, one of the essential components of hill agriculture. Climatic changes alter the pattern of blossoming, bearing and, therefore, fruit yield. The lack of early cold in December and January is understood to adversely affect the chilling requirements in horticultural crops	Replacement of barely by green pea in middle and high altitudes
		Legumes like *Macrotyloma uniflorum* and *Vigna angularis* shifted to higher altitude
		Shifted towards protected cultivation (polyhouse, shade-net, polypit, etc.) for seasonal and off-seasonal vegetable cultivation
Climate change (major changes felt by farmers)	Adjusted agricultural production systems to production environment (short, early-maturing crops, short-duration, resistant and tolerant varieties, appropriate sowing)	Innovation of new agricultural practices characterized by high labour productivity and stress-tolerant cash crops
	Early flowering, fruiting and maturity of winter crops, particularly wheat and mustard	Replacement of traditional staple crops by cash crops like potato, kidney bean and pea
	Increased use of chemical fertilizers in farming particularly for cash crops	Shifted towards vegetables cultivation in high-altitude villages like cabbage, cauliflower, tomato and hybrid variety of other vegetables which were never gown earlier
	Change in phenological characters and decline in yield of many horticulture fruits (*Prunus persica*, *P. cerasifera*, *P. armeniaca*, *Pyrus pyrifolia*, *Citrus* spp., etc.)	A significant proportion of traditional agricultural land has been brought under off-season vegetables
	Fruits like papaya, banana, mango and litchi shifted to middle-altitude areas, disappearance/low availability of apricot, one of the very common fruit of middle altitude	Increasing trend to lease the agriculture land on rent to Nepali families for cultivation of seasonal and off-seasonal vegetables
	Increased in frequency of human and animal disease	

(continued)

Table 11.1 (continued)

Climate change drivers	Perceived changes	Response/adaptation measures
	Early flowering, leafing and fruiting (20–45 days before the timing of their pheno phases before 20–30 years) in many species including medicinal and aromatic plants (i.e. *Rhododendron arboreum*, *Prunus cerasoides*, *Allium stracheyi*, *Berginia ligulata*, *Betula utilis*)	
	Decline in fodder production resulted in decrease in livestock population, which negatively impacts livelihood and agriculture systems	
	Decline in average quantity of water in streams and drying up of water springs	
	Rural-to-urban and male out-migration and labour shortages for agriculture	

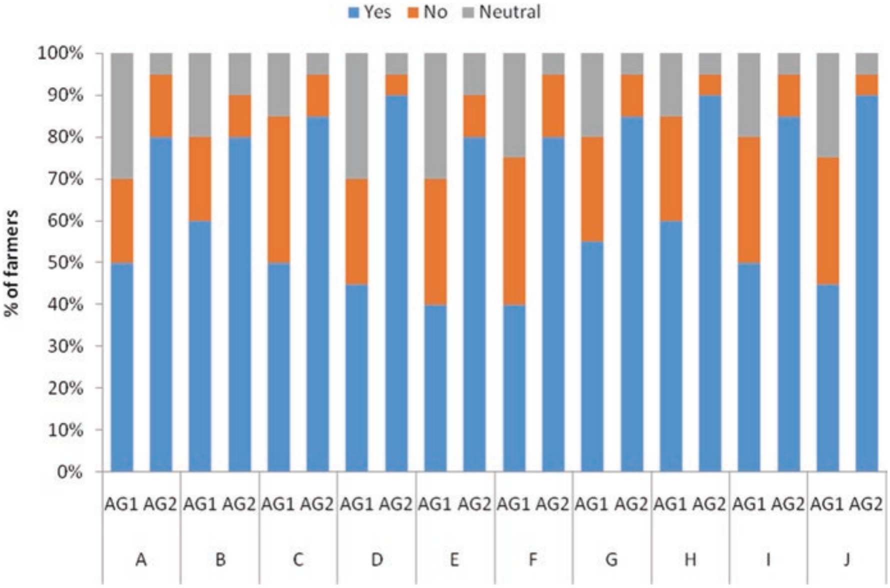

Fig. 11.2 People perceptions on indicators/driving factors of climate changes in central Himalaya **A**. Has the climate changed? **B**. Increase in temperature. **C.** Decrease in snowfall. **D.** Decrease in rainfall/shift in rainfall. **E**. Shift in crop maturation. **G.** Increase in pest/disease in crops. **H**. Change in agricultural land use. **I.** Decrease in water availability in upland. **J**. Change in agrobiodiversity; AG (age group of respondents) AG 1 = 20–50 years; AG 2 = 50–80 and above

villages, low production was reported in horticulture crops such as citrus, banana and apples due to change in the pattern of rainfall/low snowfall and rising temperature and has affected land-based income-generating avenues particularly in Chamoli, Uttarkashi and Almora districts of the state. Agriculture yield and income declined due to uncertainty of weather conditions such as low rainfall during sowing and high rainfall and hailstorms during crop maturity period. Due to low rainfall and consequently longer dry spells during the early summer months of April and May, not only forest resources particularly collection of herbaceous fodder has declined but also increased the frequency and intensity of forest fires. The transhumant pastoralists mentioned that the thickness/volume of snow in many alpine meadows has decreased significantly in recent years leading to decline in the growth of many fodder species. In the past (before 20–25 years), they used to graze their livestock for several days in an individual pasture; now, they keep moving their flocks to multiple pastures. The pastoralists also reported scarcity of water sources such as streams and seepages along the livestock migratory routes and locations of overnight camps during recent years. Farming communities of the study villages stated that earlier there were plenty of perennial sources of water in the villages which either have now been fully dried up or become seasonal due to low rainfall.

11.4.3 Community Adaptation to Climate Change Impacts

People altered traditional cropping calendars across agro-climatic zones and based on their experience and indigenous knowledge have been able to identify some crops better adapted to withstand the variability in local weather conditions (Table 11.1). For example, traditional farming practices of maintaining fragmented landholdings and mixed cropping help farmers minimizing the risk of absolute crop failure during extreme weather events. Secondly, in hailstorm-prone areas and areas experiencing heavy snowfall in higher agro-climatic zone, they selected crop varieties which were less affected under such conditions; and thirdly, the traditional practice of applying farmyard manure (produced by using forest leaf litter as bedding material in livestock sheds and composted litter-livestock excreta mixture converted as manure) in agricultural fields before sowing crops every season helped not only in replenishing soil fertility but also in conserving soil moisture. In addition, people maintained bunds in their gently outward sloping agricultural fields before each cropping season to conserve rainwater and plant local grass, millet and pulses having strong root system to bind the soil tightly from erosion at middle and lower altitudes. However, in present times adoption of cash crop in response to climate variability in middle altitude resulted in decline in the area under traditional crops such as *Perilla frutescens*, *Setaria italica*, *Panicum miliaceum*, *Pisum arvense* and *Hibiscus cannabinus* or, in some locations, complete abandonment of their cultivation. Many among local farmers particularly in higher ago-climatic zone have switched over to crops that require less water and provide cash income such as *Phaseolus vulgaris* (kidney beans) and *Solanum tuberosum* (potatoes). In order to overcome water deficiency and achieve better germination, farmers have adopted sowing overnight presoaked seeds of paddy for nursery development. Sometimes to cope with extreme weather events such as high rainfall and hailstorms during harvesting season, farmers of higher agro-climatic zones harvest their crops before full maturity in order to minimize absolute crop loss during such events. Again in higher agro-climatic zone, taking benefit of climate variability/shift in rainfall, a number of households initiated cultivation of medicinal plants such as *Arnebia benthamii*, *Angelica glauca*, *Saussurea costus*, *Picrorhiza kurrooa*, *Podophyllum hexandrum*, *Allium stracheyi* and *Pleurospermum angelicoides* for income generation and livelihood enhancement. Slight increase in temperature in high-altitude villages created favourable conditions for increased productivity and aboveground biomass of these medicinal plants. The changes in cropping pattern have been noticed in many villages of mid agro-climatic zone as well (Figs. 11.3, 11.4 and 11.5).

11.4.4 Adaptation to Climate Change and Climate Risk Management

The climate change induced weather variability is enhancing the vulnerability of local livelihoods in Uttarakhand. Many of the livelihood-earning sectors are exposed to increasing frequency and intensity of extreme events of hydro-meteorological nature and the long-term impacts of the changes in weather parameters may lead to degradation of agricultural lands into waste/barren land due to extended drought, irregular, less and no rainfall. Understanding the interrelations between changes in the climatic system and socio-economic development is therefore vital for

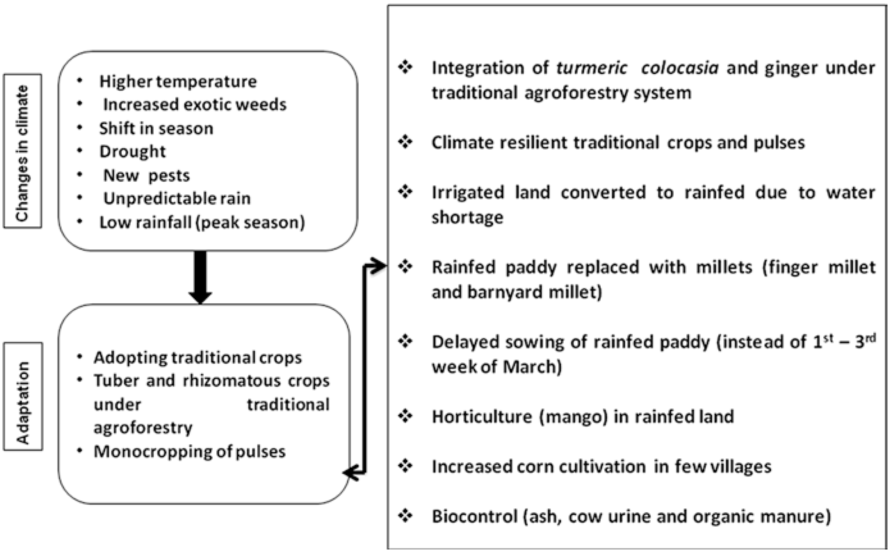

Fig. 11.3 Climate change perceptions and response/adaptation using traditional knowledge and innovations in lower agro-climatic zone (300–1000 m asl) in central Himalaya

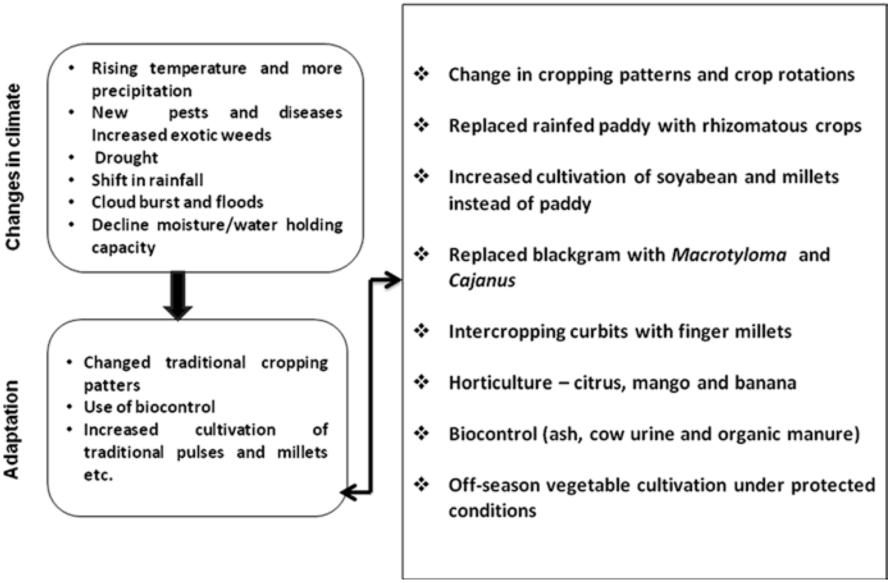

Fig. 11.4 Climate change perceptions and response/adaptation using traditional knowledge and innovations in middle agro-climatic zone (1100–1700 m asl) in central Himalaya

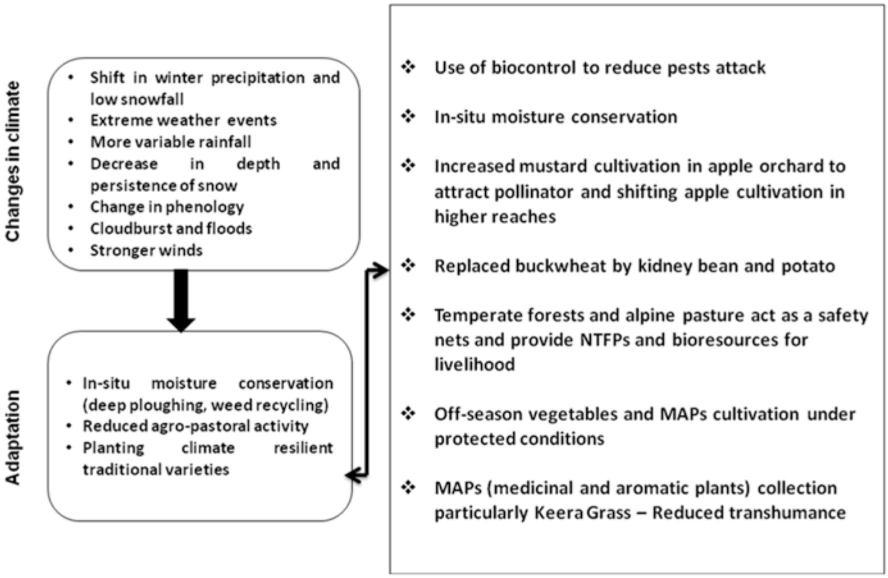

Fig. 11.5 Climate change perceptions and response/adaptation using traditional knowledge and innovations in higher agro-climatic zone (1800–3000 m asl) in central Himalaya

strengthening adaptation capacity at the local level. As with many other natural phenomena, adaptation to changing climate has so far been largely reactive. However, in order to put in place cost-effective adaptation measures, anticipatory measures and actions aimed at reducing vulnerability and increasing resilience need to be initiated considering the future climate scenarios. The Himalayan environment and economies are vulnerable specially the marginal societies and rural poor, whose subsistence largely depends on the climate sensitive natural resources. In this regard, organic cultivation and diversification of agriculture, improvement of soil fertility through bio- and vermi-composting, application of appropriate technologies for yield improvement of off-season vegetables, bio-prospecting of wild edibles, strengthening traditional agroforestry and rehabilitation of degraded lands with suitable multipurpose trees are some of the measures that have been found suitable to respond and adapt to the challenges of climate change in Uttarakhand (Maikhuri et al. 1994, 1997, 2015; Semwal et al. 2013).

11.4.5 Promotion to Cultivation of Traditional Crops for Climate Proofing

The traditional crops are an integral part of the subsistence agricultural systems adapted to small fields, organic inputs, mixed farming, diet and way of life of the local farmers of Uttarakhand. Such crops are known by different names such as underexploited crops, crops for marginal lands, poor person's crops and neglected mountain crops. More recently they have also been called life support crops, the lost crops and organic crops. These are staple crops that farmers are very familiar with, offering several ideal qualities as crops for nutraceutical security in the fragile Himalayan environment. In Uttarakhand along elevational gradient (500– > 2500 m amsl), over 40 different crops comprising cereals, millets, pseudo-cereals, pulses, oil seeds, tubers and condiments and their hundreds of locally selected cultivars/landraces are cultivated by local farmers. The rich biodiversity is maintained through a variety of traditional agronomic practices like mixed cropping and crop rotations. Harvests of certain traditional crops can, if necessary, be made within the cropping season in 50–60 days, such as *Panicum miliaceum*, *Setaria italica* and *Fagopyrum* spp., while for majority of them, a minimum period of 120 days is required. Varied edaphic, topographic and climatic factors as well as different selection pressures over centuries of cultivation resulted in immense variation in traditional crops (Maikhuri et al. 2001). These crops withstood the rigours of time, escaped attacks from insect pests and evolved with local climatic conditions (Table 11.2). They possess the desired agronomic and genetic traits from which high-yielding and stress-tolerant varieties could be developed (Altieri and Nicholls 2004). Notwithstanding the virtues of traditional crops including their nutraceutical value, the precious genetic diversity, is gradually being lost due to a variety of factors related to environment, policy and institutions that are not necessarily supportive for the

Table 11.2 Climate-resilient crops identified by the farmers of the central Himalaya, India

Scientific name	Common name	Local name
Millets		
Pannicum milliaceum	Hog-millet (rainfed)	Cheena
Echinochloa frumentacea	Barnyard millet	Jhangora
Eleusine coracana	Finger millet	Koda
Setaria italica	Foxtail millet	Kauni
Sorghum vulgare	Pearl millet	Junyali
Oil seeds		
Sesamum indicum	Sesame	Till
Pulses		
Glycine spp.	Soyabean	Kala bhatt
Macrotyloma uniflorum	Horsegram	Gahat
Pisum spp.	Kong	Bheda
Vigna angularis	Adkuki bean	Rains
Cajanus cajan	Pigeon pea	Tor

cultivation of traditional crops amidst the changed socio-economic scenario, acculturation and high rate of out-migration (Maikhuri et al. 2001).

Though good intentioned, the major policies in Uttarakhand largely benefit farmers with large land holdings as subsidies are provided for raising orchards, floriculture, medicinal plants and pisciculture and also in terms of chemical fertilizers, farm implements and irrigation. There is a huge scope for promoting cultivation of traditional crops which have high nutraceutical value and stress tolerance (Maikhuri et al. 1999). However, of late some positive changes are occurring at all scales, and value of these crops is gradually being appreciated by local as well as mainstream communities for their unique taste and nutritional qualities as brought out by several studies carried out in the region (Maikhuri et al. 1997, 1999 & 2001). With the recognition of the Himalaya as one of the 35 global biodiversity hotspots (Myers et al. 2000), organic farming is being considered as one of the means of conserving biodiversity in human-managed ecosystems and for mitigation and adaptation to climate change (Narayan-Parker 2005). Taking note of the new knowledge, the National Environmental Policy (2006) also emphasizes supporting cultivation of traditional crops organically in mountains and enables marginal farmers to realize a higher price. Many of the traditional crops grown in the region are now demanded in the national or global markets (Maikhuri et al. 2015). The government of Uttarakhand established a network of enterprising farmers and through Master Trainers trained them in organic agriculture and maintaining records required for organic certification. Alongside, the state government also made provisions for financial support to farmers by adopting 'biovillage' approach and developing community marketing facilities (Maikhuri et al. 2015). Women are the backbone of traditional hill agriculture and hence custodians of agrobiodiversity; policy support for traditional crops would help empower them and improve their control over resources as well.

11.4.6 Capacity Building in Available Climate-Resilient Technologies

Bringing about the suitable changes in technologieses in farming sector is important for the continuous process of socio-economic development. However, poor access / adoption to appropriate farming technologies has been identified one of the main constraints for securing meaningful livelihood-earning opportunities, reducing drudgery and minimizing natural resource degradation in the central Himalaya despite availability of some of the climate resilient technologies which have been successfully developed, tested, upgraded and demonstrated through action and participatory research with a sizable number of farmers in Uttarakhand (Maikhuri et al. 2010, 2013). To overcome the said constraint, a total of 30 training programmes (each of 2–3 days) on rural technologies were organized during 2010–2017 that helped in training 580 farmers, 40 NGOs, 70 government officials and 250 students from 9 hill districts of Uttarakhand (Table 11.3). As a result, a number of farmers and other stakeholders such as NGOs and educational institutions have adopted some of the potential rural and climate-resilient technologies at various levels (Table 11.4).

11.5 Conclusion

People perceptions on climate change and responses on adaptation across agro-climatic zones reflected that the traditional knowledge played an important role in adapting to climate change impacts on traditional farming practices, the key livelihood-earning activity of more than two-thirds of the population of Uttarakhand Himalaya. Integration of local perceptions and traditional knowledge in formal adaptation and mitigation strategies would help them become locally nuanced and hence more effective in addressing climate change-induced weather variability in the region. On the one hand, the socio-economic sustainability of the farming

Table 11.3 Capacity building and skill development of diverse stakeholders on climate-resilient hill-specific technologies through participatory action research in Uttarakhand

District	Programmes organized	Farmers	NGOs	Govt. staff	Students
Bageshwar	02	45	04	05	10
Almora	02	40	02	03	10
Pithoragarh	02	35	05	05	10
Champawat	02	45	03	07	15
Chamoli	03	150	10	15	70
Rudraprayag	09	165	07	20	70
Pauri	03	20	03	08	20
Tehri	04	50	05	05	35
Uttarkashi	03	30	01	02	10
Total	**30**	**580**	**40**	**70**	**250**

Table 11.4 Technology adoption by stakeholders for coping with climate change impacts in central Himalaya, India

Categories	Technology components	Size and plant used	Adoption (no. of villages)	Adoption (no. of households)	Average income/family/yr. (Rs ± SE)
Protected cultivation	Polyhouse (low-cost)	10mx5mx2.5 m	11	50	4447 ± 107.20
	Nethouse (low-cost)	10mx5mx2.5 m	07	20	4195 ± 130.15
Organic farming	Bio-composting	5mx2mx1m	20	65	1491.66 ± 32.96
	Vermi-composting	5mx2mx1m	18	85	3730 ± 60.16
	Azolla culture	10mx2mx.1 m	10	40	950 ± 29.32
Off-farm technologies	Mushroom cultivation	120 kg base material	17	83	4001.66 ± 24.63
	Honeybee rearing	Single improved wooden box	10	29	1701.66 ± 20.16
	Bioprospecting of wild/semi-domesticated fruits	Five potential plant species used	16	85	5038.33 ± 108.84
Other supporting technologies	Biobrequetting	1 m x 1 m x1m	12	40	4951.66 ± 107.91
	Sweet technology	1 ha	10	10	2816.66 ± 27.32
	Water harvesting tank	6 m x 3 m x1.5m	10	22	1591 ± 29.16
	Zero energy cool chamber	3 m x 1.5 m x 1 m	06	10	1428.33 ± 149.68

system can be strengthened through capacity building of farmers and value addition of the traditional crop products; on the other, there is a need for creating a comprehensive data base on local perceptions, knowledge and innovations for informing the existing sectoral approaches to climate change adaptation and mitigation of the state. The socio-economic sustainability of the farming system can be strengthened through capacity building of farmers and value addition of the traditional crop products. Further, there is a need for creating a comprehensive data base on local perceptions, knowledge and innovations for informing the existing sectoral approaches to climate change adaptation and mitigation of Uttarakhand.

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References

Altieri MA, Nicholls CI (2004) Biodiversity and pest management in agroecosystems, 2nd edn. Haworth Press, New York, p 236

Houet T, Loveland TR, Hubert-Moy L, Gaucherel C, Napton D, Barnes CA, Sayler K (2010) Exploring subtle land use and land cover changes: a framework for future landscape studies. Landsc Ecol 25(2):249–266

IPCC (Intergovernmental Panel on Climate Change) (2001) Climate change 2001: The scientific basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, U.K.

IPCC (Intergovernmental Panel on Climate Change) (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 University Press, U.K.

Maikhuri RK, Ramakrishnan PS (1990) Ecological analysis of a cluster of villages emphasising land use of different tribes in Meghalaya in North-East India. Agric Ecosyst Environ 31(1):17–37

Maikhuri RK, Rawat LS (2013) Climate change impact in central Himalayan agriculture: integrating local perception and traditional knowledge for adaption, climate change and Himalayan informatics. In: Scientific publisher. Jodhpur

Maikhuri RK, Semwal RL, Singh A, Nautiyal MC (1994) Wild fruits as a contribution to sustainable rural development: a case study from Garhwal Himalaya. J Sustain Dev World Ecol 1(1):56–68

Maikhuri RK, Rao KS, Saxena KG (1996) Traditional crop diversity for sustainable development of Central Himalayan agroecosystems. Int J Sust Dev World 3:8–31

Maikhuri RK, Semwal RL, Rao KS, Nautiyal S, Saxena KG (1997) Eroding traditional crop diversity imperils the sustainability of agricultural systems in Central Himalaya. Curr Sci 73(9):777–782

Maikhuri RK, Rao KS, Saxena KG, Semwal RL (1999) Traditional crop-diversity based nutrition & the prospects for sustainable rural development in the Central Himalaya. Himalayan Paryavaran 6:36–42

Maikhuri RK, Rao KS, Semwal RL (2001) Changing scenario of Himalayan agroecosystem: loss of agrobiodiversity, an indicator of environmental change in Uttarakhand Himalaya, India. The Environmentalist 20:23–39

Maikhuri RK, Rao KS, Patnaik S, Saxena KG, Ramakrishnan PS (2003) Assessment of vulnerability of forest, meadows and mountain ecosystems due to climate change. Envis Bulletin 11(2):1–9

Maikhuri RK, Rawat LS, Negi VS, Phondani PC, Bahuguna A, Chamoli KP, Farooquee N (2009) Impact of Climate Change and Coping Strategies in Nanda Devi Biosphere Reserve (NDBR), Central Himalaya, India. In: Proceedings of the International Mountain Biodiversity Conference, Biodiversity Conservation and Management for Enhanced Ecosystem Services: Responding to the Challenges of Global Change, ICIMOD, Kathmandu, pp 135–148

Maikhuri RK, Rawat LS, Negi VS, Bahuguna A, Agarwal SK, Farooquee NA, Phondani PC, Negi CS (2010) Demonstration and dissemination of simple eco-friendly technologies for natural resource management in central Himalaya. Indian J Sci Technol 7:822–830

Maikhuri RK, Rawat LS, Nautiyal S, Negi VS, Pharswan DS, Phondani P (2013) Promoting and enhancing sustainable livelihood options as an adaptive strategy to reduce vulnerability and increase resilience to climate change impact in the Central Himalaya. Environ Sci Eng:555–574

Maikhuri RK, Rawat LS, Semwal RL, Rao KS, Saxena KG (2015) Organic farming in Uttarakhand Himalaya, India. Int J Ecol Environ Sci 41(3–4):161–176

Myers N, Mittermeier RA, Mittermeier CG, da Fonseca GAB, Kent J (2000) Biodiversity hotspots for conservation priorities. Nature 403(6772):853–858

Narayan-Parker D (2005) Measuring empowerment: cross-disciplinary perspectives. World Bank, Washington, DC

Nautiyal S, Maikhuri RK, Semwal RL, Saxena KG (2002) Agroecosystem function around a Himalayan biosphere reserve. J Environ Syst 29(1):71–100

Negi VS, Maikhuri RK (2012) Socio-ecological and religious perspective of agrobiodiversity conservation: issues, concern and priority for sustainable agriculture, Central Himalaya. J Agri Environ Ethics 1:24

Negi VS, Maikhuri RK, Pharswan D, Thakur S, Dhyani PP (2017) Climate change impact in the Western Himalaya: people's perception and adaptive strategies. J Moun Sci 14(2):403–416

Pradhan NS, Khadgi VR, Schipper L, Kaur N, Geoghegan T (2012) Role of policy and institutions in local adaptation to climate change: case studies on responses to too much and too little water in the Hindu Kush Himalayas. ICIMOD, Kathmandu. ISBN/ISSN 9789291152285

Rao KS, Saxena KG (1994) Sustainable development and rehabilitation of degraded village common lands in Himalayas. Publication No. 8, Bishen Singh Mahendra Pal Singh, Dehradun

Rao KS, Maikhuri RK, Semwal RL, Rawat LS, Saxena KG (2012) Climate change adaptation and mitigation: insights from Nanda Devi biosphere in Indian Himalaya. In: Saxena KG, Luohui L, Koji T, Shimako T (eds) Land management in marginal mountain region: adaptation and vulnerability to global change. Bishen Singh mahendra Pal Singh, Dehradun

Ravera F, Martin-Lopez B, Pascual U, Drucker A (2016) The diversity of gendered adaptation strategies to climate change of Indian farmers: a feminist intersectional approach. Ambio 45(3):S335–S351

Saxena KG, Rao KS (2009) Climate change, biodiversity and livelihood in Indian Himalaya: the elements of uncertainty and diverse perceptions. In: Nautiyal S, Nayak BP (eds) Proceeding of climate change data requirement and availability. Centre for ecological economics and natural resources, Institute for social and economic change, Bangalore, pp 78–88

Saxena KG, Ramakrishnan PS, Maikhuri RK, Rao KS, Patnaik S (2004) Assessment of vulnerability of forests, meadows and mountain ecosystems due to climate change. Agriculture Forestry and Natural Ecosystems. pp 163–168

Saxena KG, Maikhuri RK, Rao KS (2005) Changes in agricultural biodiversity: implications for sustainable livelihood in the Himalaya. J Mt Sci 2(1):23–31

Schilling J, Freier KP, Hertige E, Scheffran J (2012) Climate change, vulnerability and adaptation in North Africa with focus on Morocco. Agric Ecosyst Environ 156:12–26

Semwal RL, Nautiyal S, Sen KK, Rana U, Maikhuri RK, Rao KS, Saxena KG (2004) Patterns and ecological implications of agricultural land use changes: a case study from Central Himalaya, India. Agric Ecosyst Environ 102(1):81–92

Semwal RL, Nautiyal S, Maikhuri RK, Rao KS, Saxena KG (2013) Growth and carbon stocks of multipurpose tree species plantations in degraded lands in Central Himalaya, India. For Ecol Manag 310:450–459

Zhu Y, Chan H, Fan J, Wang Y, Li Y, Chen J, Fan JX, Yang S, Hu L, Leung H, Mew TW, Teng PS, Wag Z, Mundt CC (2000) Genetic diversity and disease control in rice. Nature 406(17):718–722

Environmental Changes and Challenges of Tarai and Siwalik of Nepal

12

P. K. Jha, R. A. Mandal, and K. P. Poudel

Abstract

Nepal, a country in Central Nepal, has an elevational range of 60-8848 m in an average 180 km north-south distance. The lowland (Tarai and Siwalik) in southern part of the country is quite close to the tropical zone, which once had rich and well-known forest and wildlife area in Asia, but it has been tremendously transformed into an agricultural landscape in the second half of the last century. As a result of anthropogenic interventions, the lowland has witnessed changes in demography, land use pattern, forest, and biodiversity. In the past four decades, variability in climate has become evident, and temperature rise and erratic rain patterns have been recorded. Food security has become a challenging issue. Almost all districts in the tropical zone are vulnerable to climate risk and hazards, and a few districts in Tarai and Siwalik are the most vulnerable in the country. The lowland zone has not yet received due attention for ecological restoration and sustainable development, particularly in central and eastern Nepal. Strong political will, sound environment and development policies, and effective implementation are direly needed to maintain ecological balance with development.

12.1 Introduction

Nepal, a country located in the Central Himalaya, represents all ecosystems ranging from tropical to alpine ecosystems. Though latitude-wise Nepal is located outside the tropical belt of the earth, nonetheless bioclimatic conditions of lowland areas are

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quite similar to tropical belt. The country lies at 26° 22′ to 30° 27’ N and 80° 04′ to 88° 12′ E between India and China, stretched about 885 km from east to west and a width of 150–240 km from south to north. The total area of the country is 147,181 sq. km, covering 0.09% of the total terrestrial surface and 0.03% of the total earth surface. Altitude varies from 60 m in the south to 8848 m high Mt. Everest (Sagarmatha in Nepal) in the north, within an aerial distance of 180 km, making the physiography of the country interesting. Nepal has flat lowland in the south and hills and valleys in the middle and lofty Himalaya in the north. Thus, physiographically Nepal has five divisions: Tarai and Madhesh (the Indo-Gangetic alluvial plain); the Churia or Siwalik (outer Himalayan range); Middle Mountains (Mahabharat Lek or Lesser Himalaya); High Mountains (Greater Himalaya Range); and High Himal (Palearctic ecozone) ((LRMP 1986). Dobremez (1976) while studying phytogeography broadly categorized the country under six life zones coinciding with an altitudinal interval of 1000 m: (i) tropical (below 1000 m), (ii) subtropical (1000–2000 m), (iii) temperate (2000–3000 m), (iv) subalpine (3000–4000 m), (v) alpine (4000–5000 m), and (vi) nival (altitude>5000 m). Based on its rugged topography and physical environment, Nepal is simply classified as lowland (up to 1000 m), midland (1000–3000 m), and highland (above 3000 m). Following the Holdridge (1967) classification bioclimatically and elevationally, Nepal has lowland (Tarai), pre-montane, lower montane, montane, subalpine, alpine, and nival zones. Hence, Tarai (Madhesh) and Churia or Siwalik are similar to a tropical zone.

Administratively, Nepal is divided into 77 districts, and only 21 districts adjoining Indian borders in the south are closer to tropical and subtropical conditions. In this paper, the concept of tropical zone is around 21 districts in southern part of the country representing Tarai and Chure-Siwalik area – with a physiography below 1000 m of elevation (Fig. 12.1).

12.2 Demographic Change

Human population is not uniformly distributed in Nepal. The lowland and urban centers in the midhills have high density of human population. The population of Nepal increased 3.2 folds in the last 60 years (8.2 million in 1952–1954 to 26 million in 2011). However, at the same time, the population increased by 4.58 folds in the lowland of Nepal. The country’s population in the lowland increased from 2.9 million in 1952–1954 to 13.3 million in 2011 (CBS 2011) (Table 12.1). Thus, increase of population in the tropical zone from 35.2% to 50.27% in 60 years has accelerated because of migration of people mostly from hills and mountains as well as from cross border (Fig. 12.2). There was migration of 1,071,588 people in 2001–2011 (349,132 and 722,456 from mountains and hills, respectively). The migration was low in 1971 – only 399,925 in tropical Tarai (CBS 2001, CBS 2011) (Fig. 12.3). Hence, the lowland, which is the granary of Nepal, turned into a densely populated area affecting the whole ecosystem in the southern part of the country (Regmi and Adhikari 2007). It increased from 85.43 to 392 persons per square km between 1952/1954 and 2011, whereas it remains 56–180 persons per square km in the entire

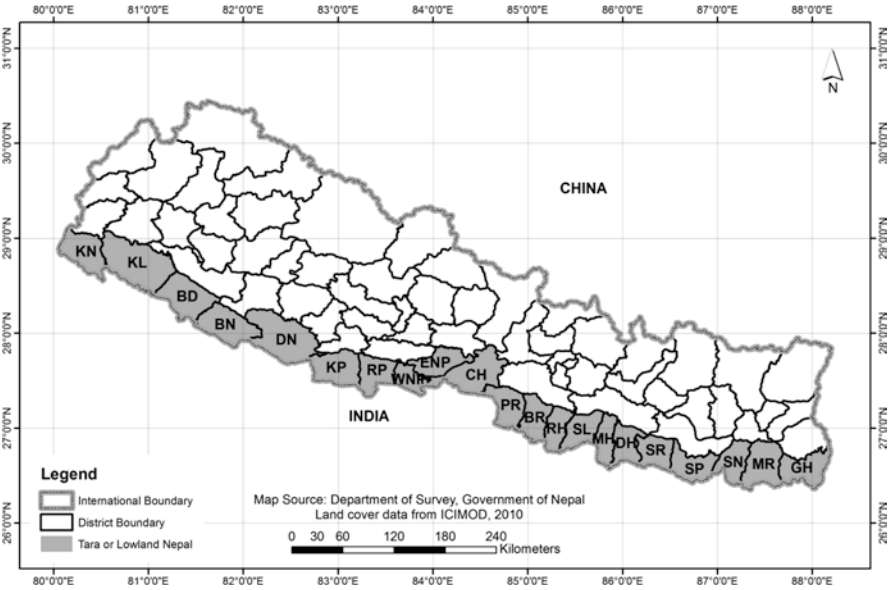

Fig. 12.1 Map of Nepal with lowland (Tarai and Siwalik) part (*KN* Kanchanpur, *KL* Kailali, *BD* Bardia, *BN* Banke, *KP* Kapilvastu, *RP* Rupandehi, *WNP* Nawalparasi west, *ENP* Nawalparasi east, *CH* Chitwan, *PR* Parsa, *BR* Bara, *RH* Rautahat, *SL* Sarlahi, *MH* Mahottari, *DH* Dhanusha, *SR* Siraha, *SP* Saptari, *SN* Sunsari, *MR* Morang, *JH* Jhapa districts)

Table 12.1 Population in Tarai, Nepal, and hills and mountains (percentage in parenthesis)

Year	Hill and mountain	Tarai	Nepal
1952/1954	5,349,988 (64.8)	2,906,637(35.2)	8,256,625
1961	5,991,297 (63.6)	3,421,699(36.4)	9,412,996
1971	7,210,017(52.5)	4,345,966(47.5)	11,555,983
1981	8,466,011(56.4)	6,556,828(43.6)	15,022,839
1991	9,863,019(53.3)	8,628,078(46.7)	18,491,097
2001	11,938,970(51.6)	11,212,453(48.4)	23,151,423
2011	13,175,799(49.73)	1,33,18,705(50.27)	2,64,94,504

Source: CBS (2011)

country (Fig. 12.4). The average annual population growth rate in this zone was 2% as compared to 1.5% in the country between 2001 and 2011 (Fig. 12.5). The country's population since the census period from 1952/1954 to 2011 has continuous high increment in the southern part. It had reached half of the country's total population in 2011. The population density is also high. Since the intial census year, the density has been continously higher in the lowland than the national average. In the most recent census year, it was over 400 persons per square kilometer (Fig. 12.4).

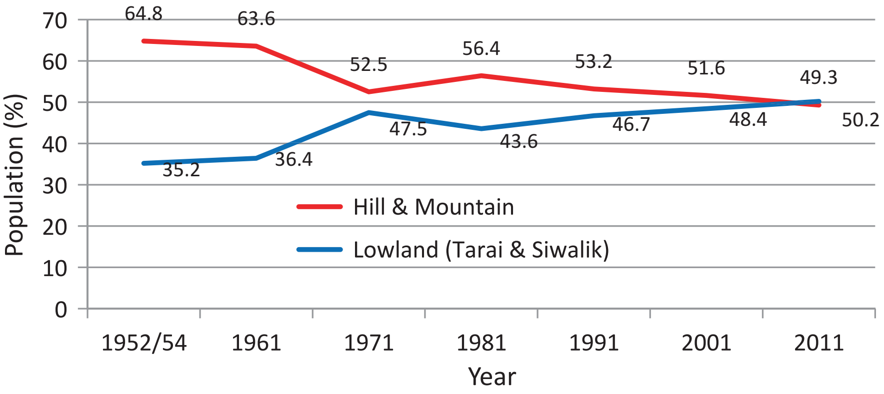

Fig. 12.2 Population share in lowland (Tarai and Siwalik) and hills and mountains

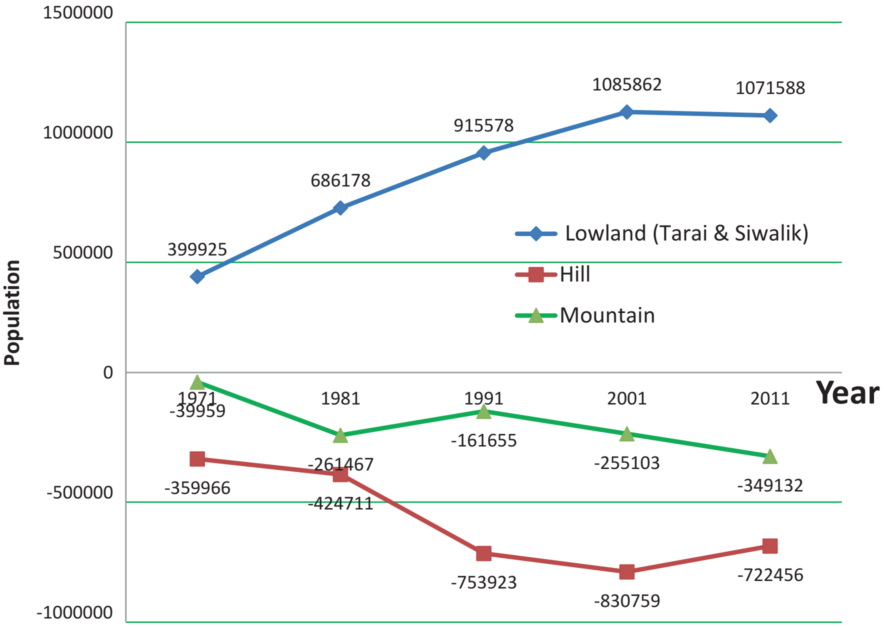

Fig. 12.3 Migration of people from hills and mountains to lowland (Tarai and Siwalik). (Source: CBS, 2001, 2011)

12.3 Land Use Pattern

The migration of people to the lowland region and increased anthropogenic activities vis-à-vis agriculture, urbanization, and industrialization altered the land use pattern in general in Nepal and particularly in the lowland. At present, 40% of the

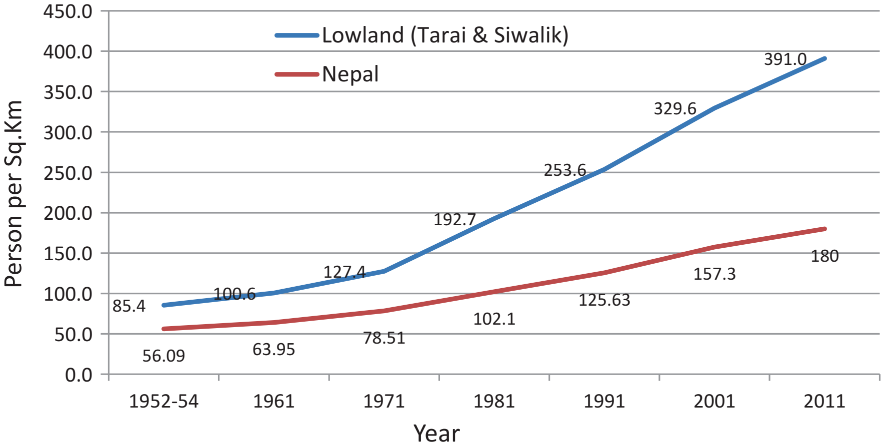

Fig. 12.4 Population density in lowland (Tarai and Siwalik) and Nepal

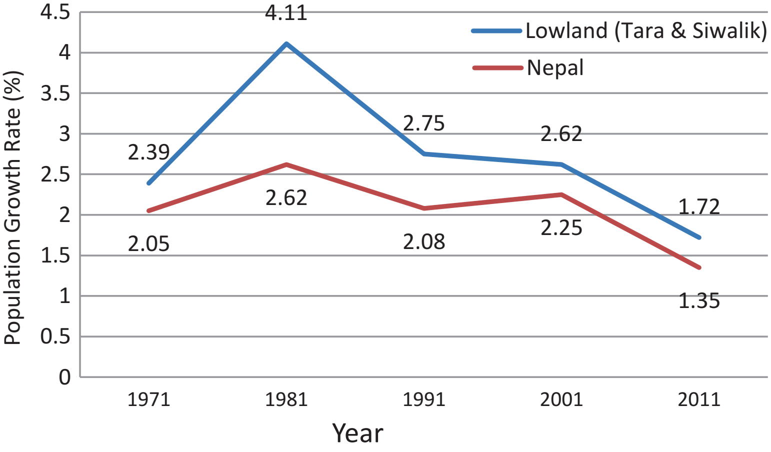

Fig. 12.5 Population growth in different periods in lowland (Tarai and Siwalik) and Nepal

land in Nepal is covered with forest (FRA 2014) and shrubs grow on 4.6% of the land. Twenty-one percent of the land is cultivated, 12% as grassland, 7% as uncultivated land, 2.6% as water body, and the rest of the land is used for other purposes. Increased demand for food and shelter has resulted in huge conversion of forest land into agriculture land for livelihood and income generation in the southern districts, resulting into depletion of biodiversity, degradation of water bodies, and change in the climate system. In the past few years, agriculture land in central Tarai has been converted to aquaculture because farmers do not get adequate production due to weather variability.

Table 12.2 Distribution of land cover types in Nepal and its lowland area in 2010

	Nepal		Lowland area below 1000 m	
Cover types	Area (km^2)	%	Area (km^2)	%
Needle-leaved closed forest	13934.4	9.4	113.9	0.2
Needle-leaved open forest	8267.1	5.6	51.7	0.1
Broadleaved closed forest	21204.3	14.4	16561.0	31.1
Broadleaved open forest	14137.8	9.6	6034.9	11.3
Shrubland	5008.8	3.4	586.4	1.1
Grassland	11635.4	7.9	389.8	0.7
Agriculture	43927.5	29.7	26440.8	49.7
Bare area	15684.5	10.6	1979.2	3.7
Built-up area	468.6	0.3	329.1	0.6
River	839.1	0.6	738.2	1.4
Lakes	45.4	0.0	9.3	0.0
Snow/glacier	12586.5	8.5		
Total	147739[a]	100.0	53234.3	100.0

Source: ICIMOD 2010

[a] Because of the raster image data source, the total area is slightly differed

The land cover map prepared by ICIMOD in 2010 acquired by Landsat TM and ETM+ data presents the tropical area has the domination of the broadleaved forest species cover some 42.4% area. That is the dominant cover beside the cultivated land (49.7%) (Table 12.2).

12.4 Forest and Biodiversity

Forest resources in Nepal have depleted at a rapid rate from 1950 to 1990. Six decades ago, forest land was around 6,343,500 ha, which shrunk to 5,962,038 ha by 2010–2011 (FRA 2014). Forest land which was 37.4% in 1986 has shrunk to 29% in 1994 and significantly increased to 40.36% of Nepal's total land area in 2010. Most of the forest land increased in the midhills. Despite increases of forest land at the national level, the opposite trend has been recorded in the lowland. The Tarai districts (lowland) had 492,100 ha of forests in 1984 which has been reduced to 408,100 ha in 2010–2011 (FRA 2014); thus a loss of 84,000 ha forests in 16 years (i.e., 0.4% per year) has been recorded (Table 12.3). A total of 32,000 ha of forest was cleared between 1991 and 2010/2011, and the highest deforestation occurred in Kailali district (8000 ha), Kapilvastu (5800 ha), Bardiya (4000 ha), Jhapa (2400 ha), and Kanchanpur (1900 ha) (FRA 2014).

The lowland of Nepal has two belts, (i) the lower flatland from 60 to 300 m in altitude (also similar to lower tropical zone or Tarai) and (ii) Chure or Siwalik from 300 to 1000 m altitude (similar to the upper tropical zone) covering 14% and 13% of land area, respectively. Most of the forest in Tarai is confined to the six protected areas (five national parks, viz., Chitwan, Banke, Bardia, Shuklaphanta, and Parsa National Parks, and one wildlife reserve (Koshi Tappu Wildlife Reserve)) (DNPWC 2017).

Table 12.3 Forest land cover change and forest land conversion to other land categories in the Tarai districts from 1991 to 2010/2011

		Rate of change of forest cover (%) – annually		Forest land ((%) converted between 1991 and 2010/2011	
SN	Districts	1991–2010/2011	2001–2010/2011	Agriculture	Water course change
1	Kanchanpur	−0.18	−0.25	73.5	12.8
2	Kailali	−0.56	−0.31	63.6	14.3
3	Bardiya	−0.43	−0.24	59.5	13.7
4	Banke	0.03	0.48	44.1	9.6
5	Kapilvastu	−0.76	−0.95	58.1	8.9
6	Rupandehi	−0.93	−0.31	82.1	2.9
7	Nawalparasi	0.02	0.2	50	4.6
8	Parsa	−0.19	−0.6	76.1	17.9
9	Bara	−0.29	−0.49	31.7	46.5
10	Rautahat	−0.43	−0.96	64.0	35.1
11	Sarlahi	−0.74	−2.07	89.9	9.6
12	Mahottari	−0.04	−0.61	62.2	14.9
13	Dhanusha	−0.76	−5.7	29.8	0.0
14	Sapatari	−1.39	−0.12	92.1	0.0
15	Siraha	0.39	2.57	54.5	19.7
16	Sunsari	−0.45	−0.57	32.5	64.9
17	Morang	−0.16	−0.09	66.8	17.4
18	Jhapa	−1.06	−2.03	51.8	5.0
	Total	−0.4	−0.44	–	–

Source: FRA (2014)

Plant species are very diverse in Nepal's lowland. *Shorea robusta*, the high value timber species in Nepal, is dominantly found in Tarai. The associate species are *Terminalia alata, Anogeissus latifolius, Mallotus philippensis,* and *Lagerstroemia parviflora.* The important riverine timber species are *Acacia catechu* and *Dalbergia sissoo* and their density has decreased as a result of overharvesting. Another species that decreased in the tropical area is *Bombax ceiba.* The lower lowland area has sal (*Shorea robusta*) and mixed broadleaved forests. The second layer of forest in Tarai sal forests has *Careya arborea, Ehretia laevis, Semecarpus anacardium, Dillenia pentagyna*, and *D. indica.* The mixed broadleaved forests in the lower tropical zone are (i) *Terminalia* forest, (ii) tropical evergreen forest, (iii) tropical deciduous riverine forest, (iv) *Dalbergia sissoo-Acacia catechu* forest, and (v) grassland. Climate change is expected to bring changes in these forests. The moist forests in the lowland may expand to north and western part of the country under the climate change scenario (Jha, unpublished data). Alien plant and animal species have invaded the lowland areas in the past three decades. *Chromolaena odorata, Parthenium hysterophorus, Mikenia micrantha, Eichhornia crassipes*, and *Lantana camara* are widely spread invasive alien species in the lowland area and have replaced a number of native plant species. *Tuta absoluta*, an invasive alien pest on tomatoes, first reported in 2016, has recorded its presence in most of the districts in the lowland area.

Churia Siwalik (300–1000 m) has hill sal (*Shorea robusta*) forest. The hill sal forest is characterized by the dominance of sal over other broadleaved trees (*Terminalia, Anogeissus, Lagerstroemia, Adina,* etc.). Several endangered plant species, viz., *Cycas pectinata, Gnetum montanum, Cyathea spinulosa*, and a number of epiphytic orchids, occur in this zone making it an important biodiversity area (TISC 2002). The Churia Siwalik has rare and protected bird species like the giant hornbill (*Buceros bicornis*) and black stork (*Ciconia nigra*) (Bhuju 2013).

Nepal has 118 types of ecosystems based on vegetation types. Of these, 23 are in the lowland (Table 12.4). Over 1800 species of angiosperms occur in the lowland revealing rich biodiversity in limited land areas (Table 12.5). There are 493 bryophytes in midhill followed by 347 in highlands and only 61 in lowland. Moreover, there are around 110 mammals in midhill, 91 in lowland, and 80 in highland (Table 12.5). Thus, 27% land of Nepal representing the lowland, which is similar to tropical and subtropical conditions, has 320 threatened animal species drawing attention to the country. The number of threatened animal species in this zone is high resulting from deforestation, habitat fragmentation and degradation, and overexploitation (Jha et al. 2013). A total of 103 butterflies are threatened in the midland and 38 in the lowland. In addition, 49 and 47 mammal species are threatened in the lowland and midlands, respectively (Table 12.6). Bird and mammal species particularly are more threatened in lowland areas of the country (BPP 1995b). Presently, tigers occupy only 7% of its historic range as a result of deforestation and fragmentation of habitat (Pokharel 2013).

Bush and Hooghiemstra (2005) state that the severest impact occurred on tropical biodiversity because of the vast majority of the species thought to have narrow restricted niches (e.g., small elevational ranges, specific moisture requirements, simple food plant/host). The challenge to conserve biodiversity is not only the preservation of habitat integrity but also maintenance of ecological niches in the wake of climate change. A majority of studies on biodiversity and climate change predict the shift in species range and changes in community composition though there is a lack of organism level studies in Nepal (Bhattacharya et al. 2017). Tarai has two types of forests based on bioclimatic conditions: dry forests and moist forests. Dry forests are mostly restricted to the lower southern-central part which may get converted to tropical moist forests at several places as rain may increase by 6–12% by 2070 (MOSTE 2014).

Table 12.4 Distribution of ecosystems according to physiographic zones in Nepal

Geographical region	Number of ecosystems
Lowland (Tarai and Siwalik: <1000 m)	10 + 13 = 23
Midhills (1000–3000 m)	52
Highlands (>3000 m)	38
Others	5
Total	118

Source: Dobremez (1976)

Table 12.5 Number of plant and animal species in different physiographic zones of Nepal (figures in parenthesis show percentage)

Group	Tarai and Siwalik (<1000 m)	Midhill (1000–3000 m)	Highland (>3000)
Plantae			
Bryophytes	61(8.4)	493(66.62)	347(46.89)
Pteridophyte	81(21.32)	272(71.58)	78(20.53)
Gymnosperm		16(84.2)	10(52.63)
Angiosperm	1885(36.53)	3364(65.19)	>2000(38.7)
Animals			
Butterflies	325(51.1)	557(88)	82(13.1)
Fishes	154(83.2)	76(41.1)	6(3.2)
Amphibians	22(57.2)	29(67.4)	9(20.9)
Reptiles	68(68)	56(56)	12(13)
Birds	648(77.8)	691(82.5)	413(49.6)
Mammals	91(50.27)	110(60.7)	80(44.2)

Source: BPP (1995a)

Table 12.6 Number of threatened species in different geographical zones in Nepal

Zone	Butterflies	Fish	Herpetofauna	Birds	Mammals	Total
Lowland	38	27	27	179	49	320
Midland	103	19	12	176	47	357
Highland	18	1	2	76	26	123

Source: BPP (1995b)

12.5 Agriculture and Food

Agriculture is the backbone of the country, and more than two-thirds of the economically active population is involved in farming, contributing to around one-third of the gross domestic product (Dahal 2013). Twenty-one percent of the land is under crop cultivation in Nepal. Tarai districts occupy 22.7% of Nepal's land and produces about 55–60% of cereals. Agriculture is still practiced along traditional lines in a large part of the country, and there exists a great potential for higher yields particularly in the southern part of the country. The cultivation area has increased in Tarai districts in the past four decades; there is now little scope for increasing land for cultivation. Therefore, the emphasis now should be to enhance agricultural productivity through technology and proper farm management.

Tarai produces about 67–73% of Nepal's paddy. About two-thirds, in different years, of paddy cultivated area in tropical zone producing around 3.0–3.3 million tons paddy annually, thus producing surplus (60000–80,000 t) paddy grain for the country (Fig. 12.6). However, the central lowland (now province number 2) faces food deficit as agriculture is mostly dependent on the weather. In 2009–2010, despite surplus (178,149 t) food production in the tropical zone (Tarai districts), the country had 6% food deficiency mostly because of weather variability. Paddy

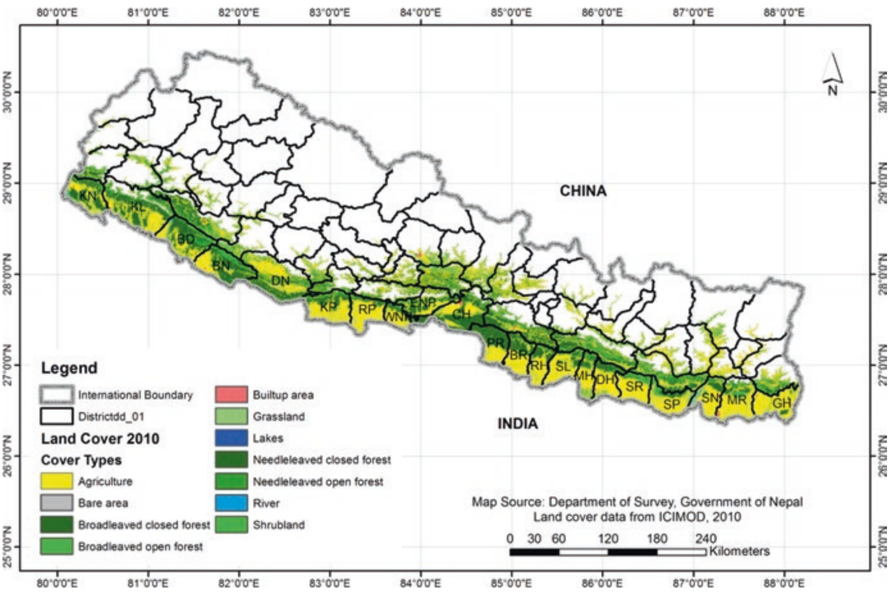

Fig. 12.6 Land cover of lowland area (below 1000 m elevation). (Data Source: ICIMOD 2010)

productivity in general was higher in Tarai (about 2204 kg/ha in Tarai and only 1700 kg/ha in Nepal in 1988/1989); however, degradation in Chure and Bhabar zones has negatively affected the productivity in Tarai. The production rate is now 3297 kg/ha in Tarai and 3271 kg/ha in Nepal in 2015/2016. Despite the high potential of agro-production in Tarai, the population growth has outstripped agricultural output in recent years (Dahal 2013). Deforestation has altered the ecosystem in Tarai; as a result, there is fluctuation in the production of food grains. Nepal now spends billions of rupees in importing food grains mostly from India.

12.6 Climate and Climate Change

Nepal has been listed as one of the most vulnerable countries affected by climate change. The climate in Nepal is governed by its topographic extremes (Shrestha et al. 2012). Nepal falls in the monsoon system of the Indian subcontinent and 80% of rain falls during the monsoon season (PSN 2009). The country receives over 5000 mm rain in Lumle (Kaski district) and around only 300 mm rain in the trans-Himalayan region of Jomsom-Mustang. Eastern Nepal receives more rain than western Nepal. Rain generally increases with altitude up to 3000 m and above that precipitation decreases. Lowland of Nepal usually receives between 1093 and 2616 mm annual rainfall at different locations with an average of 1709 mm. The wide range of temperature experienced in Nepal is from near zero to above 40^0 C in the tropical zone during winter to summer (PSN 2009).

Climate in Tarai is generally hot and rainfall is mostly confined to the monsoon season. Southern part of Nepal remains dry for 5–7 months. The average annual temperature in Tarai ranges from 24.01 to 25.62^0 C. The far western Tarai area (west of Karnali river) is around one degree cooler than central and eastern Nepal because of two reasons: (i) slightly high in latitude and (ii) high forest cover (FRA 2014). Temperature rise in the tropical area in the last 30 years was around 0.02^0 C/year and likely to increase at the same rate in the coming years. Overall annual precipitation in the country is 1663 mm, and based on the data from 1980 to 2010, it decreased by 2% (DHM 2017).

Banke, Mahottari, Dhanusha, Siraha, and Saptari districts receive less than 1500 mm rain per year. The rainfall in the tropical area in the past three decades has decreased by 1.5–2.0 mm/year, i.e., 0.1–0.2% per year. Climatic analysis of Nepal also reveals that mostly rain decreased in the lowland area between 0 and 5% in the 1980–2010 period except in some parts in eastern Nepal (DHM 2017). However, different climate change scenarios predict an increase of rainfall by 6% and 12% by 2050 and 2080, respectively (MoSTE 2014).

12.7 Climate Change, Risk, and Vulnerability

Climate change is impacting all sectors of life on the earth; particularly a developing country like Nepal is more vulnerable because of poor adaptation capacity. Temperature rise rate is high and rainfall pattern is erratic. Every year, a number of people die due to climate-induced extreme events such as floods and landslides. The National Adaptation Programme of Action (NAPA) report states that the Tarai districts have great risk of flood and drought (MoPE 2010a,b). Eastern and central lowlands have high risk of floods, whereas the western lowland has high risk of drought (Table 12.7). Saptari to Parsa districts are highly vulnerable due to climate risk as their socioeconomic adaptation capacity is low. This situation asks for the intervention of government agencies and local community to enhance adaptive capacities of the community in Tarai districts.

12.8 Conservation Efforts

The Government of Nepal has taken several steps to conserve biodiversity in the country. Five national parks and one wildlife reserve have been established in the lowland area. The mega wildlife species are now confined to these protected areas, and some endangered megafauna, viz., Royal Bengal tiger, Asian elephant, one-horned rhinoceros, swamp deer, blackbuck, gharial, etc., are increasing in number owing to conservation efforts of the Government of Nepal. Tarai Arc Landscape project was launched connecting 11 national parks (5 in Nepal and 6 in India); the Chitwan-Annapurna Landscape project has been initiated to develop north-south corridor for conservation. The Chure-Tarai Conservation project is especially created to manage the upper tropical area. Three wetlands in lowland Tarai (Koshi

Table 12.7 Ranking of Tarai districts based on climate risk and hazards

Districts	Flood risk	Drought	Socioeconomic adaptation capacity	Ecological vulnerability	Overall vulnerability
Jhapa	**	*	***	*	*
Morang	**	*	***	*	*
Sunsari	***	*	***	*	*
Saptari	***	***	**	*	***
Siraha	***	***	*	**	***
Dhanusha	**	***	**	**	***
Mahottari	***	**	*	***	***
Sarlahi	*	***	*	**	**
Rautahat	***	*	*	**	**
Bara	**	*	**	**	**
Parsa	***	**	**	**	***
Chitwan	***		***	*	***
Nawalparasi	***	*	**	*	**
Rupandehi	*	**	*	*	*
Kapilvastu	*	**	*	**	*
Dang	*	**	**	*	*
Banke	*	***	***	*	*
Bardiya	*	**	**	**	*
Kailali	*	**	**	*	*
Kanchanpur	*	***	***	*	*

Source: MOPE (2010a,b)

Tappu, Jagadishpur tal, and Ghodaghodi tal) have been declared Ramsar sites to protect wetland ecosystems.

12.9 Challenges

A realization has emerged in the last two decades about the impacts of climate change, deforestation, high migration, population growth, and growing food insecurity in the country. However, lowland area of the country has not yet received due attention at the national level. The landscape is largely altered in the lowland where water table is receding, biodiversity is dwindling, agriculture production is unpredictable, and ecological risks are increased. All these realities raise challenges to the country. Strong political will, sound policies, and effective implementation are needed to integrate conservation with sustainable development. Nepal has to initiate programs for restoration of the ecosystem, and there is a need to increase efforts and their effectiveness.

References

Bhattacharje,A, Anadon JD, Doleck T, Lakhankar T, Shrestha BB, Thapa P, Devkota D, Tiwari S, Jha A, Siwakoti M, Devkota NR, Jha PK, Krakauer N (2017) A review on the impact of climate change on biodiversity in Nepal: current knowledge, gaps and opportunities. Climate 5(4):80. https://doi.org/10.3390/cli5040080

Bhuju DR (2013) In: Jha PK, Neupane FP, Shrestha ML, Khanal IP (eds) Biological diversity and conservation. Nepal Academy of Science and Technology (NAST), Kathmandu, pp 184–189

BPP (1995a) Biodiversity Profile of the Midhills Physiographic Zone. Biodiversity Profile Project, Technical Publication No. 13, Government of Nepal, Ministry of Forests and Soil Conservation and Government of Netherland, Ministry of Foreign Affairs

BPP (1995b) Red Data Book of Nepal. Biodiversity Profile Project, Technical Publication No. 13, Government of Nepal, Ministry of Forests and Soil Conservation and Government of the Netherlands, Ministry of Foreign Affairs

Bush MB, Hooghiemstra H (2005) Tropical biotic responses to climate change and biodiversity. In: Lovejoy TE, Hannah L (eds) Biodiversity and climate change. Publ. The Energy and Resource Institute (TERI), New Delhi, pp 125–137

CBS (2001) Nepal – National Population Census 2001, Tenth Census. Central Bureau of Statistics – National Planning Commission Secretariat, Government of Nepal

CBS (2011) Nepal – National Population Census 2011, Eleventh Census. Central Bureau of Statistics – National Planning Commission Secretariat, Government of Nepal

Dahal KR (2013) Food security. In: Jha PK, Neupane FP, Shrestha ML, Khanal IP (eds) Biological diversity and conservation. Nepal Academy of Science and Technology (NAST), Kathmandu, pp 329–337

DHM (2017) Observed climate trend analysis in the Districts and physiographic zones. Department of Hydrology and Meteorology, Government of Nepal

DNPWC (2017) Protected areas in Nepal. Department of National Parks and Wildlife Coneservation, Babarmahal, Kathmandu

Dobremez JF (1976) Le Nepal Ecologie et Biogeography. Editions du Centre, National de la Becherche Scientifique, Paris

FRA (2014) Terai forests of Nepal. Forest Resource Assessment Nepal, Ministry of Forest and Soil Conservation, Kathmandu

Jha PK, Neupane F, Shrestha ML, Khanal I (eds) (2013) Biological diversity and conservation. Publ. Nepal Academy of Science and Technology (NAST), Kathmandu

LRMP (1986) Land utilization report. Land Resources Mapping Project (LRMP), Kenting Earth Sciences Limited, Government of Nepal and Government of Canada

MOPE (2010a) National Adaptation Programme of Action to Climate Change Ministry of Environment, Kathmandu, Nepal

MOPE (2010b) Climate Change Vulnerability Mapping for Nepal. Ministry of Environment, Government of Nepal

MOSTE (2014) Nepal Second National Communication to United Nations framework Convention on Climate Change. Ministry of Science, Technology and Environment, Government of Nepal

Pokharel GK (2013) Tiger. In: Jha PK, Neupane FP, Shrestha ML, Khanal IP (eds) Biological diversity and conservation. Publ. Nepal Academy of Science and Technology (NAST), Kathmandu, pp 556–562

PSN (2009) Temporal and Spatial Variability of Climate Change Over Nepal (1976–2005). Practical Solution, Kathmandu Nepal. Regions of Nepal (1971–2014). Department of Hydrology and Meteorology

Regmi BR, Adhikari A (2007) Human Development Report 2007. Climate change and human development – risk and vulnerability in a warming world Country Case Study-Nepal. Local Initiatives for Biodiversity, Research and Development (LI-BIRD). Gairapatan, Pokhara Kaski, Nepeal

Shrestha UB, Gautam S, Bawa KS (2012) Widespread climate change in the Himalayas and associated changes in local ecosystems. PLoS One 7(5):36741

TISC (2002) Forest and Vegetation Types in Nepal. Tree Improvement and Silviculture Component, Ministry of Forest and Soil Conservation, Government of Nepal

Ecological Value and Change Sensitivity-Based Approach for Defining Integrity of Forests in West Himalaya, India

13

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Abstract

The extent and diversity of Himalayan forests are well known, and evidence suggests these forests differ significantly from both tropical and temperate forests of the world. These forests are vulnerable to various perturbations (anthropogenic and natural). However, the extent of sensitivity for diverse forests is not known. Therefore, all forests are being treated equally for their conservation and management needs. Considering this, a repeat survey (conducted during 2010–2014) of some earlier studied (1988–1990) representative forest communities in part of Nanda Devi Biosphere Reserve (west Himalaya) provides evidence of importance level and change sensitivity of diverse forest communities. Various composition-based community indices, i.e., Community Importance Index (CII), Community Change Sensitivity (CSS), Community Threat Index (CTI), and Community Integrity (CI) score, were developed to signify importance of each forest community for conservation and management. Two most important communities that maintain compositional integrity in the region include Mixed Silver fir-Rhododendron-Maple (CII-90) and *Quercus floribunda* (CII-83) community. The Community Change Sensitivity (CCS) highlighted *Q. floribunda* (46) and *A. pindrow* (41) communities as most change sensitive, whereas Mixed Oak deciduous (CTI-93) and *Q. floribunda* (CTI-84) communities exhibited maximum

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threats of change. The overall scenarios of communities considering various priorities have been defined. Analysis of priorities, thus defined, suggests that different scenarios can be used for building strategies and management prescriptions depending upon the management objectives. The scenarios discussed herein are meant for (i) ensuring conservation of maximum plant diversity of the reserve by way of using CII, (ii) planning to accommodate the potential changes in community structure (i.e., shift in vegetation/species boundaries and/or change in dominance of forests, etc.) by building on CCS ranking, (iii) addressing the threats of non-native species proliferation through use of CTI, and (iv) defining community stability and resilience in a more holistic way using Community Integrity (CI) scores. The study for the first time provides diverse scenarios of management in regional forests.

Keywords
Himalayan forests · Community index · Community integrity · Conservation · Management

13.1 Introduction

The concept of "integrity," which defines the quality of being unimpaired, sound, or complete, holds great promise for informed decision-making with respect to conservation and management of a given ecosystem. Leopold (1949) stated "a thing is right when it tends to preserve integrity, stability and beauty of biotic community. It is wrong when it tends otherwise." With the growing acceptance of the term "ecological integrity" in scientific literature, as well as in national and international agreements and treaties, the notion of integrity has gone deep into a wide range of ecosystem management situations (Woodley 2010). It has gained wide recognition for prioritizing field surveys, and integrity-based ranking is being used effectively in conjunction with conservation status ranks, for the purpose of planning or action both locally and regionally (Faber-Langendoen et al. 2009).

Forest ecosystems, which play significant role in maintaining world's biodiversity, are impacted by natural and anthropogenic perturbations that affect their integrity and quality (Kapos et al. 2002). However, sensitivity to perturbations varies considerably across forest types (Rawat 2013). The Indian Himalayan Region (IHR) is a predominantly forested landscape (>40% area under forests) that supports a rich diversity of forest types. Many of these forests have high species diversity and are characterized by massive biomass and high net productivity (Singh 1997). More recently, Singh (2014) provided evidence that the Himalayan forests are different from most global forests in their structural and compositional attributes. Their uniqueness and vast contribution for maintenance of the regional biodiversity pool argue that forests in the region deserve greater management attention. However, there have been no attempts to understand change sensitivity of Himalayan forests so as to assign them the priority ranking for management of biodiversity. This study, therefore, intends to develop an integrity-based approach, based on field studies conducted in representative altitude transects in buffer zones of Nanda Devi Biosphere Reserve in west Himalaya to answer the following two management questions: (i) Do the forest communities vary significantly in their ecological importance? (ii) Do the "integrity" considerations provide definite clues for addressing diverse management objectives? The aim is to develop various indices using diverse attributes of vegetation composition to rank the forest communities as per their management priorities. This is a first of its kind approach developed for representative forests in west Himalaya.

13.2 Methods

13.2.1 Study Area and Site Selection

The vast Nanda Devi Biosphere Reserve (NDBR) (30^{0}06′–31^{0}04’ N to 79^{0}13′–80^{0}17′ E) has a unique combination of diverse ecosystems including traditional agro-ecosystems, various types of temperate forests, alpine meadows, and glaciers. It covers a total of 6407.03 km^2 (core zone 712.12 km^2; buffer zone 5148.57 km^2; transition zone 546.34 km^2) and represents the west Himalayan highland (2b) province of the biogeographic zone-Himalaya (Rawal and Rawat 2012). The study site was located in Pindari-Sunderdhunga-Kafni (PSK) area in Kumaun region of west Himalaya along the buffer zone of NDBR. Previously investigated tracts/plots (Bankoti 1990) were revisited for intensive investigation during 2010–2014.

13.2.2 Sampling and Data Collection

We investigated patterns of compositional change using data from 30 forest stands previously investigated by Bankoti (1990). Specific details of locations (altitude, latitude, and longitude) were recorded using a handheld Global Positioning System [GPS (Garmin make-12)]. Toward ensuring the compatibility of datasets with earlier studies, the approach in the present study was kept similar to the previous study (Bankoti 1990). Broadly, the components of sampling were as follows: In each stand, ten (10 × 10 m) quadrats were laid randomly for enumeration of trees and saplings. Five (2 × 2 m) sub-quadrats (within each 10 × 10 m quadrat) for shrubs and seedlings and ten (1× 1) for enumeration of herbs were standardized.

Quadrat information was generated following standard phytosociological methods (Misra 1968; Muller-Dombois and Ellenberg 1974; Dhar et al. 1997). In each quadrat circumference at breast, height (CBH at 1.37 m from the ground) of all tree individuals was measured, and individuals were grouped as tree >31 cm, sapling 11–30 cm, and seedling <11 cm CBH. The number of individuals of each species was recorded for shrubs and herbs in each sub-quadrat. The woody species having branching from the base were considered as shrubs (Saxena and Singh, 1982). Among herbs, angiosperms and pteridophytes were enumerated. In case of clumped shrub and herb species, each stump/tiller of the clump was counted as one individual.

13.2.3 Taxonomic Identity, Nativity, and Endemism

The plant specimens, collected and preserved following standard herbarium methods (Jain and Rao 1977), were identified with the help of available flora/monographs (Naithani 1984; Sharma et al. 1993; Kumar and Panigrahi 1995; Hajra et al. 1997; Gaur 1999; Mukherjee and Constance 1993; Dikshit and Panigrahi 1998). Plant identity was further authenticated by consulting the regional herbarium in the Botany Department, Kumaun and Garhwal Universities, India.

Species having their origin and distribution in the Himalayan region were considered as native species (Samant et al. 1998). The extent of geographical distribution was used to define endemism (Dhar and Samant 1993; Dhar et al. 1996; Dhar 2002), and species restricted to Indian Himalayan Region (IHR) were considered as endemic. Species with slightly extended distribution in neighboring Himalayan countries were identified as near endemics (Dhar and Samant 1993).

13.2.4 Sensitivity Assessment and Identification of Priority Areas for Conservation

13.2.4.1 Ecological Importance of Communities

Importance of forest communities was defined using richness, representativeness, and uniqueness values as follows:

(i) Richness of the target communities was defined as a sum of species richness in different tree species strata (i.e., trees, saplings, and seedlings) along with abundance and dominance. The absolute values under each richness feature were assigned equal weight by assigning an index value of 100 to the best score in given category and calculating corresponding Community Richness Index (CRI).
(ii) Using native species as indicators of representativeness (Dhar et al. 1997), Community Representativeness Index (CRPI) was calculated as a mean of cumulative values of Native Species Richness (NR) and Native Species Abundance (ND). The absolute values under each representative feature were assigned equal weight by assigning an index value of 100 to the best score in given category and calculating corresponding Community Representativeness Index (CRPI).
(iii) Further, recognizing that endemics represent unique biodiversity elements (Dhar 2002), the Community Uniqueness Index (CUI) was calculated to represent the mean of the Richness and Abundance Index of endemics for respective communities. Communities were thus assigned equal weight by assigning an index value as in earlier cases (e.g., CRPI).
(iv) Finally, Community Importance Index (CII) was calculated as mean of the richness (CRI), representativeness (CRPI), and uniqueness (CUI) index values

Table 13.1 Weight values for community richness, representativeness, uniqueness, and threat parameters

	Phytosociological												Uniqueness						**CII**	Threat			**CTI**
Site	RS	RSP	RT	DS	DSP	DT	DIS	DISP	DIT	TBA	CNP	**CRI**	RN	DN	**CRPI**	RE	DE	**CUI**		RNN	DNN	DST	
AN	47	50	61	65	55	86	88	89	71	43	78	**67**	76	77	**77**	63	73	**68**	**71**	100	75	66	**80**
MOD	68	67	91	100	37	90	91	100	100	85	100	**85**	81	66	**74**	63	100	**82**	**80**	94	84	100	**93**
HS	26	17	17	39	100	79	28	24	22	12	63	**39**	77	64	**71**	72	42	**57**	**56**	100	85	59	**81**
QF	100	100	87	69	27	71	98	93	84	84	86	**82**	99	47	**73**	93	94	**94**	**83**	69	100	82	**84**
QS	84	78	100	57	32	88	90	79	81	91	61	**77**	90	56	**73**	79	48	**64**	**71**	80	93	79	**84**
MD	84	83	91	38	31	76	93	93	98	58	79	**75**	89	82	**86**	97	63	**80**	**80**	83	71	45	**66**
MSO	47	72	91	41	23	49	90	90	95	59	69	**66**	85	71	**78**	70	80	**75**	**73**	88	80	62	**77**
MSRM	74	83	100	44	32	69	100	100	96	100	79	**80**	91	100	**96**	97	89	**93**	**90**	80	55	37	**57**
AP	47	56	74	26	30	60	83	96	80	30	75	**60**	92	92	**92**	80	89	**85**	**79**	79	62	27	**56**
MBS	47	61	43	32	33	66	99	99	60	23	75	**58**	88	96	**92**	69	69	**69**	**73**	83	59	41	**61**
BU	26	33	35	18	37	100	60	68	46	18	81	**48**	100	61	**81**	100	61	**81**	**70**	67	89	41	**66**

for individual community in target site. The communities in ascending order of CII reflected increasing ecological importance. Weight values for community richness, representativeness, uniqueness, and threat parameters are preseneted in Table 13.1.

13.2.4.2 Change Sensitivity

The Community Change Sensitivity (CCS) of representative forest communities was analyzed using the changes in values of compositional features that have occurred between two time intervals (i.e., 1988–1990 and 2010–2014). Various compositional features, such as species richness, density, diversity, and Total Basal Area (TBA), were considered for analysis. CCS was determined as follows:

(i) Absolute values of change w.r.t. each compositional feature for individual community were calculated as CV = PV-EV (CV, Change Value; PV, Present Value; EV, Earlier Value). The absolute values of changes for various compositional features (Richness, RI; Diversity, DI; Abundance, AB; Dominance, DO) are presented. While CVI (Change Value Index) in case of RI, DI, and AB was assessed for trees, saplings, and seedlings, the DO was considered for trees only.
(ii) Possible bias for any feature and community was removed by assigning values in each case with equal weight. An index value of 100 was assigned to the best score for each compositional feature and corresponding index score (i.e., CVI) was calculated for each of the community.
(iii) The mean of all CVIs (i.e., CVI, Richness; CVI, Diversity; CVI, Abundance; CVI, Dominance) for the target community was considered as the Community Change Sensitivity (CCS). Following CCS, communities were thus scored as 1–11 (1 representing the highest sensitivity and 11 the lowest) (Table 13.2).

13.2.4.3 Community Integrity

Considering that “ecological integrity” measures the composition, structure, and function of an ecosystem, as compared with its natural or historical range of variation (Tierney et al. 2009), integrity of forest communities was assessed for study area. As the current datasets were limited to compositional features of forests, therefore, the integrity of communities (CI) was evaluated using these features only. In fact, these features have been used in defining various indices developed for the communities based on compositional features (i.e., CCS, CRI, CRPI, and CII) along with Community Threat Index (CTI) as follows:

(i) CII of the communities, which integrates richness, representativeness, and uniqueness status, was used as the reflection of current state of community integrity. The communities with higher CII values reflected higher level of integrity. In this way, communities were assigned integrity scores from 1 to 11 (1 being at minimum level of integrity and 11 at highest).
(ii) Considering that the communities with higher level of CCS are more likely to affect the integrity of communities in near future, the CCS scores of communities were used in reverse direction (i.e., community with minimum CCS score

Table 13.2 Weight values for change sensitivity

Communities	CVI (Change Value Index) for Vegetation parameters											Scores
	RIS	RISP	RIT	ABS	ABSP	ABT	DIS	DISP	DIT	DO	**CCS**	
AN	0	67	0	16	40	9	18	34	28	36	**25**	8
MOD	25	33	0	100	83	14	25	28	26	74	**41**	3
HS	13	17	50	16	100	100	0	5	100	6	**41**	4
QF	100	83	50	45	22	1	39	5	60	53	**46**	1
QS	13	100	50	17	2	6	93	33	56	22	**39**	5
MD	100	67	0	14	40	4	29	5	5	9	**27**	7
MSO	0	0	0	13	32	7	14	100	5	1	**17**	11
MSRM	38	67	0	3	32	12	0	10	49	100	**31**	6
AP	0	50	100	3	69	5	100	24	51	4	**41**	2
MBS	38	0	0	15	28	4	61	14	53	11	**22**	9
BU	0	17	50	7	20	13	4	19	33	23	**18**	10

AN A. nepalensis, *MOD* Mixed Oak deciduous, *HS H. salicifolia*, *QF Q. floribunda*, *QS Q. semecarpifolia*, *MD* Mixed deciduous, *MSO* Mixed Silver fir-Oak, *MSRM* Mixed Silver fir-Rhododendron-Maple, *AP A. pindrow*, *MBS* Mixed Birch-Silver fir, *BU B. utilis, RS, DS, DIS*, richness, density, and diversity of seedlings, *RSP, DSP, DISP*, richness, density, and diversity of saplings, *RT, DT, DIT, TBA*, richness, density, diversity, and total basal area of tree species, *RN, RNN, RE, DN, DNN, DE*, richness and density of native, non-native, and endemic species, *CNP* canopy cover, *DST* disturbance, *CRI* Community Richness Index, *CRPI* Community Representativeness Index, *CUI* Community Uniqueness Index, *CII* Community Importance Index, *CTI* Community Threat Index to retain maximum integrity). The communities were thus scored w.r.t. CCS from 1 to 11 (i.e., 11 being least affected and 1 being highest).

(iii) Further, assuming that level of existing threats (e.g., level of canopy disturbance, grazing) and proportional representation of non-native species in the community (i.e., richness and abundance) would also define the potential of threat to the forest composition, the Community Threat Index (CTI) was calculated (Table 13.1). In this case, richness of non-native species and their proportional density along with level of canopy disturbance were used to arrive at various weights, as in case of other parameters (i.e., richness, representativeness, uniqueness, etc.). Finally, the CTI values were obtained as mean of all weights in each case (Table 13.1). The CTI scores (1–11) were assigned to highlight status of threat in each community (11 representing the minimum threat and hence highest integrity and 1 the maximum threat and minimum integrity).

(iv) Finally, the Community Integrity (CI) value for each community was calculated as CI = CII Score + CCS Score + CTI Score. The communities were arranged in highest to lowest CI range. The higher CI values thus reflect the greater integrity of community (Table 13.3).

13.3 Results

13.3.1 Ecologically Important Communities

The relative importance of communities in the study area, using various compositional attributes, is presented (Fig. 13.1a–c). Collectively, all these indices determined the Community Importance Index (CII), which reveals that Mixed Silver fir-Rhododendron-Maple (CII-90) and *Q. floribunda* (CII-83) communities with higher CII values are the most ecologically important communities in the study area (Fig. 13.1d).

13.3.2 Vulnerable Communities

Based on Community Threat Index (CTI), Mixed Oak deciduous (CTI-93), *Q. floribunda* (CTI-84), *Q. semecarpifolia* (CTI-84), and *Alnus nepalensis* (CTI-80) communities with higher CTI values emerged as vulnerable communities that are facing greater threats regarding compositional integrity. *A. pindrow* (CTI-56) and Mixed Silver fir-Rhododendron-Maple (CTI-57) were, however, least threatened (Table 13.1).

Table 13.3 Compositional integrity of communities and scenarios for addressing diverse management issues

Site/communities	Indices and Scores						CI (CII+CCS+CTI Scores)	Rank based on CI Score
	CII	Scenario I	CCS	Scenario II	CTI	Scenario III		Scenario IV
PSK								
AN	71	4	25	8	80	5	17	**VII**
MOD	80	9	41	3	93	1	13	**IX**
HS	56	1	41	4	81	4	9	**XI**
QF	83	10	46	1	84	2	13	**VIII**
QS	71	3	39	5	84	3	11	**X**
MD	80	8	27	7	66	8	23	**II**
MSO	73	6	17	11	77	6	23	**III**
MSRM	90	11	31	6	57	10	27	**I**
AP	79	7	41	2	56	11	20	**V**
MBS	73	5	22	9	61	9	23	**IV**
BU	70	2	18	10	66	7	19	**VI**

AN A. nepalensis, *MOD* Mixed Oak deciduous, *HS H. salicifolia*, *QF Q. floribunda*, *QS Q. semecarpifolia*, *MD* Mixed deciduous, *MSO* Mixed Silver fir-Oak, *MSRM* Mixed Silver fir-Rhododendron-Maple, *AP A. pindrow*, *MBS* Mixed Birch-Silver fir, *BU B. utilis, CII* Community Importance Index, *CCS* Community Change Sensitivity, *CTI* Community Threat Index

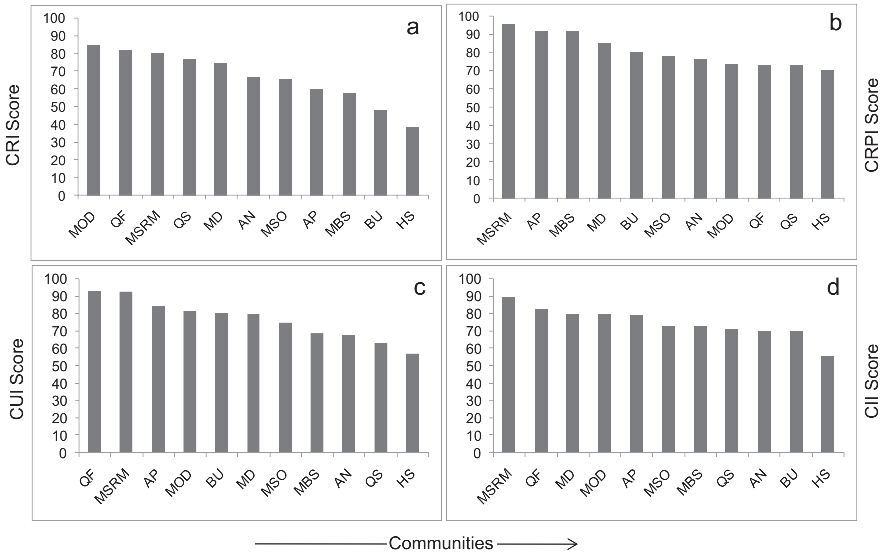

Fig. 13.1 Relative importance of communities in PSK (**a**–**d**) using various compositional attributes; (**a**) CRI, Community Richness Index; CRPI, Community Representativeness Index; CUI, Community Uniqueness Index; CII, Community Importance Index. (*AN Alnus nepalensis*, *MOD* Mixed Oak deciduous, *HS Hippophae salicifolia*, *QF Quercus floribunda*, *QS Quercus semecarpifolia*, *MD* Mixed deciduous, *MSO* Mixed Silver fir-Oak, *MSRM* Mixed Silver fir-Rhododendron-Maple, *AP Abies pindrow*, *MBS* Mixed Birch-Silver fir, *BU Betula utilis*)

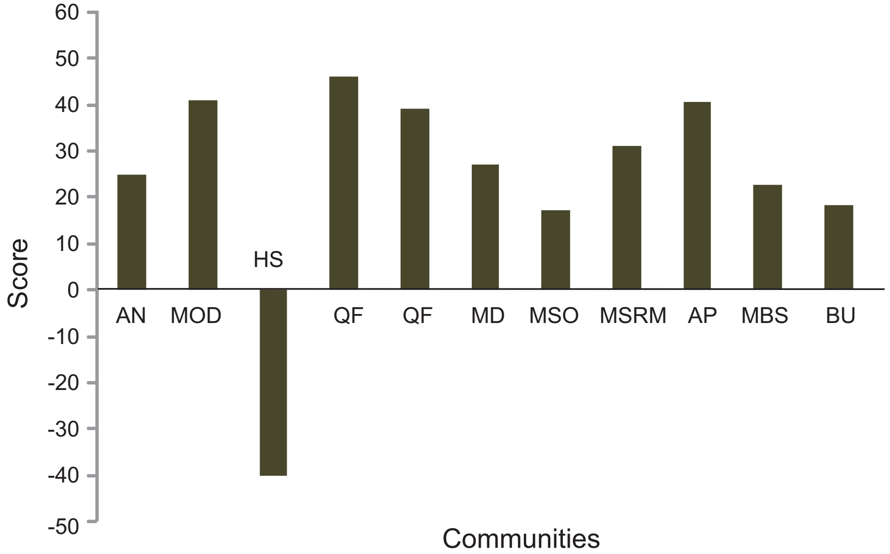

Fig. 13.2 Identification of change-sensitive communities in PSK based on changes in phytosociological parameters between 2010–2014 and 1988–1990. (*AN Alnus nepalensis*, *MOD* Mixed Oak deciduous, *HS Hippophae salicifolia*, *QF Quercus floribunda*, *QS Quercus semecarpifolia*, *MD* Mixed deciduous, *MSO* Mixed Silver fir-Oak, *MSRM* Mixed Silver fir-Rhododendron-Maple, *AP Abies pindrow*, *MBS* Mixed Birch-Silver fir, *BU Betula utilis*)

13.3.3 Change-Sensitive Communities

Considering the Community Change Sensitivity (CCS), the maximum CCS was recorded for *Q. floribunda* (46) followed by *A. pindrow* (41) and Mixed deciduous community (41) and minimum for Mixed Silver fir-Oak (17) community (Fig. 13.2). One community, *H. salicifolia* (41), also showed case of higher sensitivity but in reverse direction by losing quantity of compositional attributes.

13.3.4 High Integrity Communities

The overall integrity scores of communities using various indices have been depicted (Table 13.1). Mixed Silver fir-Rhododendron-Maple (CI-27), Mixed deciduous (CI-23), and Mixed Silver fir-Oak (CI-23) reflected higher integrity, while *H. salicifolia* (CI-9) was identified to have minimum compositional integrity.

13.4 Discussion

13.4.1 Change Sensitivity

Indications of change, as generated through studies over two time periods, provide a strong basis to explore the change sensitivity of forest communities. As the Community Change Sensitivity (CCS) concurrently considered various compositional features (i.e., species richness, density, diversity, and TBA), the outcome (i.e., CCS-based ranking) reflects relatively unbiased sensitivity ranking of communities. These findings can benefit overall process of conservation and management planning for target area. Specifically, the sensitivity ranking of forest communities is likely to help the reserve managers to define short- and long-term management priorities within the Nanda Devi Biosphere Reserve – a world heritage site.

Among forest communities, *Q. floribunda*, Mixed Oak deciduous, *H. salicifolia*, and *A. pindrow* communities exhibit higher sensitivity for compositional change. Changes in case of *H. salicifolia* can be assigned to natural hazards (i.e., uprooting of community due to river flooding). Change sensitivity of other communities can be explained on account of various compositional features. For example, CCS in Mixed Oak deciduous and *Q. floribunda* community can be largely attributed to decline in TBA and increase in total seedling density along with species richness. The *A. pindrow* community, however, owes greater sensitivity to increased richness of species in saplings and tree layers, along with the decline of density and diversity in the seedling layer. Among forest communities, Mixed Silver fir-Oak and *Betula utilis*, with lower values of CCS, emerged as change-resilient. In general, the communities in lower elevation zone, with relatively higher CCS scores, appear in a more dynamic state.

The management interpretations of results suggest that the reserve requires differential management strategies to accommodate the pace and direction of changes.

These emerging results also need to be seen in the perspective for re-defining the successional status of communities. For instance, riverine communities, like *A. nepalensis*, *H. salicifolia*, etc., in the region (Dhar et al. 1997) are usually considered comparable with “habitat pioneer” communities of Ohsawa (1991). However, *A. nepalensis* stands in present study behave more of a mid-successional community that tends to be stable (i.e., less sensitive to changes). Two sub-alpine forests, *A. pindrow* and *B. utilis*, are reported as climatic climax for west Himalaya (Champion and Seth 1968; Dhar et al. 1997). Of these, *B. utilis* with low CCS scores behave more of a stable community, whereas *A. pindrow* exhibits greater dynamism of change. Oaks, which form climatic climax for the region (Champion and Seth 1968; Singh and Singh 1987), interestingly exhibit relatively higher sensitivity. This seems to be an important observation in the context of wider generalizations. Theurillat and Guisan (2001) reviewed potential impacts of climate change on vegetation in European Alps and suggested differential responses of climatic climax communities and reported that for climatic climax communities, new plant communities are likely to develop and, partially or totally, replace present ones. However, the physiographic, edaphic, and land use factors will still play a determining role locally and regionally. In this context, while discussing distribution of west Himalayan forest types in eastern part of Nepal Himalaya, Ohsawa et al. (1986) observed changes in the successional status, from climax to habitat pioneers, of west Himalayan forest-forming species (*P. wallichiana*, *P. roxburghii*, and *Q. semecarpifolia*) along the west to east gradient of the Himalaya. Therefore, present observations of higher change sensitivity of two climatic climax Oak forests tend to agree with this generalizations. However, more datasets from future long-term studies on patterns of composition as well as climatic and edaphic features would provide more convincing explanations.

13.4.2 Important Communities/Areas

The inherent complexity makes prediction of behavior of ecological systems difficult, yet the management and policy decisions require information on status, condition, and trends of ecosystems (Andreasen and Neill 2001). In this context, we presume relative conservation and management status of forest communities may be considered as evidence-based inputs for determining the compositional integrity of target communities.

The first level of ranking of communities for maintaining the local and regional species pool emerges from Community Richness Index (CRI). Thus, Mixed Oak deciduous, *Q. floribunda*, and Mixed Silver fir-Rhododendron-Maple communities are most promising. Community Representative Index (CRPI) and Community Uniqueness Index (CUI) have particular relevance under the emerging principle of habitat management for biodiversity. Sampson and Knopf (1982) argued that many local-scale management practices that encourage diversity within local communities ignore large-scale problems of maintaining viable populations of endemic and native species within the landscape. In this context, CRPI that demonstrates the contribution of native species to the community composition identified Mixed

Silver fir-Rhododendron-Maple, Mixed Birch-Silver fir, and *A. pindrow* as most representative communities. In general, the trend of native species occurrence suggested higher representation of native species in communities located on higher elevation that corresponds with earlier reports of significant increase in nativity in higher elevations in the region (Dhar et al. 1997). The CUI further narrows the conservation priority by way of defining the endemic (unique species) support potential to communities. The communities having best CUI were not necessarily the best CRI and/or CRPI communities. *Q. floribunda* and Mixed Silver fir-Rhododendron-Maple with highest contribution of unique species (endemics) emerged as the best. Endemics being the high priority biodiversity elements (Dhar 2002), communities with greater CUI in the reserve would deserve higher management attention.

Individually, each of the above index helps in defining different management priorities and ranking of communities. However, ranking through one index does not always match with the ranking of other index. This makes the situation complex for a manger to decide upon a particular value/ranking. To facilitate quick decision-making, the Community Importance Index (CII) as a cumulative reflection serves the purpose. The ranks of communities thus reflect the cumulative contribution of all these attributes. In this respect, the most important two communities that maintain compositional integrity include Mixed Silver fir-Rhododendron-Maple and *Q. floribunda*.

13.4.3 Change Scenarios and Alternative Strategies

Having defined the compositional integrity-based Community Importance ranks for forest communities, it is important to further elaborate on scenarios under ongoing and potential changes that may affect the compositional integrity of communities. Community Integrity (CI), defined on the basis of two indices (i) the Community Change Sensitivity (which defines the intensity of change dynamism in a given community) and (ii) the Community Threat Index (CTI) that provides an indication of existing and looming threats on a community (defined as an outcome of non-native contribution and canopy disturbance), gives a more refined and more realistic ranking of individual community. It is clear that depending on the conservation and management objective to be addressed, different scenarios can be used for building strategies and providing management prescriptions. For instance, scenario I management interventions would follow ecological value ranking of forests communities, whereas scenario II management would largely focus on change sensitivity of communities, while scenario III calls for management interventions to address levels of vulnerability, and scenario IV suggests priorities on more holistic level by considering overall integrity of forests. We believe this ranking provides a basis for looking into issues of management in a more holistic manner. The Community Integrity (CI) score not only defines the level of relative resistance of a community to environmental changes but also exhibits its ability to recover to original state after perturbation (Andreasen and Neill 2001) and, therefore, emerges as the best way of defining conservation and management priorities across forest communities in the region and elsewhere.

13.5 Conclusions

- The Community Change Sensitivity (CCS)-based ranking helps in identification of most sensitive communities and thus determines potential intervention sites as management priorities within the reserve.
- The Community Integrity (CI) scores highlight relative resilience of the communities. The four indices, as suggested through this study, would help in addressing different management objectives: (i) the CCS-based prioritization is recommended for accommodating potential changes over short- to long-term duration; (ii) the CII-based ranking is recommended to ensure conservation of maximum plant diversity; (iii) the Community Threat Index (CTI) can be used as a potential tool to address issues like non-native species proliferation and canopy disturbances; and (iv) on a more holistic level, CI (Community Integrity) forms a basis for prioritization based on relative resilience and stability of forest communities.
- The impact score simply defines a conservation value that has been estimated across some important forest sites and can contribute to the identification of those sites whose loss is likely to have greater impact on the conservation status of the area/region.
- On a wider scale, for instance, India as a nation or Himalaya as region, the present approach of assigning scores can be incorporated into conservation priority analyses.

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References

Andreasen JK, Neill RVO (2001) Consideration for the development of a terrestrial index of ecological integrity. Ecol Indic 1:21–35

Bankoti NS (1990) Woody vegetation along elevational gradient (2000–3600 m) of upper Pindari catchment (Kumaun Himalaya). Ph.D. Thesis (Botany), Kumaun University, Nainital

Champion HG, Seth SK (1968) A revised survey of forest types of India. Government of India, Publication Division, New Delhi

Dhar U (2002) Conservation implication of plant endemism in high-altitude Himalaya. Curr Sci 82(2):141–148

Dhar U, Samant SS (1993) Endemic diversity of Indian Himalaya I: Rannunculaceae and II: Paeoniaceae. J Biogeogr 20:659–668

Dhar U, Rawal RS, Samant SS (1996) Endemic plant diversity in Indian Himalaya III: Brassicaceae. Biogeographica 72(1):19–32

Dhar U, Rawal RS, Samant SS (1997) Structural diversity and representativeness of forest vegetation in a protected area of Kumaun Himalaya, India: implications for conservation. Biodivers Conserv 6:1045–1062

Dikshit BK, Panigrahi G (1998) The family Rosaceae in India - revisionary studies on Potentilla L., Sibbaldia L. & Brachycaulos, vol 4. Bishen Singh Mahendra Pal Singh, Dehradun, pp 1–348

Faber-Langendoen D, Master L, Tomaino A, Snow K, Bittman R, Hammerson G, Heidel B, Nichols J, Ramsay L, Young B (2009) NatureServe conservation status ranking system: methodology for rank assignment. NatureServe, Arlington

Gaur RD (1999) Flora of district Garhwal, north west Himalaya with ethnobotanical notes. Transmedia Publication, Srinagar

Hajra PK, Nair VJ, Daniel P (1997) Flora of India, Vol 4, Malpighiaceae-ichapetalaceae. BSI, Calcutta

Jain SK, Rao RR (1977) A handbook of field and herbarium methods. Today and tomorrow’s Printers and Publishers, New Delhi

Kapos V, Lysenko I, Lesslie R (2002) Assessing forest integrity and naturalness in relation to biodiversity, Forest Resources Assessment Working paper 54. FAO, Rome, p 65

Kumar A, Panigrahi G (1995) The family Rosaceae in India - revisionary studies on Cotoneatster Medik, vol 3. Bishen Singh Mahendra Pal Singh, Dehradun, pp 1–292

Leopold A (1949) A sand county almanac and sketches here and there. Oxford University Press, New York, p 228

Misra R (1968) Ecological work book. Oxford & IBH Publishing Company, Calcutta

Mukherjee PK, Constance L (1993) Umbelliferae (Apiaceae) of India. Oxford & IBH, Calcutta

Muller-Dombois D, Ellenberg H (1974) Aims and methods of vegetation ecology. Wiley, New York

Naithani BD (1984) Flora of Chamoli district. Vol. I & II. Botanical Survey of India, Howrah

Ohsawa M (1991) Montane evergreen broad- leaved forests of the Bhutan. In: Ohsawa M (ed) Life zone ecology of Bhutan Himalaya II. Chiba University Press, Japan, pp 89–156

Ohsawa M, Shakya PR, Numata M (1986) Distribution and succession of west Himalayan forest types in the eastern part of the Nepal Himalaya. Mt Res Dev 6(2):143–157

Rawal RS, Rawat B (2012) Nanda Devi Biosphere Reserve-west Himalaya, India. In: Palni LMS, Rawal RS, Rai RK, Reddy SV (ed) Compendium on indian biosphere reserve: Progression during two decades of conservation. Ministry of Environment and Forests, Government of India

Rawat B (2013) Changes in vegetation diversity and plant responses in Nanda Devi biosphere reserve over the last two decades. PhD thesis. Submitted to Kumaun University Nainital, Uttarakhand, India. pp. 182

Samant SS, Dhar U, Palni LMS (1998) Medicinal plants of Indian Himalaya: diversity distribution potential values. Gyanodaya Prakashan, Nainital

Sampson FB, Knopf FL (1982) In search of a diversity ethic for wildlife management. Trans North Am Wildlife Natl Resour Conf 47:421–431

Saxena AK, Singh JS (1982) A phytosociological analysis of woody species in forest communities of a part of Kumaun Himalaya. Vegetatio 50:3–22

Sharma BD, Balakrishnan NP, Rao RR, Hajra PK (1993) Flora of India, Vol 1, Rannunculaceae-Balrclayaceae. BSI, Calcutta

Singh JS (1997) Forests of Himalaya with particular reference to man and forest interactions in Central Himalaya. Proc Indian Natn Sci Acad B 63(3):151–173

Singh SP (2014) Attributes of Himalayan Forest ecosystems: they are not temperate forests. Proc Indian Natn Sci Acad B 80(2):221–233

Singh JS, Singh SP (1987) Forest vegetation of the Himalaya. Bot Rev:52–53

Theurillat JP, Guisan A (2001) Potential impact of climate change on vegetation in the European alps: a review. Climate Change 50:77–109

Tierney GL, Faber-Langendoen D, Mitchell BR, Shriver WG, Gibbs JP (2009) Monitoring and evaluating the ecological integrity of forest ecosystems. Front Ecol Environ 7:308–316

Woodley S (2010) Ecological integrity and Canada’s national parks. George Wright Forum 27(2):151–160

Himalayan Biodiversity in the Face of Climate Change

14

G. C. S. Negi and Ranbeer S. Rawal

Abstract

The Himalayan mountains have emerged among the most sensitive ecosystems under the global climate change (CC) scenario. Impacts of CC are realized all across the physical, biological and socio-economic components of the ecosystems. Particularly, community distribution range of plant species and their phenologies are predicted to experience varying level of shifts across these gradients and, thus, act as potential indicators of CC. It has been anticipated that the richness of endemic species with restricted distribution and life support values (goods and services) of this global biodiversity hotspot is highly vulnerable under the changing climate. Unfortunately, poor availability of systematic long-term data sets from the region has severely limited our capability to objectively define the intensity of impacts on biodiversity and develop suitable conservation strategies to respond to the emerging challenges of climate change. This chapter provides an overview of research-based evidence of the impacts of CC on flora and fauna of the Himalayan region and provides some mitigation measures for biodiversity conservation.

Keywords
Climate change · Himalayan forests · Biodiversity conservation · Ecosystem services · Sacred forests

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14.1 Himalaya Is Sensitive to Climate Change

The Himalayan mountains are geologically young and geotectonically active, which make them most vulnerable to the impact of climate change, particularly the extreme rainfall events leading to soil erosion, landslides and flash floods. Temperature trends in most Himalayan regions substantially exceed the global mean increase of 0.85 °C (between 1880 and 2012), with winter season temperature trends being generally higher than those of other seasons (Schikhoff et al. 2015). The greatest average increase, 0.07 °C/year, is observed in winter, whereas the least increase, 0.03 °C/year, in summer (Shrestha et al. 2012); and the warming rate increases with altitude, peaking between 4800 and 6200 m altitudes (Wolfe 1979; Singh et al. 2011). Location-specific reports in the Himalayas show increases of atmospheric temperature, viz. 0.04 °C/year during 1962–2004 in Kullu Valley in the Western Himalayas (Kumar et al. 2009), 0.06 °C/year during 1977–2000 in Nepal in the central Himalayas (Shrestha et al. 1999) and 0.03 °C/year from 1985 to 2005 in Bhutan (Tsering 2003; Sharma and Tsering 2009), and Gangtok (Sikkim) in the Eastern Himalayas, across 1981–2010, recorded an average increase in minimum temperature (0.85–0.99 °C) and mean annual temperature (0.42–0.58 °C) per decade (Anonymous 2015). The IPCC predicts that average annual mean temperature over the Asian land mass, including the Himalayas, will increase by about 3 °C by the 2050s and about 5 °C by the 2080s (Kumar et al. 2006; IPCC 2007), and average annual precipitation will increase by 10–30% by 2080s (IPCC 2007). In contrast, Schickhoff et al. (2015) have reported that in this region precipitation patterns are spatio-temporally differentiated but show rather decreasing than increasing trends, in particular during summer. It has been stated that in the face of climate change most ecosystems and landscapes will be impacted through changes in species composition, productivity and biodiversity (Leemans and Eickhout 2004).The magnitude of potential response of climate-sensitive Himalayan ecosystems needs to be better understood (Chakraborty et al. 2016).

14.2 Biodiversity in the Himalayan Mountains

Mountains are remarkably diverse and globally important as centres of biological diversity. In Chap. 13 of Agenda 21, adopted at the United Nations Conference on Environment and Development (UNCED 1992), mountains are defined as "storehouses of biological diversity and endangered species". Mountains have been recognized as important ecosystems by the Convention on Biological Diversity. A programme of work on mountain biodiversity was specifically developed in 2004, which aimed at reducing the loss of mountain biological diversity at global, regional and national levels by 2010. This great wealth of biological diversity is attributed to the wide variety of environments in the mountains, particularly the Himalayas which is one among the 34 biodiversity hotspots of the globe. The Eastern Himalaya is one of the four biodiversity hotspots of India and known as the "centre of origin of cultivated plants", as over 50 important tropical and sub-tropical fruits, cereals and types of rice originated in the region (Chettri et al. 2010). In response to this recognition, the government of India

(GOI), under its protected area (PA) programme, has established 36 national parks, 123 wildlife sanctuaries and 7 biosphere reserves in the Indian Himalayan region (IHR), accounting for about 2% of the country's total geographical area (MoEF&CC, Govt. of India; http://envfor.nic.in). Approximately 15% of the area of the Himalayan Biodiversity Hotspot has some form of legal protection; this percentage comes down to 10% when only areas under the IUCN categories I to IV are considered.

The IHR harbours nearly 50% of the total flowering plants of India, of which 30% are endemic to the region. The IHR supports about 8000 species of angiosperms (40% endemics), 44 species of gymnosperms (15.91% endemics), 600 species of pteridophytes (25% endemics), 1737 species of bryophytes (32.53% endemics), 1159 species of lichens (11.22% endemics) and 6900 species of fungi (27.39% endemics) (Singh and Hajra 1996). In this region, about 1748 species with medicinal uses (Samant et al. 1998), 675 wild edibles (Samant and Dhar 1997), 279 fodder species (Samant et al. 1998), 155 sacred plants (Samant and Shreekar 2003) and 118 essential oil plants with medicinal values (Samant and Palni 2000) have been recorded. Of the total plants, the species richness is maximum in herbs (1020 spp.), followed by trees (339 spp.), shrubs (338 spp.) and pteridophytes (51 spp.). Using IUCN criteria, about 121 species have been recorded in the Red Data Book (RDB) of Indian plants from the IHR; of these, 17 are medicinal plants (Nayar and Shastry 1987, 1988 1990). Similarly, out of the total records from India, 65% of mammals, 50% of birds, 35% of reptiles, 36% of amphibians and 17% of fish are reported from the IHR (Ghosh 1997). Moreover, 29 out of 428 species of reptiles from India, 35 species of amphibians (out of 200) and 36 species of freshwater fish (out of 1300) are endemic to this region (Ghosh 1997).

The wealth of biodiversity of this region supports peoples' livelihood directly and indirectly through a range of ecosystem goods and services (Semwal et al. 2007; Joshi and Negi 2011). In this region, over 675 (67% of India) wild plant species are used by different communities for food/edible, medicine and other purposes. About 1743 species (23% of India) of medicinal value are found in the IHR that provide life-saving drugs (e.g. *Aconitum heterophyllum*, *A. balfourii*, *Gentiana kurroo, Picrorrhiza kurroa*, *Podophyllum hexandrum*, *Taxus baccata*, *Valeriana jatamansi*, etc.) and accrue immense economic wealth. Many medicinal plants in the IHR have multipurpose use and about 81 medicinal plant species are also the source of important fatty and essential oils used for edible as well as industrial purposes. People use the biodiversity in many ways for supporting livelihood. For example, an edible fungus (*Morchella esculenta*) found in the forests is collected by local people (average ~ 1.5 kg dry wt. per household and sold at Rs. 5000/kg) (Prasad et al. 2002). Similarly, kafal (*Myrica esculenta*) an edible fruit found in the forests is sold at Rs. 20–40/kg, earning approximately Rs. 1.4 million/year in the Kumaun region alone (Bhatt et al. 2000). The approximate value of carbon sequestration of Himalayan forest ecosystem has been estimated at Rs. 943 billion/year (Singh 2007). Inevitably, any change in the forest species composition would influence forestry, agriculture, livestock husbandry, NTFPs and medicinal plant-based livelihoods and many other intangible services of the forests and biodiversity. Thus, climate change will not only threaten the biodiversity but will also affect the socio-economic condition of the indigenous people (Bharali and Khan 2011).

14.3 Impact of Climate Change on Biodiversity: Research-Based Evidence

14.3.1 Impact on Flora

Plants may respond to climate change in four possible ways: (a) phenotypic plasticity enabling species survival, with alterations in ecophysiological processes, (b) evolutionary adaptation to new climate, (c) emigration to favourable habitats and (d) extinction (Bawa and Dayanandan 1998; Saxena and Purohit 1993). In spite of species range shifts reported worldwide (viz. alpine plants; Grabherr et al. 1994; butterflies; Parmesan 1996; Parmesan et al. 1999; birds; Thomas and Lennon 1999; marine invertebrates; Barry et al. 1995; and mosquitoes; Epstein et al. 1998), only a few systematic studies have been conducted to understand the impact of climate change on the flora of the IHR. Studies so far conducted reveal that (i) in the last decade the common Himalayan alder (*Alnus nepalensis*) which is found between 900 and 1800 m asl has reached up to 2400 m asl; (ii) invasive weeds such as *Eupatorium adenophorum* have moved up to 2800 m asl (Sekar et al. 2012). Similarly, the density of *Parthenium hysterophorus* has increased from 13.24 ind./m^2 to 17.43 ind./m^2 (Dogra et al. 2009, 2011). Spread of alien invasive species such as *Lantana* has already altered vast stretches of Indian forests, and now it is spreading rapidly in the Himalayan region beyond 1500 m asl (Negi et al. 2019); and (iii) in Arunachal Pradesh (Eastern Himalayas), which is rich in endangered, endemic and threatened floral and faunal species with restricted distribution and narrow habitat ranges, flora like *Amentotaxus assamica*, *Gleditsia assamica*, *Gymnocladus assamicus*, *Coptis teeta*, *Rhododendron* spp., *Paphiopedilum fairrieanum*, *Aneilema glanduliferum*, *Capparis pachyphylla*, *Acer oblongum* and *Acer sikkimense* are at risk due to climate change. The highland species are now facing the additional threat of warming temperatures as they have limited scope to move up further, such as many Rhododendron species in Arunachal Pradesh (Paul et al. 2010). The timberline ecotone, marked by the culmination of the forested zone, will advance to higher altitudes with increasing global temperature (Parmesan 2006; Weiser and Tuasz 2007). Studies in the Western Himalayas have recorded an upward shift of treeline species of 19 and 14 m over 10 years on south and north slopes, respectively (Dubey et al. 2003). The ability of species to respond to climate change will depend on their being able to “track” shifting climatic zones and colonize new territory or to adapt their physiology and seasonal behaviour to changing conditions (Menzel et al. 2006).

14.3.2 Impact on Fauna

With the warming and climate change pattern, many species of animals are migrating towards higher elevation. Warmer temperature, alteration in habitats and changed climatic pattern may alter animal’s reproductive strategies. Studies from Sikkim, Himalaya (Acharya and Chettri 2012), have reported that many species of birds such as blood pheasant, snow pigeon, rusty-bellied shortwing and white-winged redstart now occur well above their range of occurrence from around 1500–3000 m asl to now up to 3000–4000 m asl (Ali 1962). The cheer pheasants (*Catreus wallichii*), with a range primarily limited to the Jammu and Kashmir regions, are declining due to loss of the habitat (Wikramanayake et al. 1998). Similarly, in the NE region, narrowly endemic species like the Namdapha flying squirrel (*Biswamoyopterus biswasi*), takin (*Budorcas taxicolor*), white-winged wood duck, Sclater's monal (*Tragopan temminckii*), black-necked crane, Mishmi wren and Bengal florican face the risk of extinction. Among the fish fauna unique to the Himalayan Hotspot (IUCN 2004), the genus *Schizothorax* is represented by at least six endemic species in the high mountain lakes and streams, snowtrout, the genus *Ptychobarbus* and the Ladakh snowtrout (*Gymnocypris biswasi)*, a monotypic genus now thought to be extinct. Changes in the weather pattern influencing river flow in the Eastern Himalaya would disrupt the alluvial grasslands (e.g. *Saccharum spontaneum*) on which *Sus salvanius*, tiger (*Panthera tigris*) and the charismatic mega herbivore one-horned rhinoceros, critically endangered species, depend (Karanth and Nichols 1998). These species and many others, by virtue of their specific habits and habitat needs, are more vulnerable to climate change and more likely to face extinction risk (ICIMOD 2009).

14.4 Earlier Flowering in Trees Will Affect Pollination, Reproductive Success and Biodiversity

In the Western Himalayan mountains, early flowering in several members of Rosaceae (e.g. *Pyrus*, *Prunus* spp.) and *Rhododendron* has often been linked with global warming. Earlier onset of flowering in *Rhododendron* about 2 weeks as compared to the past has been reported in Kumaun Himalaya (Singh 2014; Shah et al. 2014). Generalized additive model (GAM) using real-time field observations (2009 to 2011) and herbarium records (1893–2003; Hooker 1849) predicted 88–97 days early flowering over the last 100 years in *Rhododendron arboreum* (Fig. 14.1;

Fig. 14.1 Early flowering (1 mon) in *Rhododendron* is a common example of impact of CC in the Himalaya

Table 14.1 Climate change impact on rhododendron flowering time

Species	Hooker (1849)	Current observations (2008–2010)
R. arboreum	Mar	Feb
R. capylocarpum	Jun	Apr-May
R. decipiens	May	Apr-May
R. edgeworthii	May-Jun	Apr-May
R. pendulum	May	Apr-May
R. thomsoni	Jun	Apr-Mid May
R. wightii	Jun	Apr-May

Source: FAR: KSLCDI-India, GBPNIHESD, Sikkim Unit, 2014

Table 14.1) due to the rise in annual mean maximum temperature in Western Himalaya (Gaira et al. 2011). Ranjitkar et al. (2014) have reported that tree *Rhododendron* might expand its distributional range in response to global warming under the recent trend of rising winter-spring temperature (Xu et al. 2009; INCCA 2010). Telwala et al. (2013) reported range shifts of 87% of the 124 endemic species as consequence of winter warming in the alpine zone of Sikkim Himalaya using historical climatic and taxonomic data from1849–1850 to 2007–2010. Similarly, significantly ($p < 0.01$) earlier flowering was recorded for 11 alpine/subalpine and 12 temperate high-value medicinal herbs as an impact of increasing winter temperature (Gaira 2011). In large cardamom (*Amomum subulatum* Roxb.), a perennial cash crop of the Eastern Himalaya, it was observed that the visits of pollinators (*Apis cerana* and *Bombus* sp.) were sunlight dependent and correlated positively with number of flowers and higher crop yield (i.e. 17–41 g/plant) (Gaira et al. 2015). Phenological studies in Kumaun Himalayan forests (sal, pine and oak forests) have revealed that the mean date of leafing and flowering has advanced by 1–2 weeks within a period of 30 years (1985–2015) due to the increase in temperature (0.005 °C/year) and decline in rainfall (3.3 mm/year) over the years (Singh and Negi 2016). In these species, a rise in temperature and water stress may advance seed maturation, which might break down the synchrony between monsoon rains and seed germination, thus reducing forest regeneration, taxonomic composition of forests and biodiversity (Singh et al. 2010). However, many of these studies suffer from the lack of long-term meteorological data to establish a clear-cut relationship with CC, and there is a pressing need for observational climatic data in the region.

14.5 Impact of Climate Change on Agro-Biodiversity

Mountain agriculture presents a rich repository of agro-biodiversity and resilience to crop diseases and changes in weather cycle will have a major effect on crop yield and food supply. For example, in Uttarakhand, over 40 different crops and hundreds of cultivars selected by farmers, comprising cereals, millets, pseudo-cereals, pulses and tuber crops, are cultivated (Maikhuri et al. 1997; Agnihotri and Palni 2007). Mixed cropping of 12 crops (Baranaja) is another best example of rich agri-diversity in this region (Ghosh and Dhyani 2004) to achieve food security. However, in the recent decades, the area under traditional crops has drastically declined (>60%), and many of the crops are at the brink of extinction, such as *Glycine* spp., *Hibiscus sabdariffa*, *Panicum miliaceum*, *Perilla fruitescens*, *Setaria italica*, *Vigna* spp., to name a few (Maikhuri et al. 2001; Negi and Joshi 2002). The Indian Agriculture Research Institute (IARI), New Delhi, has indicated that every rise of 1 °C temperature throughout the growth periods will incur a loss of 4–5 million tons in wheat production (Uprety and Reddy 2008). Deficit in food production in Kashmir region has reached 40% in 2007 from 23% in 1980–1981, which has been linked with CC (Sinha 2007). Alterations in the floral diversity due to land use and land cover change and extinction of local cultivars will also affect the population of pollinators. In HP, decline in honeybees due to land use changes, increasing monoculture and use of pesticides and herbicides have reduced apple production due to inadequate pollination (Partap and Partap 2003), compelling farmers to rent colonies of honeybees for pollinating the apple orchards at Rs. 500/colony (Ahmad et al. 2002) and use flower "bouquets" to attract pollinators. A decline in apple production in HP was associated with the increase in atmospheric temperature and decrease in snow fall (Box 14.1). A change in climatic conditions can cause a pest or disease to expand its normal range into a new environment, extending losses and affecting natural plant communities (Rosenzweig et al. 2001).

Box 14.1: Shifting Apple Cultivation to Higher Altitudes in HP

In Himachal Pradesh as an adaptive measure to global warming, people are now planting new apple orchards towards higher altitudes due to inadequate chilling at lower altitudes. Early snow (December to early January) is preferred for its favourable effect in apples as it provides a chilling period of about 10 weeks below 5 °C, which is necessary for bud break in springtime (Abbott 1984). In the Kullu Valley (HP), rainfall decreased by about 7 cm and snowfall by about 12 cm, and the mean minimum and maximum temperatures increased by 0.25–1 °C, respectively, in the 1990s as compared to the 1880s that has reduced apple production (Fig. 14.2; Vishvakarma et al. 2003; Vedwan and Rhoades 2001).

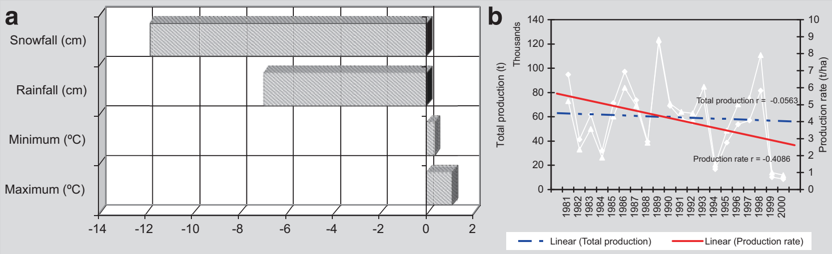

Fig. 14.2 (**a**) Changes in temperature and precipitation in the 1990s as compared to the 1980s in Kullu Valley (HP). (**b**) Reduced apple yield. (Adapted from Vishvakarma et al. 2003)

14.6 Threats to Himalayan Biodiversity

Climate change is likely to have certain impacts at levels from species to ecosystems that are not yet fully understood. The compounding effect of anthropogenic activities and CC together has aggravated the problem further (Kumar and Chopra 2009). The life cycles of many wild and domestic plants and animals are closely linked to the seasonal rhythms driven by the climate. Warming in the Himalayan region suggests moderate to large-scale shifts in vegetation types, with implications for forest dieback and biodiversity (Ravindranath and Sukumar 1998), change in the timing of phenological events of plants, changes in species abundance and range, shifts in habitat, etc. In the viviparous tree species of Himalayan forests, CC would have an impact on forest regeneration and community structure due to mistimed seed maturation and rainfall (Singh 2014). Spread of alien invasive species such as *Lantana*, *Eupatorium* and *Parthenium* spp. in the natural forests is already outcompeting the native biodiversity (Sharma et al. 2009). Increased incidences of forest fire are another major threat to biodiversity, which has a direct bearing on rise in atmospheric temperature and decline in rainfall (Singh et al. 2016). For example, in one of the most devastating forest fires in May, 1995, in Uttarakhand, a total of 2115 km^2 was burnt (Semwal and Mehta 1996), which was found to be associated with high air temperature (Sharma and Rikhari 1997). It is expected that with the increase in drought cycles and concomitant increase in forest fires, the pine (*Pinus roxburghii*) forests will encroach upon the area under oak (*Quercus* spp.) forests and also reduce the yield of non-timber forest products. Forest structure and functioning are likely to change as gaps created by fire are invaded by exotic species such as *Cassia tora*, *C. occidentalis*, *Eupatorium glandulosum*, *Imperata cylindrica*, *Lantana camara*, *L. indica*, *Parthenium hysterophorus*, etc., which would deplete biodiversity through allelopathic pathways (Parashar and Biswas 2003). A rise in temperature and water stress may advance seed maturation, which might result in the breakdown of synchrony between monsoon rains, vivipary and seed germination in major forest trees such as *Shorea robusta*, *Quercus floribunda* and *Q. semecarpifolia* (Singh et al. 2010). In the alpine area, growth and life cycle of vegetation is closely tied with snow melt (Negi et al. 1992), and their life cycle will be disturbed because of reduced snow melt water. The impacts of CC on forest ecosystems include shifts in the latitude of forest boundaries and the upward movement of

Box 14.2: Threats to Himalayan Biodiversity

- Global climate change.
- Alteration of habitats and land use change.
- Land fragmentation due to road construction and infrastructure.
- Human-induced changes such as biomass harvesting, deforestation, forest fire, livestock grazing and agricultural expansion into forest lands.
- Illegal trade of timber and medicinal and aromatic plants.
- Indiscriminate use of chemicals and fertilizers.

tree line to higher elevations; changes in species composition and in vegetation types; and an increase in net primary productivity (Ramakrishna et al. 2003). Inevitably, any change in the forest (distribution, density and species composition) under CC would immensely influence economies like forestry, agriculture, livestock husbandry, NTFPs and medicinal plant-based livelihoods and many others. An indicative list of threats to Himalayan biodiversity is given (Box 14.2).

14.7 Dev Vans: A Rich Tradition in Uttarakhand for Biodiversity Conservation

In Uttarakhand, a practice of conserving forests and sacred groves on the ground of religious belief by the traditional communities has been a part of rich tradition for generations (Khan and Tripathi 2004; Anthwal et al. 2006; Pala et al. 2013)**.** Village communities protect their forests by declaring them as sacred forests and offer the forest patches to local god/goddess forever or for a fixed duration of several years. These patches of forests as a rule are treated piously and no one is allowed to cut trees, graze cattle, or in some cases even has to enter barefoot within them (Kandari et al. 2014). However, in some rituals, special sanction to use fallen wood and dried trees from the sacred forests is made to the community (Pande and Negi 2004). These sacred forests, groves, water sources or even landscapes, referred to as sacred natural sites (SNS), present the best example of community participation in biodiversity conservation (Vipat and Bharucha 2014) and serve as repository of many endangered and rare species of flora and fauna (Khan and Tripathi 2004; Sukumaran and Jeeva 2008; Ray and Ramachandra 2010; Singh et al. 2010). In India, around 100,000–150,000 sacred groves are reported (Malhotra et al. 2001), and they are named differently in various parts such as *Dev Vans* in Uttarakhand (Singh et al. 2010; Singh et al. 2014). In the recent years, these sacred groves are increasingly looked upon for biodiversity conservation and carbon stocking (Kemeuze et al. 2015). In a case study on 6 sacred forests and 2 sacred groves in Pithoragarh district of Kumaun Himalaya (1541–1948 m asl) carried out by us, species richness among tree species was found up to 12 and the Shannon-Weiner diversity index (H-index) was found having a range of 0.36–1.66. Among the shrub species, the species richness and Shannon-Weiner diversity index were found to have a range of 4–8 and 0.55–1.44, respectively. Average soil organic carbon (SOC %) was found ranging from 0.35 ± 0.22% to 1.14 ± 0.39% across these forests. Total C-stock (above- + belowground biomass + soil) was estimated having a range of 130–748 t/ha across the forests (Table 14.2). The forest patches smaller in size (sacred groves) offered to local goddess recorded the maximum stocks of C but low in tree diversity. The sacred forests, larger in size, recorded greater number of tree species but low C-stock. The mean C density in a sacred forest of this region has been reported as 775 t/ha (Pala et al. 2013). The total C-stock in the sacred forests reported by us was found much higher than the 29–108 t/ha reported by Kumar and Sharma (2015) and the 59–245 t/ha reported by Sharma et al. (2010) for reserved forests of this region. Also, the Shannon-Weiner diversity

Table 14.2 Carbon stock (t/ha) in vegetation and soil pool across 8 sacred forests in Uttarakhand

Sacred forest (SF)/sacred grove (SG)	C-stock (vegetation)	C-stock (soil)	C-stock total (soil + vegetation)
Dharapani SF	137.3	81.6	218.9
Dhungabhool SF	227.3	52.7	280.0
Chhanapande SF	106.5	23.3	129.8
Chitgal SF	124.1	43.9	168.0
Jajut SF	178.9	36.7	215.6
Uprara SF	104.9	37.1	142.1
Chamunda Devi SG	660.5	87.2	747.7
Mahakali Mandir SG	288.9	72.1	361.0

Source: Negi 2015

index we found in our study was greater than several forests of this region (0.28–1.20; Sharma et al. 2010).

14.8 Mitigation and Adaptation Measures

The awareness that mountain regions are an important component of the earth's ecosystems, in terms of the resources and services that they provide to both mountain communities and lowland residents, has risen in the last decade. Therefore, ensuring conservation of biodiversity has become one of mankind's important responsibilities. Some management options to mitigate climate change impact on biodiversity are presented below:

(i) Increasing forested area and restoration of barren and degraded forest areas to maximize the resilience of the species and ecosystems to climate change.
(ii) Increasing protected areas network within the region and maintenance of corridors/connectivity of the protected areas and the natural landscapes to provide opportunities for fauna to adapt to climate change by migration.
(iii) Prioritizing area/habitats/species and ecosystems with high endemism for conservation using both in situ and ex situ mechanisms.
(iv) Promoting utilization of traditional knowledge and its integration into formal climate change mitigation and adaptation strategies for biodiversity conservation, such as sacred groves, forests and landscapes.
(v) Citizen's science approach for conservation and creating environmental awareness of climate change impacts biodiversity among the people, especially the students.

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References

Abbott DL (1984) The apple tree: physiology and management. Growers Association, Kullu Valley, H.P

Acharya BK, Chettri B (2012) Effect of climate change on birds, herpetofauna and butterflies in Sikkim Himalaya: a preliminary investigation. In: Arrawatia ML, Tambe S (eds) Climate change in Sikkim patterns, impacts and initiatives. Information and Public Relations Department, Government of Sikkim, Gangtok

Agnihotri RK, Palni LMS (2007) On farm conservation of landraces of rice (*Oryza sativa* L.) through cultivation in the Kumaun region of Indian, central Himalaya. J Moun Sci 4:354–360

Ahmad F, Partap U, Joshi SR, Gurung MB (2002) Please do not steal our honey. Bees for Dev J 64:9

Ali S (1962) The birds of Sikkim. Oxford University Press, New Delhi, p 414

Anonymous (2015) Feasibility assessment report: India: Khangchendzonga landscape conservation and development initiative –India. G.B. Pant Institute of Himalayan Environment & Development, Sikkim Unit, Pangthang-Gangtok, p 128

Anthwal A, Sharma RC, Sharma A (2006) Sacred groves: traditional way of conserving plant diversity in Garhwal Himalaya, Uttaranchal. J Am Sci 2(2):35–38

Barry J, Baxter C, Sagarin R, Gilman S (1995) Climate related, long-term faunal changes in a Californian rocky intertidal community. Science 267:672–675

Bawa KS, Dayanandan S (1998) Global climate change and tropical forest genetic resources. Climate Change 39:473–485

Bharali S, Khan ML (2011) Climate change and its impact on biodiversity; some management options for mitigation in Arunachal Pradesh. Curr Sci 101(7):855–860

Bhatt ID, Rawal RS, Dhar U (2000) The availability, fruit yield, and harvest of *Myrica esculenta* in Kumaun (west Himalaya), India. Mount Res Dev 20(2):146–154

Chakraborty A, Sachdeva K, Joshi PK (2016) Mapping long-term land use and land cover change in the central Himalayan region using a tree-based ensemble classification approach. Appl Geogr 74:136–150

Chettri N, Sharma E, Shakya B, Thapa R, Bajracharya B, Uddin K, Oli KP, Choudhury D (2010) Biodiversity in the Eastern Himalayas: Status, Trends and Vulnerability to Climate Change – ICIMOD, Kathmandu, Nepal, p 23

Dogra KS, Sood SK, Sharma R (2011) Distribution, biology and ecology of *Parthenium hysterophorus* L.(congress grass) an invasive species in the North-Western Indian Himalaya (Himachal Pradesh). African J Plant Sci 5(11):682–687

Dogra KS, Kohl RK, Sood SK (2009) An assessment and impact of three invasive species in the Shivalik hills of Himachal Pradesh, India. Int J Biodivers Conserv 1(1):4–10

Dubey B, Yadav R, Singh J, Chaturvedi R (2003) Upward shift of Himalayan pine in Western Himalaya. India Curr Sci 85:1135–1136

Epstein P, Diaz H, Elias S, Grabherr G, Graham N, Martens W, Thompson EM, Susskind J (1998) Biological and physical signs of climate change: focus on mosquito borne diseases. Bull Am Meteorol Soc 79:409–417

Gaira KS (2011) Application of statistical tools for analysing phenological changes over time series among selected herbaceous species of the Himalaya -a herbarium study. PhD thesis, Garhwal University, Srinagar (Garhwal), Uttarakhand, India, p 132

Gaira KS, Dhar U, Belwal OK (2011) Potential of herbarium records to sequence phenological pattern – a case study of *Aconitum heterophyllum* wall. in the Himalaya. Biodivers Conserv 20:2201–2210

Gaira KS, Rawal RS, Singh KK (2015) Variation in pollinator density and impact on large cardamom (*Amomum subulatum* Roxb.) crop yield in Sikkim Himalaya. Ind J Asia-Pacific Biodiversity 9:17–21

Ghosh AK (1997) Himalayan fauna with special reference to endangered and endemic species. In: Dhar U (ed) Himalayan biodiversity: action plan. GB Pant Institute of Himalayan Environment & Development, Almora, pp 53–59

Ghosh P, Dhyani PP (2004) Baranaaja: the traditional mixed cropping system of central Himalaya. Outlook on Agric 33:261–266

Grabherr G, Gottfried M, Pauli H (1994) Climate effects on mountain plants. Nature 369:448

Hooker JD (1849) The rhododendrons of Sikkim-Himalaya. Bishen Singh Mahendra Pal Singh, Dehra Dun

INCCA (2010) Assessment of the Greenhouse Gas (GHG) Emission (2007) Indian Network for Climate Change Assessment (INCCA), The Ministry of Environment & Forests, Govt. of India

IPCC (2007) In: Solomon S, Qin D, Manning M, Chen Z, Marquis M et al (eds) Contribution of working group I to the fourth assessment report of the intergovernmental panel on climate change. Cambridge Univ Press, Cambridge, UK

Joshi G, Negi GCS (2011) Quantification and valuation of forest ecosystem services in the western Himalayan region, India. Int J Biodivers Sci Ecosyst Serv Manage 7(1):2–11

Kandari LS, Bisht VK, Bhardwaj M, Thakur AK (2014) Conservation and management of sacred groves, myths and beliefs of tribal communities: a case study from North-India. Environ Syst Res 3:16

Karanth KU, Nichols JD (1998) Estimation of tiger densities in India using photographic captures and recaptures. Ecology 79(8):2852–2862

Kemeuze VA, Mapongmetsem M, Sonwa DJ, Fongnzossie EF, Nkongmeneck BA (2015) Plant diversity and carbon stock in sacred groves of semi-arid areas of Cameroon: case study of Mandara Mountains. Int J Environ 4(2):308–318

Khan ML, Tripathi RS (2004) Sacred groves of Manipur- ideal centers for biodiversity conservation. Curr Sci 87(4):430–433

Kumar V, Chopra AK (2009) Impact of climate change on biodiversity of India with special reference to Himalayan region- an overview. J Appl Nat Sci 1(1):117–122

Kumar A, Sharma MP (2015) Estimation of carbon stocks of Balganga Reserved Forest, Uttarakhand, India. For Sci Technol 11(4). https://doi.org/10.1080/21580103.2014.990060

Kumar KR, Sahai AK, Kumar KK, Patwardhan SK, Mishra PK et al (2006) High resolution climate change scenario for India for the 21st century. Curr Sci 90:334–345

Kumar P, Yadav S, Sharma S, Lal S, Jha D (2009) Impact of climate change on seed production of cabbage in north western Himalayas. W J Agric Sci 5:18–26

ICIMOD (2009) Climate change impact and vulnerability in the Eastern Himalayas, Synthesis Report

IUCN (2004) The IUCN red list of threatened species. The world Conservation Union, Gland

Leemans R, Eickhout B (2004) Another reason for concern: regional and global impacts on ecosystems for different levels of climate change. Global Env Change 14(3):219–228

Maikhuri RK, Rao KS, Semwal RL (2001) Changing scenario of Himalayan agroecosystems: loss of agrobiodiversity, an indicator of environmental change in Central Himalaya, India. *Environmentalist* 21:23–39

Maikhuri RK, Semwal RL, Rao KS, Nautiyal S, Saxena KG (1997) Eroding traditional crop diversity imperils the sustainability of agricultural systems in central Himalaya. Curr Sci 73:777–782

Malhotra KC, Gogkhle Y, Chatterjee S, Srivastava S (2001) Cultural and Ecological Dimensions of Sacred Groves in India. India National Science Academy and Indira Gandhi Rashtriya Manav Sangrahalaya, Bhopal

Menzel A, Sparks TH, Estrella N, Koch E (2006) European phenological response to climate change matches the warming pattern. Glob Chang Biol 12:1969–1976

Nayar MP, Shastry ARK (1987/1988/1990) Red book of Indian plants, Vol. I/II/III, Botanical Survey of India, Calcutta

Negi GCS, Rikhari HC, Singh SP (1992) Phenological features in relation to growth form and biomass accumulation in an alpine meadow of the Central Himalaya. Vegetatio 101:161–170

Negi GCS (2015) Managing ecosystems for sustaining services. Final Technical Report submitted to Kailash Sacred Landscape Development Initiative Project, GBPIHED, Almora

Negi GCS, Joshi V (2002) Studies in the western Himalayan micro-watersheds for global change impact assessment and sustainable development. In: Shrestha KL (ed) Global Change and Himalayan Mountains. Institute for Development and Innovation, Kathmandu, pp 153–165

Negi GC, Sharma S, Vishvakarma SC, Samant SS, Maikhuri RK, Prasad RC, Palni LM (2019) Ecology and use of *Lantana camara* in India. Bot Rev 85(2):109–130

Pala NA, Negi AK, Gokhale Y, Aziem S, Vikrant KK, Todaria NP (2013) Carbon stock estimation for tree species of SemMukhem sacred forest in Garhwal Himalaya, India. J For Res 24:457. https://doi.org/10.1007/s11676-013-0341-1

Pande G, Negi GCS (2004) Dev vans: a cultural way of nature conservation in Uttaranchal. Hima-Paryavaran 16(1):7–8

Parashar A, Biswas S (2003) The impact of forest fire on forest biodiversity in the Indian Himalayas (Uttaranchal). XII World Forestry Congress, 358 B-1

Parmesan C (1996) Climate and species range. Nature 382:765–766

Parmesan C (2006) Ecological and evolutionary responses to recent climate change. Ann Rev Ecol 37:637–669

Parmesan C et al (1999) Poleward shifts in geographical ranges of butterfly species associated with regional warming. Nature 399:579–583

Partap U, Partap T (2003) Warning signals from the apple valleys of the Hindu Kush - Himalayas: productivity concerns and pollination problems. ICIMOD, Kathmandu

Prasad P, Chauhan K, Kandari LS, Maikhuri RK, Purohit A, Bhatt RP, Rao KS (2002) *Morchella esculenta* (Guchhi): need for scientific intervention for its cultivation in central Himalaya. Curr Sci 82(9):1098–1100

Paul A, Khan ML, Das AK, Dutta PK (2010) Diversity and distribution of rhododendrons in Arunachal Himalaya, India. J Am Rhod Soc 64:200–205

Ramakrishna RN, Keeling CD, Hashimoto H, Jolly WM, Piper SC, Tukder CJ, Myneni& RB, Running SW (2003) Climate-driven increases in global terrestrial net primary production form 1982 to 1999. Science 300:1560–1563

Ranjitkar S, Kindt R, Sujakhu NM, Hart R, Guo W, Yang X, Shrestha KK, Xu J, Luedeling E (2014) Separation of the bioclimatic spaces of Himalayan tree rhododendron species predicted by ensemble suitability models. Glob Ecol Conserv 1:2–12

Ray R, Ramachandra TV (2010) Small sacred groves in local landscape: are they really for conservation. Curr Sci 98(9):1178–1180

Ravindranath NH, Sukumar R (1998) Climatic change and tropical forests in India. Clim Chang 39:563–581

Rosenzweig C, Iglesias A, Yang XB, Epstein PR, Chivian E (2001) Climate change and extreme weather events: implications for food production, plant diseases, and pests. Global Change & Human Health 2:90–104

Samant SS, Dhar U (1997) Diversity, endemism and economic potential of wild edible plants of Indian Himalaya. Int J Sust Dev World 4:179–191

Samant SS, Shreekar P (2003) Diversity, distribution pattern and traditional knowledge of sacred plants in Indian Himalayan region. Ind J For 26(3):222–234

Samant SS, Dhar U, Palni LMS (1998) Medicinal plants of Indian Himalaya: diversity distribution potential values. Gyanodaya Prakashan, Nainital

Samant SS, Palni LMS (2000) Diversity, distribution and indigenous uses of essential oil yielding medicinal plants of Indian Himalayan region. J Med Arom Plant Sci 22:671–684

Saxena KG, Purohit AN (1993) Greenhouse effect and Himalayan ecosystems. In: Narain P (ed) Proceedings of first agricultural science congress-1992. National Academy of Agricultural Sciences, Indian Agricultural Research Institute, New Delhi, pp 83–93

Schickhoff U, Bobrowski M, Böhner J, Bürzle B, Chaudhary RP, Gerlitz L, Heyken H, Lange J, Müller M, Scholten T, Schwab N, Wedegärtner R (2015) Do Himalayan treelines respond to recent climate change? An evaluation of sensitivity indicators. Earth Syst Dynam 6:245–265

Sekar KC, Manikandan R, Srivastava SK (2012) Invasive alien plants of Uttarakhand Himalaya. Proc Natl Acad Sci 82(3):375–383

Semwal RL, Mehta JP (1996) Ecology of forest fires in Chir pine forests of Garhwal Himalayas. Curr Sci 70:426–427

Semwal RL, Tewari A, Negi GCS, Thadani R, Phartiyal P, (Editors & Research Contributors), Verma M, Joshi S, Godbole G, Singh A (Research Contributors) (eds) (2007) Valuation of ecosystem services and forest governance: a scoping study from Uttarakhand. Leadership in Environment & Development, LEAD-India Publication, New Delhi

Shah S, Verma A, Tewari A (2014) Timing of shifts in phenological events in *Rhododendron arboreum* smith. Influenced by climatic irregularities in Kumaun regions of central Himalaya. Global J Sci Res 2(2):56–59

Sharma S, Rikhari HC (1997) Forest fire in the central Himalaya: climate and recovery of trees. Int J Biometeorol 40:63–70

Sharma E, Tsering K (2009) Climate change in the Himalayas: the vulnerability of biodiversity. Sustainable Mount Dev 55:10–12

Sharma P, Singh P, Tiwari AK (2009) Effects of *Lantana camara* invasion on plant biodiversity and soil erosion in a forest watershed in Lower Himalayas, India. Indian J For 32(3):369–374

Sharma CM, Baduni NP, Gairola S, Ghildiyal SK, SK SSS (2010) Tree diversity and carbon stocks of some major forest types of Garhwal Himalaya, India. For Ecol Manag 260:2170–2179

Shrestha A, Wake C, Mayewski P, Dibb J (1999) Maximum temperature trends in the Himalaya and its vicinity: an analysis based on temperature records from Nepal for the period 1971–1994. J Clim 12:2775–2787

Shrestha UB, Gautam S, Bawa KS (2012) Widespread climate change in the Himalayas and associated changes in local ecosystems. PLoS One 7(5):e36741

Singh DK, Hajra PK (1996) Floristic diversity. In: Gujral GS, Sharma V (eds) In: changing perspectives of biodiversity status in the Himalaya. British Council, New Delhi, pp 23–38

Singh SP (2007) Himalayan forest ecosystem services. Central Himalayan Environment Association, Nainital, Uttarakhand, p 53

Singh H, Husain T, Agnihotri P (2010) Haat Kali sacred grove, Central Himalaya, Uttarakhand. Curr Sci 98(3):290

Singh N (2014) Flowering phenology of tree *rhododendron arboreun* along an elevation gradient in different sites of Kumaun Himalayas. Int J Sci & Nature 5(3):572–576

Singh SP, Singh V, Skutsch M (2010) Rapid warming in the Himalayas: ecosystem responses and development options. Climate & Dev 2:1–13

Singh SP, Bassignana-Khadka I, Karky BS, Sharma E (2011) Climate change in the Hindu Kush-Himalayas: the state of current knowledge. ICIMOD, Kathmandu

Singh H, HusainT AP, PandePC KS (2014) An ethnobotanical study of medicinal plants used in sacred groves of Kumaon Himalaya, Uttarakhand, India. J Ethnopharmacol 154(1):98–108

Singh P, Negi GCS (2016) Impact of climate change on phenological responses of major forest trees of Kumaun Himalaya. ENVIS Cent Himal Ecol 24:112–116

Singh RD, Gumber S, Tewari P, Singh SP (2016) Nature of forest fires in Uttarakhand: frequency, size and seasonal patterns in relation to pre-monsoonal environment. Curr Sci 111(2):398–403

Sinha S (2007) Impact of climate change in the highland agroecological region of India. Sahara Time Magazine. http://www.saharatime.com/Newsdetail.aspx?newsid=2659

Sukumaran S, Jeeva S (2008) A floristic study on miniature sacred forests at Agastheeshwaram, southern peninsular India. Eura J Biosci 2:66–78

Telwala Y, Brook BW, Manish K, Pandit MK (2013) Climate-induced elevational range shifts and increase in plant species richness in a Himalayan biodiversity epicentre. PLoS One 8(2):e57103

Thomas CD, Lennon JJ (1999) Birds extend their ranges northwards. Nature 399:213

Tsering K (2003) Constructing future climate scenarios of Bhutan. In: Project report on climate change vulnerability and adaptation study for rice production in Bhutan; climate change studies in Bhutan. Ministry of Agriculture, Bhutan

UNCED (1992) United Nations conference on environment & development, 1992

Uprety DC, Reddy VR (eds) (2008) Rising atmospheric carbon dioxide and crops. Indian Council of Agricultural Research, New Delhi

Vedwan N, Rhoades RE (2001) Climate change in the western Himalayas of India: a study of local perception and response. Clim Res 19:109–117

Vipat A, Bharucha E (2014) Sacred groves: the consequence of traditional management. J Anthropol 2014:1. https://doi.org/10.1155/2014/595314

Vishvakarma SCR, Kuniyal JC, Rao KS (2003) Climate change and its impact on apple cropping in Kullu Valley, north-west Himalaya, India. In: 7th international symposium on temperate zone fruits in the tropics and subtropics, pp 14–18, October, Nauni-Solan (H.P.)

Weiser C, Tuasz M (2007) Synopsis, 219-223 p. In: Wieser G, Tuasz M (eds) Trees at their upper limit: tree life limitations at the alpine tree line. Springer, Dordrecht

Wikramanayake E, Dinerstein E, Allnut T, Oucks C, Wettengel W (1998) A biodiversity assessment and gap analysis of the Himalayas. World Wildlife Fund-U.S., Conservation Science Program/UNDP Report

Wolfe JA (1979) Temperature parameters of humid and mesic forests of eastern Asia and relation to forests of other regions of the Northern hemisphere and Australasia. US Geol. Survey Prof. Paper 1106

Xu J, Grumbine RE, Shrestha A, Eriksson M, Yang X, Wang YUN, Wilkes A (2009) The melting Himalayas: cascading effects of climate change on water, biodiversity, and livelihoods. Cons Bio 23(3):520–530

Phenological Pattern and Community Structure of Dipterocarpaceae in Bukit Barisan Selatan National Park, Lampung

15

Diny Hartiningtias, Noviar Andayani, and Nurul L. Winarni

Abstract

Most of the tropical lowland dipterocarp forests in Indonesia have been fragmented and isolated due to excessive logging and forest fire. In addition, the growth of dipterocarp is generally slow. If these conditions persist, it will eventually decrease the dipterocarp's population. Information on phenological patterns and community structure are one of the basic pieces of knowledge for conservation management. We conducted research to answer these questions: what was the phenological pattern and population dynamics of Dipterocarpaceae, and to what extent did phenological pattern affect population dynamics of Dipterocarpaceae?

We found that the dipterocarps in Bukit Barisan National Parks show sub-annual flowering seasons, in which flowering occurs more than one cycle per year. It is different from the common dipterocarps' phenological pattern that usually has super-annual pattern or mass flowering. The pattern of the flowering and fruiting season in dipterocarp phenology begins with the emergence of new leaves in March, flowering in April, and ends with the appearance of the fruit in May or 3–5 months later.

There are 11 dipterocarps species in the study area; 3 of them are critically endangered, 2 are endangered, 1 is vulnerable, and the others are not listed in the

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IUCN Red List. Dipterocarps' populations were dominated by *Vaticaobovata*, *Dipterocarpushumeratus*, and *D. haseltii*. However, even though mass flowering and fruiting season are believed to represent an evolutionary adaptation of flowering and fruiting patterns in plants to face high mortality, the phenological pattern does not seem to have affected the population dynamics of dipterocarps.

Keywords
Bukit Barisan Selatan National Park · Community structure · Dipterocarpaceae · Phenology analysis

15.1 Introduction

Dipterocarpaceae is one of the important plant families in the tropics. Economically, Dipterocarpaceae is used as major source of hardwood timber in international trade (Ashton 1988), while ecologically the dipterocarps play an important role as the main constituent of plant and emergent species in lowland tropical forests (Ashton, et al. 1998; Whitmore 1990; Ghazoul and Sheil 2010), which affect the distribution of rainfall, light (Poppenborg and Holscher 2009), and microclimate for the plants beneath (Cadelus and Chazdon 2005).

Most of the tropical lowland dipterocarp forests have been fragmented and isolated due to excessive logging and forest fire. In addition, the growth of dipterocarps is generally slow. If the condition persists, it will eventually decrease the population (Appanah and Turnbull 1998; Purwaningsih 2004). Nowadays, many dipterocarp species have been listed in IUCN Red List as endangered. Therefore, appropriate conservation management is necessary for the survival of tropical lowland dipterocarp forest. Some basic knowledge that can be used in conservation management of dipterocarp is information on phenological patterns and community structure.

Phenology is the study of the life cycle of an organism, such as the growth of new buds, leaf growth, flowering season, fruiting season, fertilization, and seed dispersal (Newstorm, et al. 1994; Fenner 1998; Morison and Morcroft 2006). Dipterocarpacae generally experienced the flowering season every 3–5 years or so-called mass flowering season (Brearley, et al. 2007). Frequency of mass flowering and fruiting season is influenced by environmental factors, which can vary depending on the location.

Community structure is the study of the interaction between populations of a community (Molles 2010). Community structure can be in the form of species composition, diversity, and density. Knowledge of community structure can also be used to find out the source of the food available for wildlife, of timber availability, the quality of wildlife habitat, and so forth (Molles 2010).

The research aimed to understand the phenological pattern of dipterocarp species and community structure WCRS, as well as to understand the effect of phenological pattern on community structure of dipterocarps in Way Canguk Research Station.

15.2 Study Area Methods

One of the lowland dipterocarp rainforests that remained on the island of Sumatra is located in Bukit Barisan Selatan National Park (BBSNP) (O'Brien and Kinnard 1996; PHKA 2003). The area is increasingly threatened by land conversion to coffee plantations, agricultural areas, and the threat of deforestation (PHKA 2003). The study was conducted at the Way Canguk Research Station (WCRS), which is located in the south of the Bukit Barisan National Park (Fig. 15.1). WCRS is managed by the Ministry of Forestry Republic of Indonesia and the Wildlife Conservation Society – Indonesian Program (WCS – IP).

Based on preliminary observations, 11 dipterocarp species exist in WCRS. Three species are critically endangered, two are endangered, one is vulnerable, and the rest have not been listed on the IUCN Red List (Table 15.1).

Phenological data provided by WCS – IP was collected since February 1998 at 100 phenology plots measuring 50 × 10 m. Phenological data was taken using binoculars. Data collected consists of an estimated score of flowers and fruits, estimated number of fruits, the percentage of ripe fruit, and the presence or absence of a new leaf. Scoring of flowering and fruiting follows: score 0, no flowers/fruit; score of 1, there are 1–25% flowers/fruit in the canopy; score of 2, there are 26–50% flowers/fruit in the canopy; score of 3, there are 51–75% flowers/fruit in the canopy; and a score of 4, there are 76–100% flowers/ fruit in the canopy. Data were analyzed in the form of a diagram.

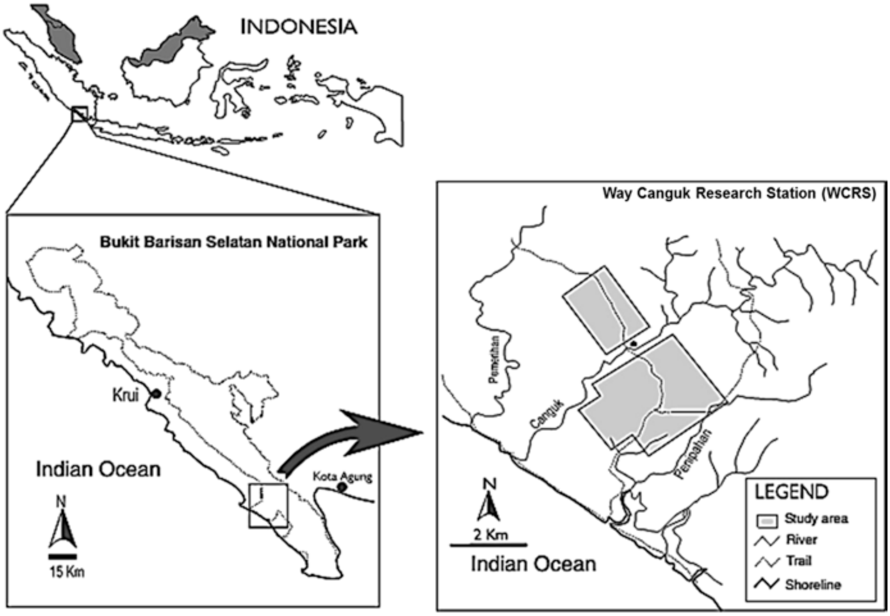

Fig. 15.1 Location of Bukit Barisan National Park and Way Canguk Research Center (WCS-IP, 2001)

Table 15.1 Eleven dipterocarps species in WCRS and its conservation status

No	Species	Conservation status (IUCN 2012)
1	*Anisoptera costata*	Endangered
2	*Dipterocarpus gracilis*	Critically endangered
3	*D. hasseltii*	–
4	*D. humeratus*	–
5	*D. Palembanicus*	–
6	*D. retusus*	Vulnerable
7	*Hopea sangal*	Critically endangered
8	*Shorea javanica*	–
9	*S. ovalis*	–
10	*S. Ovate*	Endangered
11	*Vatica obovata*	Critically endangered

The community structure data collection was conducted between July until October 2012. It was performed using quadrat transect methods. Along the transect line, we created plots of 200 × 20 m. The distance of each transects was 200 m. In total, the entire transect was 108 plots. The data collected in each plot were the total number of sapling, poles, and trees from each species.

The data were analyzed using the Importance Value Index (IVI). IVI is the sum of relative dominance, relative density, and relative frequency. IVI calculations were performed on every species on every stage of growth (saplings, poles, and trees). The IVI magnitude indicates the role of the relevant species in a community or research sites. The effect of phenological pattern on community structure was analyzed by comparing the pattern of the flowering season with the community structures.

15.3 Results and Discussion

15.3.1 Phenology

Dipterocarps usually show super-annual phenological pattern, which is known as mass flowering (Brearley et al. 2007). However, in WCRS, dipterocarps mostly had a sub-annual flowering season, in which flowering occurs more than one cycle per year. The first cycle occurs throughout the year. Peak flowering season in this cycle are referred as minor peaks. In several years, there are second cycles that have higher peak flowering season than the minor peaks and are referred as the major peaks. Peak flowering season is clearly visible on the graph percentage of flowering trees (Fig. 15.2).

In WCRS, major peaks occurred five times, in October 1999, November 2002 (20.2%), September 2006 (21%), during September to October 2008 (16.67%), and 20.3% during October 2011 (Fig. 15.2). The major peaks indicated that the peak of the second cycle of the dipterocarp occurred within the last 3 months of each year (Fig. 15.2). Flowering percentage was calculated by the number of flowering trees compared with the rest of dipterocarp trees in WCRS.

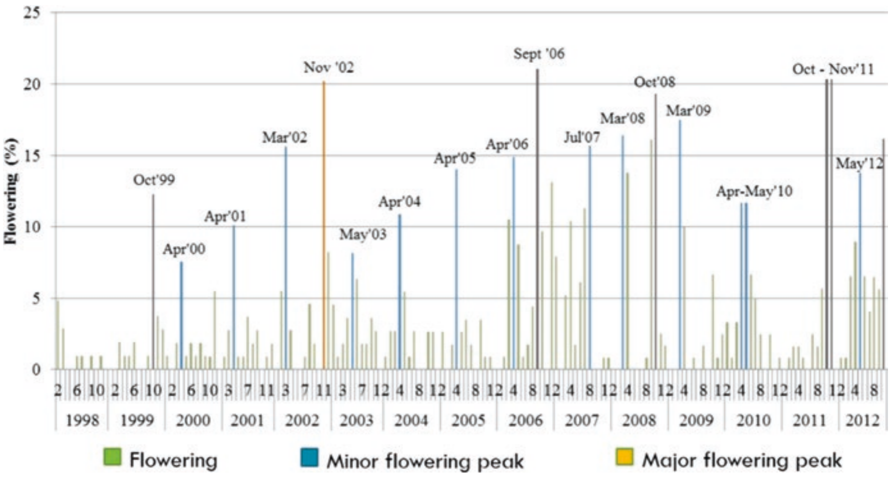

Fig. 15.2 Percentage of flowering trees in WCRS

The maximum percentage of the major flowering peak in WCRS ranged between 16.67% and 21% (Fig. 15.2). This value was quite low compared to the major peak season that occurs in a mass flowering at Barito Ulu, Central Kalimantan, which ranges between 46 and 68% (Brearley et al. 2007). Brearley et al. (2007) also mentioned that the percentage of flowering trees in the mass flowering season occurred on as much as two thirds, or about 60% of dipterocarp species in Sabah in 1955 and 80–93% of Dipterocarpaceae in Kalimantan in 1987 and 1991(Brearley et al. 2007).

Compared to other studies that examined reproductive activity in a variety of trees, including dipterocarps and non-dipterocarps, the percentage of the flowering season in WCRS was lower. Sakai et al. (1999) showed that the percentage of mass flowering occurs as much as 69% from all trees in Sarawak in 1996, while Schaik (1986) showed that approximately 65% of trees are flowering in Katambe, Aceh in the second peak of the mass flowering season.

The lower percentage of the major flowering peak in WCRS was likely due to differences in the composition of the seasonal patterns of flowering trees found in different regions. More than 50% of the species in dipterocarp forests of Barito Ulu and Katambe showed a super-annual flowering pattern and trees only flowered during mass flowering season (general flowering only) (Brearley et al. 2007; Sakai et al. 1999). In effect, this pattern means dipterocarp species can store the products of photosynthesis in the long term until the environmental conditions allow (Ashton et al. 1998). Therefore, when mass flowering season occurs, almost all individuals flower in large quantities at the same time.

On the other hand, the WCRS forest is composed of dipterocarp species which have more varied flowering season patterns such as annual, sub-annual, and super-annual patterns. Annual flowering pattern can be seen in *Dipterocarpus retusus* and *D. hasseltii*, while sub-annual patterns can be seen in *Vatica obovate* and *D. gracilis*, and the super-annual pattern can be seen in *D. humeratus*, *D. palembanicus*, *S. ovalis*, *Hopea sangal,* and *Anisoptera costata*. These various flowering patterns

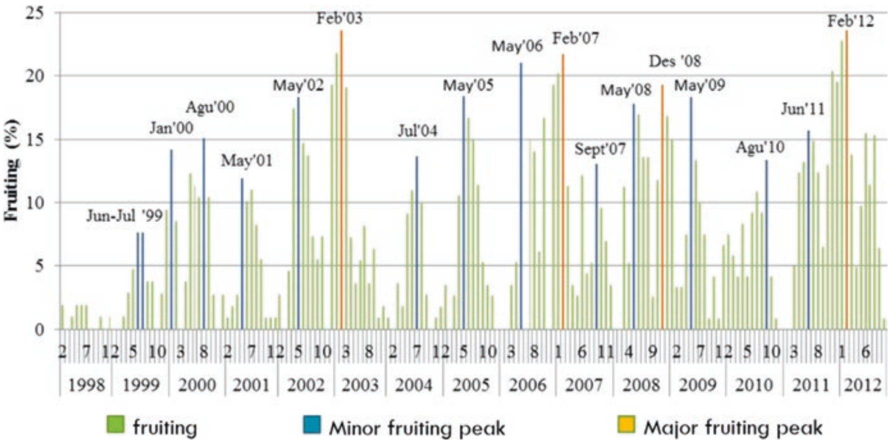

Fig. 15.3 Percentage of trees that bear fruit

shaped the general sub-annual flowering season of dipterocarps in WCRS and the percentage of trees flowering when the major peak is lower than that in other locations.

In WCRS, the fruiting season started 1 month after the flowering season and last for three to 5 months. Peak fruiting season may occur within those 5 months. For example, the fruiting season in 2001 occurred during 4 months (May–August 2001) with the peak fruiting season in May (the first month), while the fruiting season in 2000 occurred during 5 months (May to September) with the peak season fruiting in August (the fourth month) (Fig. 15.3). The length of the fruiting season indicated that the time required to develop the young fruits into ripe fruits was approximately 3–5 months.

The percentage of fruiting trees during the peak of the major fruiting season were 23% (February 2003), 21.7% (February 2007), and 23.5% (February 2012) (Fig. 15.3), relatively similar to the percentage of flowering trees.

Similar to flowering season, the pattern of peak fruiting season showed minor and major peaks. Minor peak season occurred five times in May, once in January, June, July, September, and twice in August. In general, the percentages of the fruiting trees were smaller when compared to the flowering percentages. This difference shows that not all flowers turned into fruit. The flowers' failure to develop fruits was caused by different reasons, such as predatory insects, abnormal growth, high rainfall rate, and strong winds.

The emergence of new leaves reached peaks once every year, which occurred eight times in March, once in January, February, April, May, June, and October (Fig. 15.4). The dipterocarps in WCRS generally produced new leaves in March or a month before the minor peak of flowering season. New leaves grow to produce more nutrients for the development of flowers and fruit (Raju et al. 2011). Also, the photosynthetic capacity of new leaves is higher than the old leaves, so more nutrients from photosynthesis will be accumulated (Hopkins and Huner 2006).

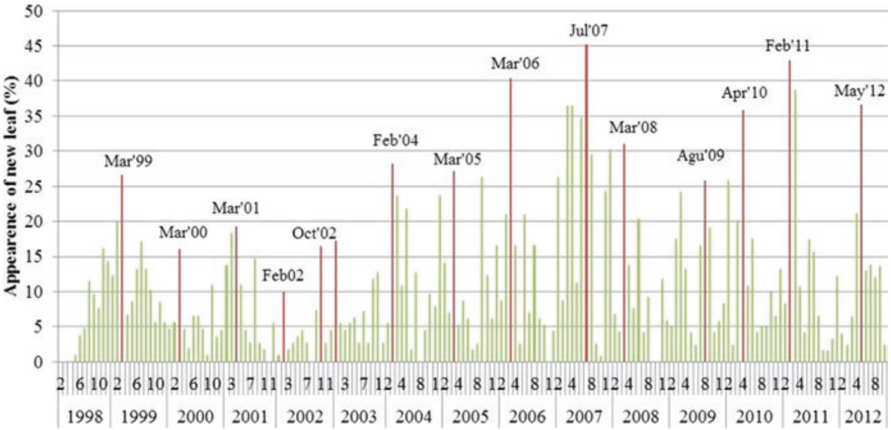

Fig. 15.4 Peak of emerging leaves

In general, the pattern of the minor flowering and fruiting season in dipterocarp phenology began with the emergence of new leaves in March, flowering in April, and ended with the appearance of the fruit in May or 3–5 months later. The distance between each phenophase was approximately 1 month. This pattern is a common pattern in the Dipterocarpaceae. The phenological patterns in Katambe (Sumatra), Kepong (Malaysia), and La Selva (Costa Rica) showed that new leaves usually emerged between January and April. The flowering season began 1 month after the emergence of new leaves and the fruiting season started 1 to 2 months after that (Sakai et al. 1999).

15.3.2 Community Structure of the Dipterocarpaceae

In general, the highest IVI in WCRS was dominated by *Vaticaobovata*, *Dipterocarpus humeratus*, and *D. hasseltii*. Six of the eleven species found in WCRS also existed in Java, *Anisoptera costata*, *Hopea sangal*, *Dipterocarpus retusus*, *D. humeratus*, *D. hasseltii*, and *Shorea javanica* (Fig. 15.5). The species *Dipterocarpus retusus*, *D. humeratus*, and *D. hasseltii* that dominate WCRS also dominate some areas in Java (Kalima 2010).

The low number of the *Shorea* species in WCRS might be because of seeds that did not survive during the early stage of growth, as well as the illegal use of *Shorea javanica* by the indigenous people. The inability to survive during the early stages of growth may be attributed to pollination failure, inbreeding pressure, resource limitation, and low synchronous flowering at the population level (Nakagawa et al. 2005). *Shorea* population in WCRS does have low synchronous flowering. Unfortunately, the data on phenology of *Shorea* was only available for *S. ovalis*. The number of *Shorea javanica* and *S. ovate* present at WCRS were quite low and were not included on phenology plot observations.

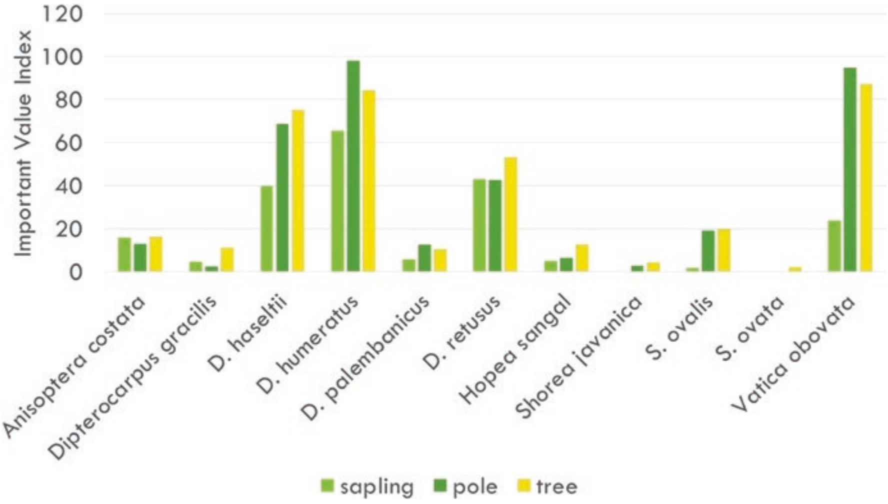

Fig. 15.5 Important Value Index (IVI) of dipterocarps in WCRS

Shorea javanica is commonly used for non-timber forest products around BBSNP where local people used to collect the sap of *S. javanica*. This was done by tapping or wounding the tree trunk. Tree tapping can cause brittleness and premature death of the parent trees at the age of 40–50 years old (Torquebiau 1984). Effects of excessive tapping were observed in the WCRS as many trees fell due to the effects of tapping. Death of the parent tree can reduce the chance of the tree to regenerate. Therefore, at this time, *S. javanica* sapling cannot be found in WCRS and only a few poles can be found. If this continues, it is feared that the population of *S. javanica* in WCRS will disappear in the future.

15.3.3 Effect of Phenology to Community Structure

A mass flowering and fruiting season is said to be an evolutionary adaptation of flowering and fruiting patterns in plants to face high mortality. Through the mechanism of mass flowering, plants can store energy of photosynthesis only when environmental conditions have supported the growth of seed (Ashton et al. 1998; Sunarto et al. 1999).

However, phenological pattern may not be consistently associated with the community structure of dipterocarps. This is illustrated by comparing the pattern of flowering species *D. humeratus*, *D. retusus,* and *D. hasseltii*. *Dipterocarpus humeratus* have super-annual phenological patterns, which only bloom in specific years. Although they only bloom in a given year, numerous individual saplings survive and dominate the region in WCRS (Fig. 15.5). In addition, although *D. hasseltii* and *D. retusus* have annual phenological patterns that bloom throughout the year, they are also able to dominate the region WCRS (Fig. 15.5).

Other factors that might affect the survival and regeneration of dipterocarp might be the fruit quality, fruit protection skin, fruit size, speed of growth, the presence of predators, and soil nutrients. The species that have the highest IVI in WCRS have a relatively large fruit with a thick rind. Fruit that has a thicker skin to protect it can withstand the attacks of insects or mammals. Large fruit generally has larger sprouts. Thus, it can serve to protect its embryos from the physical and chemical attack (Turner 2001).

Growth rate is one of the factors that determine the ability of certain species to win the competition between species in the tropical rainforest. Species that can tolerate the sunlight will grow faster and win the interspecies competition. In addition, the species that grow faster will also able to avoid herbivorous predators faster.

Known seed predators of dipterocarp species in WCRS include the cream-colored giant squirrel (*Ratufa affinis*), horse-tailed squirrel (*Sundasciurus hippurus*), squirrel (*Callosciurus* sp.), and the bearded pig (*Sus barbatus*) (WCS – IP 2001; Nakagawa et al. 2005). Also, some herbivores such as sambar deer (*Cervus unicolor*) and muntjac deer (*Muntiacus muntjac*) (WCS – IP 2001) are known to eat the sprouts. There may be other seed predators and herbivorous animals in WCRS, but more research is needed in this issue.

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References

Appanah S, Turnbull JM (1998) A review of dipterocarps: taxonomy, ecology, and silviculture. Center for International Forestry Research Indonesia, Bogor

Ashton PS (1988) Dipterocarp biology as a window to the understanding of tropical forest structure. Annu Rev Ecol Syst 19:347–370

Ashton PS, Givnish TJ, Appanah S (1998) Staggered flowering in the Dipterocarpaceae: new insight into floral induction and the evolution of mast fruiting in the aseasonal tropics. Univ Chic 132(1):44–66

Brearley FQ et al (2007) Reproductive phenology over a 10 year period in a lowland evergreen rain forest of Central Borneo. J Ecol 95:828–839

Cadelus CL, Chazdon RL (2005) Innercrown microenvironments of two emergent tree species in a lowland wet forest. Assoc Trop Biol Conserv 37(2):238–244

Fenner M (1998) The phenology of growth and reproduction in plants. Gustav Fischer Verlag 1(1):78–91

Ghazoul J, Sheil D (2010) Tropical rain forest, ecology, diversity, and conservation. Oxford University Press Inc, New York

Hopkins WG, Huner NPA (2006) Introduction to plant physiology, 4th edn. Wiley, New York

Kalima T (2010) Status populasi Dipterocarpaceae di Hutan Lindung Capar, Brebes, Jawa Tengah. Jurnal Penelitian Hutan dan Konservasi Alam 7(4):341–355

Molles MC (2010) Ecology: concept and application, 6th edn. The McGraw-Hill Companies, Inc, New York

Morison JIL, Morcroft MC (2006) Plant growth and climate change. Blackwell Publishing Ltd, Oxford

Nakagawa M, Takeuchi Y, Kenta T, Nakashizuka T (2005) Predispersal seed predation by insects vs. vertebrates in six dipterocarp species in Serawak, Malaysia. Biotropica 37(3):389–396

Newstorm LE, Fankie GW, Baker HG (1994) A new classification for plants phenology based on flowering pattern in lowland tropical rain forest tree at La Selva, Costarica. *Biotropica* 26(2):141–159

O'Brien TG, Kinnard MF (1996) Birds and mammals of the Bukit Barisan Selatan National Park, Sumatra, Indonesia. Oryx 30(3):207–217

Perlindungan Hutan dan Konservasi Alam (PHKA) (2003) Submission for nomination of tropical rainforest heritage of Sumatra. Directorate General of Forest Protection and Nature Conservation Ministry of Forestry, Jakarta

Poppenborg P, Holscher D (2009) The influence of emergent trees on rainfall distribution in a cacao agroforest (Sulawesi, Indonesia). Flora 204:703–739

Purwaningsih (2004) Review: sebaran ekologi jenis-jenis Dipterocarpaceae di Indonesia. Biodiversitas 5(2):89–95

Raju AJS, Ramana KV, Chandra PH (2011) Reproductive ecology of Shorea roxburhhii G. Don (Dipterocarpaceae), an endangered semievergreen tree species of peninsular India. J Threatened Taxa 3(9):2061–2070

Sakai S et al (1999) Plant reproductive phenology over four years including an episode of general flowering in a lowland dipterocarp forest, Sarawak, Malaysia. Am J Bot 86(10):1414–1436

Schaik CP v (1986) Phenological change in a Sumatran rain forest. J Trop Ecol 2(4):327–347

Sunarto Kinnard MF, O'Brien TG, Iqbal M (1999) Effects of the 1997 El Nino fires in Bukit Barisan National Park: post fire tree mortality and recruitment. PHKA/WCS – Indonesia Program, Bogor

Torquebiau E (1984) Man-made dipterocarp forest in Sumatra. Agrofor Syst 2:103–127

Turner IM (2001) The ecology of trees in the tropical rain forest. Cambridge University Press, Cambridge

Whitmore TC (1990) An introduction to tropical rain forest. Oxford University Press, New York

Wildlife Conservation Society Indonesia Program (WCS-IP) (2001) Taman Nasional Bukit Barisan Selatan dalam ruang dan waktu: laporan hasil penelitian 2000–2001. PHKA/WCS – Indonesia Program, Bogor

16

Integrated Landscape Modelling in India: Evaluating the Scope for Micro-Level Spatial Analysis over Temporal Scale

Mrinalini Goswami, Chaya Ravishankar, Sunil Nautiyal, and Rüdiger Schaldach

Abstract

In the field of computational intelligence research, "modelling" has emerged as important tool when integrated with traditional interdisciplinary methods to address complexities involved in land-use planning at landscape scale. This paper examines the existing spatio-temporal models in Indian context to look into their suitability to be used in micro-scale sustainable landscape planning. The research is expected to enhance the understanding of need of landscape as a planning unit for sustainable land-use/land-cover development. It will focus on integration of biophysical agents of the system with human agents by incorporating detailed socio-economic conditions of a micro-landscape to enhance the pragmatism of computational model. LandSHIFT, a spatially explicit model that uses mid-to-long-term scenarios of land-use and land-cover changes involving multiple biophysical and socio-economic factors, has been considered to embrace scope for modification to use in Indian context more proficiently.

Keywords

Landscape · Land-use/land-cover change modelling · Integrated landscape approach · LandSHIFT · Micro-level spatial analysis

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16.1 Introduction

Anthropogenic changes in land use and land cover have been recognized as critical factors influencing global change. Hence, land management and socioecological development are becoming the prime concern among researchers, policy planners, and decision-makers. Land cover refers to the biophysical earth surface; land use is shaped by human, socio-economic, and political influences on land (Geist and Lambin 2001). In this approach, landscape is mosaic patches of area with interacting cluster of ecosystems (Forman and Godron 1986) and heterogeneous wherein at least one factor of interest (Turner et al. 2001) is to be considered as unit of analysis. It is only prudent to study the process of land-use and land-cover change simultaneously to understand the human impacts on environment as well as the impacts of global change like climate change. Nautiyal et al. (2010) emphasized on the need to analyse the complexity of interactions and feedbacks between socio-economic and ecological systems at landscape scale and, therefore, proposed a conceptual modelling framework that helps understanding the science behind sustainable landscape management and hence finding feasible solutions through a problem-solving approach. It requires knowledge that transcends disciplinary blinds and help integrating multiple objectives of researchers from different disciplines. One of such interdisciplinary approaches is "LandSHIFT modeling framework". Model realizations which have been built using this framework have successfully been implemented and evaluated at global as well as in country levels across Africa, South America, Europe, and Asia (Schaldach et al. 2006; Lapola et al. 2010). The model has focus on different aspects of land-use systems (urban development, agriculture, animal husbandry, supply and demand for ecological resources). This empirical data-based model involves data collection at five levels; and the database is divided into three sections—(1) spatial base data for the micro-level, intermediate level, and country level, (2) data model parameters, and (3) scenario data. This facilitates not only individual stakeholder's decision in planning of economic activities but also helps in ecological performance of the landscape.

Models help to understand the complexities of socio-economic and ecological systems, specially the spatial dynamics and processes over temporal scale. Landscape models are typically mathematical kind, although we often construct and use abstract conceptual models to portray landscape relationships. Modelling land-use, land-cover, and environmental change is a field of increasing importance, and a broad range of models has been developed for this discipline (Haase and Richter 1983; Schaldach and Priess 2008). Integrated land-use model provides a link between the human dimension of global land-use change with global scale simulations of climate, the water cycle, biodiversity risk, and other global change processes (Schaldach et al. 2011).

Integrated modelling approaches are able to deliver the outcome towards evaluating different policy scenarios at various spatial and temporal scales, facilitating formulation of effective policy recommendations for sustainable development of a landscape considering all the probable driving factors of landscape change. Models on land-use/land-cover change (LULCC) are powerful tools that can be used to understand and analyse the important linkage between socio-economic processes associated with land development, agricultural activities, and natural resource management strategies and the ways by which these changes affect the structure and function of ecosystem (Agarwal et al. 2002).

Having realized the importance of landscape modelling, there are two major dimensions of modelling which this paper tries to emphasize upon: space and time. Models of biophysical and/or human processes operate in a temporal context, a spatial context, or both (Agarwal et al. 2002). The "space" dimension refers to the geographical extent and may refer to as micro-scale which in India could be a cluster of a few villages, a wetland with surrounding socioecologically dependent area, or a protected area. Secondly, the time dimension is to be considered in terms of assessment of future socioecological consequences of land-use policies, which will be useful for supporting decisions about where and how to progress with socio-economic activities. It is important to integrate spatially explicit land-use change models with socio-economic and bio-physical variables. Such models can be more efficiently used for the projection of suitable future development alternatives and for conducting experiments to enhance understanding of key processes concerning LULCC (Verburg et al. 2004).

Modelling land-use and land-cover dynamics in the context of developing countries like India will rely on micro-scale data since the drivers of change in these countries are individual agents with distinct characteristics (Peter et al. 2014). These drivers vary widely because of highly heterogeneous bio-physical and socio-economic environment and unequal development within the country. The emergence of multiple anthropogenic environmental change and rapid transition of land use/land cover cannot be distinguished from socio-economic agents of change. In developing countries like India, land-use activities largely decide the land cover rather than environmental determinants (e.g. soil quality, climate, etc.) of land cover, e.g. land-use activities associated with logging leads to a deforested land cover (Lambin 1997). Though rich in biological diversity, culture, and traditions, India is also growing fast economically, resulting in proliferation of industries, expansion of urban areas with conversion of agricultural land (Pandey and Seto 2015; Moghadam and Helbich 2013; Mallupattu et al. 2013; Fazal 2000), and change in the livelihood patterns of peri-urban communities (Narain 2009). Therefore, land-use planning at landscape scale is essential for striking a balance between economic growth, conservation of the biodiversity, and safeguarding the cultural heritage. The linkage between human and environment subsystem is influenced by several driving forces including—population growth, economic growth, technological change, political and economic institutions, and cultural attitudes and beliefs. These factors are accounted in global environmental change research. The importance of the said linkage should be acknowledged at landscape level planning for two reasons: (1) characteristics of driving forces for land-use change vary at different spatial scales, and (2) the decisions for land management are taken at individual stakeholder's level (such as farmer).

Therefore, satellite images can be used to detect LULCC through observations of the biophysical characteristics of the land and empirical data from socio-economic survey can be integrated to assess the role of driving forces. For instance, forestry can be modelled as a land-use activity that responds to economic, social, and demographic drivers (Mauldin et al. 1999; Geist and Lambin 2001). But such drivers do not provide a holistic idea of vegetation cover and cannot necessarily be used to forecast the probable land-cover modifications in the landscape.

With the above backdrop, an attempt has been made to analyse sustainable socio-ecological development pathways, particularly regarding the relationship between human land-use activities and environmental processes at landscape level over a large temporal scale with the following objectives:

1. To apply a landscape model (most appropriately, LandSHIFT model) realization that supports the decision-making at micro-landscape scale in the study regions.
2. To look into the diverse landscape models developed worldwide which are applicable to Indian context.
3. To identify the gaps in integrated micro-scale landscape modelling and assess the scope of integrated modelling approach for landscape management.
4. To examine the feasibility of using LandSHIFT in varied micro-level spatial analysis over large temporal scale.

Land-use/land-cover change models could be reviewed using various criteria including the techniques or methods of modelling, performance in particular area like multiple scales, capabilities to integrate different subsystems of a landscape, and contribution towards policymaking. The scope of the research includes only review of papers and previous studies to achieve the aforementioned objectives. It does not intend to explore empirical evidences for appraising prospects and feasibility for using modelling for micro-level landscape analysis in India. The description of technical attributes of models and technical specifications required for an efficient model for the aforementioned purposes don’t come under the scope of this paper.

16.2 Methods

All the models used in Indian context have been listed and examined for different components including special features of the model, applicability, scale of use, and limitations. India has varied geo-climatic conditions (snow-covered Himalayas, vast coastal zone to desert), biodiversity areas (rich tropical rainforest to drylands), and 127 agro-climatic zones. Looking at the magnitude of present environmental problems, land-use planning, natural resource management, and threats to biodiversity and livelihoods of the large population of the country, the gaps in research for integrated landscape modelling have been identified.

A systematic approach can be adapted for this from the Integrated Environmental Modelling components as referred by Laniak et al. (2013), which involves (1) systems thinking, (2) stakeholders involvement, (3) community development, (4) openness (e.g. transparency), (5) reusable products (software design or outcomes), and (6) investments.

The ability of LandSHIFT model to compute land suitability, cropland and grassland productivity, location of deforestation, and its sensitivity towards global environmental changes like climate change have made it a potential candidate for further evaluation. Detailed investigation on LandSHIFT model has been done to look into its features, applicability, flexibility, scope for downscaling, and usability of outputs for further analysis and informing policy. The model is based on a "land-use systems" approach, which describes the interplay between anthropogenic and environmental system components as drivers of land-use change. LandSHIFT's modular structure facilitates the integration of different components that cover key components of land-use systems. A unique feature of LandShift is its "Socio-Economic Translation Shell" which translates qualitative knowledge about land-use processes into various input data and parameters of the model (Alcamo and Schaldach 2006). The other spatial variables that LandSHIFT deals with are information on crop yields and grassland net productivity that are calculated with a biophysical model as well as GIS data on land use, human population density, river network density, road infrastructure, and conservation areas. Consequently, in the centre of our efforts stands the modelling for micro-level decision-making processes such as farmers in a protected area where sustainable use by indigenous community and small-scale agriculture is permitted (e.g. IUCN V and VI PA categories). These processes are not unidirectional but always influenced by several driving factors such as:

(a) Biophysical conditions.
(b) Demographic and socio-economic factors.
(c) Policies.
(d) National economy.

These driving factors influence individual stakeholder's choices of action in a landscape. For sustainability standpoint, the decisions on every branch have to be assessed in economic and ecological perspectives. The steps to develop the model realization within the LandSHIFT framework are depicted in Fig. 16.1.

A major difference among the various working models is the level of land-use decision-making. Substantial progress has been made on different aspects that are part of the modelling cycle (Verburg 2006). These include (a) areas of model validation, (b) participatory approaches to develop storylines of scenarios, (c) incorporation of biophysical and social impacts, and (d) the linkage between process-based and pattern-based descriptions of the driving forces. With these advancements, decision-making for land use has been integrated with analytical and simulation models for spatio-temporal changes of landscape. With the higher assimilation of "people's" choice making in the model, the complexity of the model increases with regard to linkages among actors and data demand.

LandSHIFT is a model with intermediate complexity. It is a dynamic and spatially explicit land-use and land-cover change model (Schaldach et al. 2011) that can be used to quantify potential LULCC with respect to future socio-economic

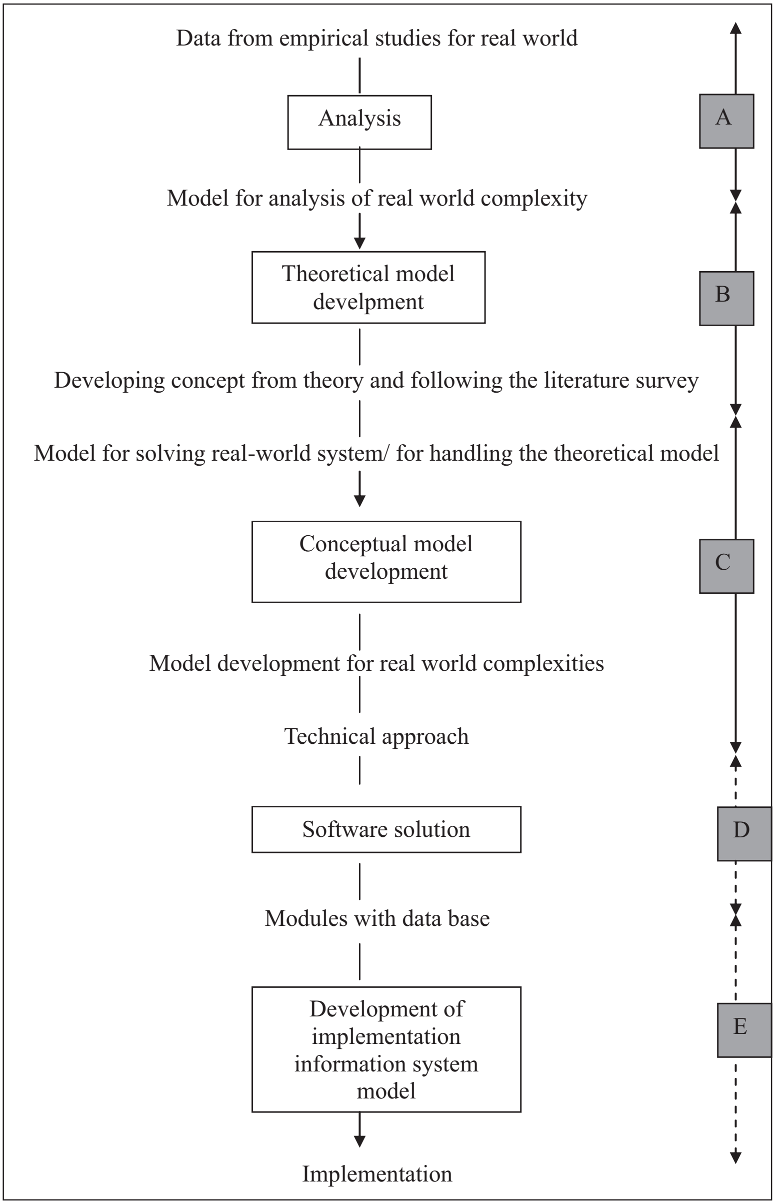

Fig. 16.1 Research design for LandSHIFT Model development for the real world modelling. (Nautiyal et al. 2010)

changes. Several research studies have applied the model for impact assessments under varied conditions (Alcamo et al. 2011; Schaldach et al. 2012) adjusted to the South-East-Asian study region to operate at a spatial grid cell resolution of 1 km^2. LandSHIFT provides maps in 5-year time steps. It consists of three sub-models, crop, urban, and livestock. Each sub-model uses weighted multi-criteria analysis at the grid cell level which considers biophysical, geographical, and socio-economic conditions. LandSHIFT's output includes grid maps in time series which have future projections for LULCC and can be further used for impact analysis such as climate change, urbanization, and deforestation.

16.2.1 Types of Land-Use Models

Various land-use models, such as equation-based models, econometric models, empirical statistical models, spatial system models, agent-based models, and cellular automata models, have been used to simulate large-scale land-use/land-cover change. EURuralis (Europe's Rural Areas) study used a top-down modelling approach to provide information on LULCC on different scales by applying downscaling procedures to continental level data. Henseler et al. (2009) stated that there are various studies that differed in their focuses, approaches to modelling land-use changes, base years, time horizons, and the numbers of scenarios, as well as their investigation areas and spatial scales. The allocation of land at the local level is represented using a spatially explicit model that considers variations in biophysical, socio-economic, and policy characteristics (Verburg 2006). In scale-sensitive analysis, it follows that spatially explicit modelling of land-use change should be scale-dependent and interdisciplinary (Verburg et al. 2004). Many models have amalgamated socio-economic and biophysical dynamics in land-use change model. No single model is able to capture all key processes essential to explore land-use change in Europe at the different scales relevant to make a full assessment of driving factors and impacts (Verburg 2006). The integration of cellular automata with socio-economic and bio-physical model has been a challenge and data extensive. It has also encouraged researchers to develop bottom-up approach by analysing spatial data with associated human, ecological, and economic actors of the landscape.

In a study contracted by European Commission (Tucker et al. 2009), "Scenarios and models for exploring future trends of biodiversity and ecosystem services changes" has tried to clarify which models and scenarios are being used and can be used to explore the biodiversity and development of ecosystems in light of different assumptions of drivers and policies. In the category of terrestrial integrated assessment models, the IMAGE model, the AIM model, MIMES, and the related GUMBO models received the best scores. The GUMBO and MIMES models are from the same modelling group; MIMES is still under development to provide a spatially explicit version of GUMBO. Models with a complex set of sub-models make it difficult to assess the capabilities. For example, the AIM model has been evaluated through downscaling exercise and found as efficient in estimating global changes (Hasegawa et al. 2017). But the evaluation of quality of the model is complex because of its sub-models with different levels of description (IEEP et al. 2009). Despite being comprised of a complex set of sub-models which are well described, the IMAGE model has emerged as extensive in global assessments and has also been used as a basis for GUMBO/MIMES. These mechanistic models (e.g. AIM, IMAGE, CLUE) require less computational effort but high degree of understanding to link the sub-models in the modelling framework. The ATEAM and InVEST assessment tools include cultural services, mainly related to recreation and aesthetic values of landscapes. The regional assessment tools that were evaluated, i.e. ATEAM and InVEST, follow an interesting approach that could provide the necessary framework to combine model outputs and assess impacts on value of ecosystem goods and services. These models build on existing models and use their outputs, while increasing feedbacks and inter-linkages between components. Disadvantage is that they are relatively data demanding.

Upadhyay et al. (2006) assessed the features and use of different categories of land-use change models including economic models, conceptual models, empirical regression models, linear and non-linear programming, and simulation models. Agarwal et al. (2002) reviewed 136 relevant papers and summarized 19 models in terms of temporal dynamics and spatial interactions as well as human decision-making. Status of computer simulation models from around the world for evaluating the possible ecological, environmental, and societal consequences of global change has been presented in a paper by Kickert et al. (1999). Schaldach and Priess (2008) developed a framework for reviewing the models, taking a strict system-oriented perspective, which is based on the conceptual model of land systems defined within the science plan of the Global Land Project (GLP 2005). They assessed the structure and functionality of eight different land system models from regional to global scale.

Land-use planning and analysis system (LUPAS) was designed as a decision support system for strategic land-use planning having a promising perspective for providing effective policy support to the possible link of the regional LUPAS approach with farm household model (Roettera et al. 2005). CLUE-S model was originally designed for the regional or watershed level; it has recently been applied on the continental scale for a European-wide scenario analysis, together with the IMAGE model (Verburg 2006). Schaldach and Priess (2008) demonstrated that the CLUE framework does not explicitly represent processes of the environment sub-system. Nevertheless, different studies illustrate how the CLUE model can be combined with environmental models of different levels of complexity.

16.3 Uncertainties in Modelling

Land-use change model outcomes are vulnerable to multiple types of uncertainties related to structure of the model input data and model parameters (Verburg et al. 2013). Translating macro-level uncertainties to uncertainties in spatial patterns of land change makes it possible to better understand and visualize the consequences of uncertainties in LULCC in model input variables.

Several research papers on landscape modelling have been revisited to assess the use of different landscape modelling in Indian context at national, regional, and local scale. The review of usage of these models has been performed with special focus on specific features, applicability, scale, and limitation of those models. The review revealed practice of seven such landscape models in India with special respect to land-use planning using construction of sustainability scenarios, resource availability, yield estimation, carbon dynamics, BIOME mapping, bio-economics of land-use system, biodiversity, and species richness (Table 16.1).

Five of the seven models listed have been used at regional scale, viz. LUPAS (Land use Planning and Analysis System), FOPIA (Framework for Participatory Impact Assessment), DIMA, BLC (Biome Level Characterization), and BIO_CAP. The only model used at micro-scale is IMGLP (Multiple Linear Programming) integrated with Stakeholder Community Matrix, which has been applied to explain the bio-economics of the land-use system that interrelates among different stakeholders with communication as an indicator.

LUPAS is applicable for trade-off analysis of multiple goals in complex situations, fine-tuning resource management system, and comparison of resource use intensity and environmental cost of alternative production activities. Scenario analysis of impact assessment of land-use functions based on qualitative knowledge of stakeholder and experts can be done by applying FOPIA and IMGLP-SCM, which is also used at farm scale. Mapping of biological richness, bio-prospecting, prioritizing conservation, and monitoring biodiversity loss and gain in spatial term are key applicability of BIO-CAP (Roy and Giriraj 2008). Though the aforementioned models are able to address varied issues with relation to land use, integration of all imperative aspects of ecological and human subsystems including policies contextual to spatial variability is needed. Limitation of the described models can be summarized in following points:

1. Lack of adequate spatial and temporal differentiations.
2. Low integration of sustainability aspects.
3. A few are preliminary assessment tools that lack evidences and data from ground level monitoring and support of analytical tool.
4. Identification of environmental management alternatives at local or regional levels to support policymaking is not well-oriented in all the models.
5. Use of insufficient parameters to arrive at conclusion on landscape processes with relation to other components of biophysical environment.

Table 16.1 Various models, special features, applicability, and limitations

Model	Built by	Special features	Applicability	Scale	Limitations
LUPAS (Land-Use Planning and Analysis System) Paassenet al. (2007), Roettera et al. (2005) developer	"Systems Research Network for Ecoregional Land-Use Planning in Tropical Asia" (SysNet), 1996	(i) Land evaluation, including assessment of resource availability, land suitability, and yield estimation; (ii) scenario construction based on policy views; (iii) land-use optimization	(i) Trade-off analysis of multiple goals in complex decisions situations; (ii) fine-tuning of resource management systems; (iii) comparison of resource use intensity and environmental costs of alternative production activities	Regional	1. Lack of adequate spatial differentiation 2. Models lack the relevant time dimension. Lack of Sustainability aspect
FOPIA (Framework for Participatory Impact Assessment)	Morris et al. (2008)	The core aspect of FoPIA is the stakeholder-based assessment of alternative land-use scenarios	Scenario analysis for impact assessment of land-use functions based on qualitative knowledge of stakeholders and experts	Regional (Bijapur – India)	FoPIA is suitable as a first-step assessment tool, but it can't replace a comprehensive quantitative impact assessment. Should be supported with external data and evidence.
DIMA	Built on Model built by Benítez and Obersteiner	Ecological modelling of carbon dynamics with economic modelling of land use	Models the interactions and feedbacks between ecosystems and human land-use activities spatially under given biophysical and socio-economic constraints	Regional / Global	P.C. Benítez, I., Global Supply for Carbon Sequestration: Identifying Least-cost Afforestation Sites Under Country Risk Consideration, International Institute for Applied Systems Analysis

BLC (BIOME LEVEL CHARACTERIZATION)	National Remote Sensing Agency (NRSA) 2005	Three major inputs: (1) vegetation cover type information derived from the satellite data, (2) prominent life zones from the HLZ model, and (3) interaction of actual vegetation cover boundaries existing in the different life zones defines biome boundaries	For biome mapping, identification of Holdridge Life Zone(HLZ), Vegetation cover classes, and types and sub-biomes	Regional	Based on vegetation types and the socioecological parameters are not integrated
IMGLP WITH SCM (STAKEHOLDER COMMUNICATION MATRIX)		An Interactive Multiple Goal Linear Programming (IMGLP) model is developed that considers objectives of multiple stakeholders, i.e. different farmer groups, district agricultural officers, and agricultural scientists for agricultural land-use analysis	The IMGLP model is a useful tool to model bio-economics of the land-use system; the social context in which its results should be implemented determines their relevance. The SCM is a relevant tool to describe the interrelations among stakeholders with communication as an indicator	Local/ micro-scale/ farm-scale	Weights have relative importance and one objective can only be optimized. Prime use may be for assessing farmers' choice

(continued)

Table 16.1 (continued)

Model	Built by	Special features	Applicability	Scale	Limitations
BIO_CAP (Roy and Giriraj 2008)	Roy and Behera (2005)	To establish the relationship existing between biological richness and biotic disturbances across an altitudinal gradient, divided into six zones based on different vegetation types. Biological richness was determined as a function of ecosystem uniqueness, species diversity, biodiversity value, terrain complexity, and disturbance index	(i) Rapid assessment for monitoring biodiversity loss and/or gain, (ii) assessing the nature of habitat and disturbance regimes, (iii) evolving species–habitat relationship, (iv) mapping biological richness and gap analysis, and (v) prioritizing conservation and bioprospecting sites	Regional	This study used only one parameter, i.e. altitude to establish the relationship. Many other parameters, if considered, would provide further insight into the understanding of biodiversity characteristics
GEOMOD2	GEOMO2 is calibrated using the land-use map from one point in time. The model is then run to the second point in time, and the results are validated by comparing the simulated map with the actual land cover and land use, of the second point in time	GEOMOD2 is a spatially explicit model which simulates future land-use change based on either a statistical analysis of how people have used land in the past or alternative hypotheses of how people will use land in the future	GEOMOD2 can then make a projection of future deforestation to highlight locations that are most susceptible to forest loss	Regional	Simple assumptions about decision-making process. It doesn’t predict the quantity of land-use change

16.4 Integrated Landscape Modelling at Micro-scale in Indian Context

Land-use change is driven by multiple factors such as resource scarcity leading to an increase in the pressure of production on resources, changing opportunities created by markets, outside policy intervention, loss of adaptive capacity and increased vulnerability, and changes in social organization, resource access, and attitudes (Roy and Giriraj 2008). Deriving estimates from models of LU and LUC in time and space, driven by aforesaid factors for the assessment of future forest cover and its impacts on economic, social, and policy changes, would be useful for land-use managers and policymakers. Some socioecological functions (maintenance of environmental qualities, food production, socio-cultural habits) are operational and structures are dominant in much smaller scale which hardly can be considered in global and regional models (Schaldach and Priess 2008).

The complex rural landscape in Indian situation is depicted in Fig. 16.2. In the centre of this complexity, there are decision-makers (farmers) who need to be supported in the decision-making process. In the entire process, the integrated approach will encompass relations between researchers and other actors in the policy process, allowing for exchanges, co-evolution, and joint knowledge creation with the aim of enriching the decision-making process at different levels. Thus, scientific information (scenario results, externalities, landscape level land-use change process and pattern, visualization of future through modelling, and socio-economic analysis from our research) will support the requirements of policy demand and will be

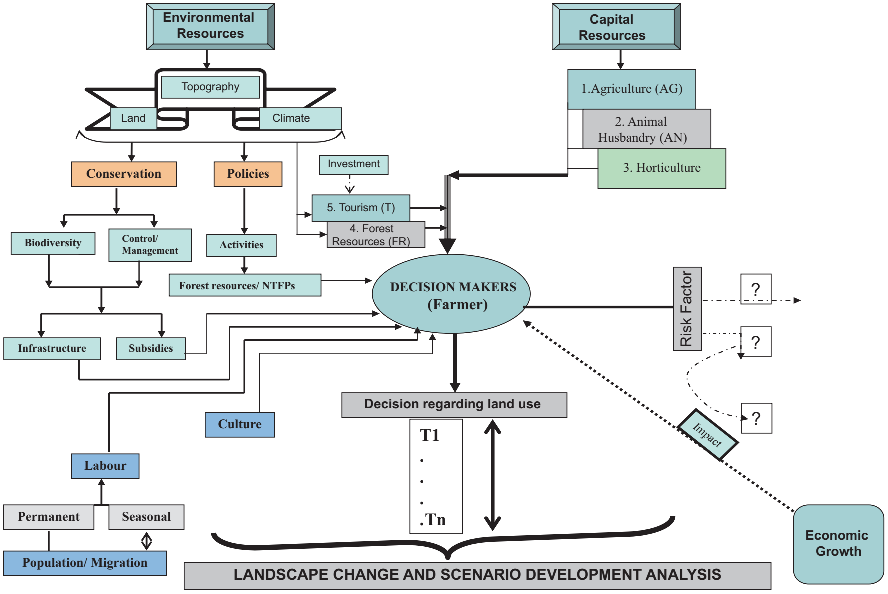

Fig. 16.2 Complex rural ecosystem in India influences the decision on land use. (Nautiyal 2011)

easily accessible to the different stakeholders. This approach could help evolve effective responses for land-use management (i.e. impact, causes, effects, adaptation, conservation, and management of biodiversity) at regional, local, and landscape levels.

India has not progressed so much towards integrated landscape modelling which can essentially contribute to land-use planning in both rural and urban arena. There are no established procedures for analysing conflicts using quantitative methods to aid decision-making by reducing the circumstances where economic development clashes with environmental and conservation aspirations which are qualitative in nature.

This type of integration of modelling requires land-use researchers to transcend disciplinary boundaries. Linking socio-economic and geographical data is a means to provide information on the context that shapes social phenomena (Verburg et al. 2004). Lack of extensive and systematic data and holistic research adds up to the limited implication of integrated landscape modelling in Indian context. Key challenges identified by survey respondents of a study (Curran and Atkinson 1998) include the long time horizon required to achieve results at scale, unsupportive policy frameworks, and difficulty in engaging the private sector and other important stakeholders. Overcoming these challenges would offer insights for improving the future effectiveness of integrated landscape initiatives.

Other important factors to be considered at micro-scale level are farm household as unit of analysis, the physical and economic constraints individual household faces, and social and psychological factors. Hence, when a policy change occurs, the model is able to capture the direction and speed with which groupings of households adopt or respond to the change for assessing the effectiveness of the policy (McGregor and Simon 2012). To reduce spatial heterogeneity which is vital in describing the land surface processes, Curran and Atkinson (1998) and Garrigues (2004) proposed to define proper pixel size to capture the spatial variability of data and minimize intra-pixel variability.

There are two basic approaches to biophysical assessment with respect to landscape processes. The most common approach consists of identifying the constraints that could inhibit a particular land use and looking for sites where the constraints are manageable. The other is determining the best uses for each site within a given parcel of land, i.e. suitability analysis. The effective models for predicting scenarios should be able to embrace both the approaches including all the possible biophysical parameters with accurately assigned weights.

Agent-Based Models (ABMs) have gained immense importance in recent year as they provide insight into the process of interaction between human and biophysical environment (Huigen 2003) and incorporate the influence of human decision-making on land use in a mechanistic, formal, and spatially explicit way, taking into account social interaction, adaptation, and decision-making at different levels (Mathews et al. 2007). TheABMs have many applications and explain a rapidly developing field of enquiry, spanning both the physical–mathematical and social sciences. The agent-based modelling focuses on approach to model decision-making in the context of human environment relationships. This application contributes to methodological innovations in multi-criteria evaluation and modelling of coupled human–environment systems (Manson 2005). The ABMs are seen as most appropriate in situations where decisions or actions are location-specific, i.e. at micro-scale, and individuals modify the structure based on their interactions (Clifford 2008). Agent-based LULCC models give a holistic picture by integrating landscape interests with decision-making entities (d'Aquino et al. 2002) and hence are more efficient for analysis of spatial processes, spatial interactions, and multi-scale phenomenon.

By providing these causal relations, policy-makers can be given the opportunity to spatially investigate the driving forces and the current state of development. However, it is also vitally important for decision-makers to understand how trends will develop in the short-term and the possible impacts of their decisions on the development process. Hence, there is a need to look into feasibility of scenario analysis.

16.5 Scenario Analysis: Planning for the Future

Scenarios are not used to predict the future, but to explore a range of possible futures by considering alternative long-term developments. However, scenarios acknowledge fundamental uncertainties with regard to future development, but do not predict (Pearson et al. 2010). Swetnam (2010) tried to build scenarios using local stakeholders and experts to define how major land-cover classes may change under different sets of "drivers". Scenarios enhance effective planning and management of environment by:

(a) Looking into the future.
(b) Amalgamating neighbourhood effects, instead of planning for independent ecosystem, especially in case of ecological conservation and sustainability.
(c) Projecting the impact of existing policy.
(d) Incorporating formulation of new policy for sustainable management of landscape.

Integrated land-use models provide a link between the human dimension of global LUC (including economic drivers) with global scale simulations of climate, the water cycle, biodiversity risk, and other global change processes and issue (Schaldach et al. 2011). Identification of drivers for future changes and hypothetical extrapolation of those to assess various levels of sustainability is an important step in this regard. Simulation models are construction of reality and at best are tools that should provide insights into a problem which may affect the landscape adversely in future. In the field of studying complex landscapes, such models can help organize or synthesize our understanding of the ecology of a system, and this understanding may also be applied in making relative comparisons among scenarios of future changes.

The future events or pathways of changes are incorporated as storylines in modelling as “scenarios”. “Scenarios are plausible, proactive and relevant stories about how the future might unfold” (MEA 2005). Mapping socio-economic scenarios for land-use change and consequent changes in the ecosystem is an important aspect of integrated ecological study (Alberti 2003; Peterson et al. 2003; Reeds 2009).

16.6 Need for Integration

Integration of ecological, economic, and social subsystems in land-use models encompassing all kinds of activities within a landscape helps to move towards sustainable development through mitigation, adaptation, learning, and innovation. A micro-plan is apparently effective for forest management, sustainable agricultural development, and overall management of activities in a landscape. Unlike the traditional top-down approach for landscape development, micro-plan was formulated through a bottom-up approach with stakeholders involvement, extensive interaction and consultation among communities, forest department, development authorities, and other stakeholders. Integration of spatial change of the landscape over time to envisage the future changes makes such plans more pragmatic with focus on people’s priorities and managing environmental externalities as well.

16.6.1 Biodiversity Conservation, Protected Area, and Policy

Habitat fragmentation is considered one of the most serious threats to biological diversity and is a primary cause of species extinction. Knowledge on land-use/land-cover change and consequent response of species to habitat processes should be analysed as a planning input in a spatial format which is more meaningful and usable for conducting scenario analysis and thus help planners and PA managers.

The requirements of a harmonized and sound monitoring and management of the designated protected areas are a challenging task for local authorities responsible for the individual site, as well as for the concerned authorities on the regional, national, and global levels who are in charge of the assessment and making sense of the huge data flow arising with the monitoring activities from the local sites. For the monitoring of ecologically important aspects of landscape patterns, landscape metrics are in principal applicable as structural indicators. Haines-Young and Chopping (1996) proposed landscape metrics as structural indicators to monitor ecologically important aspects and landscape patterns. However, they also identified that understanding the precise impacts of patterns on landscape processes has been the greatest challenge for applying landscape metrics which can be addressed through effective use of landscape modelling for trend analysis.

16.6.2 Urban and Peri-Urban Growth

Urbanization, in particular, fundamentally alters both biotic and abiotic ecosystem properties within, surrounding, and even at great distances from urban areas (Grimm et al. 2008). The linkage between ecosystem services and livelihood is very much apparent in rural and peri-urban landscapes where people are closer to the nature and with direct dependency on natural resources. Integration of socio-economic conditions of the population with changing land-use/land-cover and consequent ecosystem services in a model to look into the future for sustainable management of the ecosystem is critical.

In the present context of urban planning in India, it is important to look into the aspects of modelling for the future because of the unpredictable growth pattern of the urban areas. Land Change Modeller for prediction of urban growth and future land use has been very widely used in India (Kumar et al. 2011; Bharath and Ramachandra 2013; Ramachandra et al. 2013; Sudhira et al. 2004). Models of land-use change should be linked with critical socio-economic parameters, and the future changes can be envisaged by incorporating scenarios for various driving forces. City development authorities, city corporations, and department of town planning of Indian cities and towns have been developing City Development Plan, Master Plan for future developments in the urban limit for the next 10–20 years. Urban planning and development involved different institutions, which generally lack co-ordination. Integrated modelling will facilitate incorporating all the features crucial for forecasting urban growth, resource requirement and availability, and necessary planning to facilitate sustainable urban development in the country.

16.6.3 Livelihood Security and Rural Planning

There are a large number of modelling tools available today (which differ in focus, timeline, assumptions, spatial resolution, sensitivities, and in choice of indicators of biodiversity and ecosystem services) and most are able to capture various forms of ecosystem service provisioning to a reasonable degree (Tucker et al. 2009).

Most of the bio-physical models are sensitive to micro-climatic conditions and the climatic variables are significantly influenced by local topography. These models aid the understanding of biological, ecological, geological, hydrological, and anthropogenic processes within the ecosystem and are key to accurate prediction and long-term sustainability of rural landscapes. Integration of downscaled global-seasonal climate forecast with micro-scale landscape model can contribute to local environmental decision support systems. Sustainable Livelihood Framework (DFID 1999) for assessment of livelihood sustainability coupled with LULCC models can be efficiently used to understand natural resource flows in rural landscapes and enhance and help informed decision-making. However, such approach poses a great challenge due to the inherent complexity of human-landscape system. Multi-agent system models (e.g. agent-based modelling) have been recognized to be well suited to express the co-development of the human and bio-physical components in response to policy interventions.

16.7 Challenges in Indian Context

Very few modelling exercises have been able to uptake the findings of recent researches carried out on landscape modelling. The national spatial databases enabling monitoring of temporal dynamics of agricultural ecosystems, increase in mining, industrialization and urbanization, forest conversions, and surface water losses are lacking (Roy and Giriraj 2008). In contrast to developed economies where land-use change models have reached a matured state, the state of application in developing countries is still lagging much behind (Elias et al. 2012). Among the range of challenges, in India, following can be stated as the major ones:

Lack of available data

1. Conventional gap between research-community-policy (science-policy and practice?)
2. Growing economy and unpredictable growth (uncertainties).
3. Lack of interdisciplinary and transdisciplinary research.

Multi-agent system models are considered as compatible to express the historical developments of human and landscape system as a response to policy interventions. A major challenge in environmental planning in countries like India is the general lack of (reliable) data sets, especially of long-term data records (Ritzema et al. 2010).

For most analyses, data scarcity is considered as the main hindrance. Crossman (2009) argues that even though there is lack of quantitative data for assessing ecosystem services, spatial distribution of many elements of natural capital can be quantified using modelling techniques. The proposed model should be able to overcome this challenge as it is expected to be implemented at a smaller spatial extent with primary empirical data which can be collected keeping the specific need of input parameters. Menon and Bawa (1997) have done such work on spatially representing the biodiversity of forests and many important ecosystem hotspots of India. But there is a need to look for predictive and participatory modelling which help inform policies. An institutional framework is required to be put in place that emphasizes good data collection which feed into model and help in the development and maintenance of an appropriate environmental monitoring system. The system so developed would provide enough observations so that trends in the development processes can be analysed and subsequently simulated in the form of scenarios, which can be incorporated into a model in order to provide policy-makers an idea of the possible future land-use change in a given landscape.

In the Western Ghats and the Himalayas, the global biodiversity hotspots, for the last three decades, tremendous increase in population coupled with a variety of other factors such as socio-economic and cultural changes, change in land use, vast erosion of traditional germplasm, introduction of the modern agro-technologies, encroachment to forest for expansion of agriculture, creation of protected areas, implementation of the policies for conservation, human–resource conflicts, over-exploitation of the forest resources, and impact of climate change on ecosystem services such as water, carbon flux, and carbon sequestration are the factors for the rapid structural and biophysical changes of the landscapes. Apart from these, efforts for sustainable development of arid land-use system are other important aspects for further research. Arid regions are important but unfortunately have been less prioritized for research in India due to less diversity in bio-resource as compared to the Himalayan and Western Ghats regions. The arid regions of India cover over about 300,000 km^2 area and are spread over six states of the country and most vulnerable to climate change. Therefore, it is imperative to develop the land-use models for arid regions for conservation and management of this vulnerable agro-climatic region.

16.8 Landscape Modelling Approach in Indian Perspective

The land use in case of Indian rural landscape is not an independent sector and this is interwoven with other biophysical and socio-economic entities. Thus, change in land use makes significant impacts on various components and processes of existing ecosystems within a landscape. To understand the entire change process, it requires adequate information and scientific methodologies to evaluate scenario results of the system functioning. For example, under the Joint Forest Management and other forms of Participatory Forest Management programmes, village communities are efficiently involved in sustainable forest management in India; integration of socio-economic and demographic parameters with the pattern of bio-physical landscape change in modelling would help in better management of the forested landscapes in the country .

From the understanding of the reviewed models, LandSHIFT has the capabilities to perform optimally except its present functionality at higher scale (regional). The integrated modelling approach of LandSHIFT facilitates the assessment of both the socio-economic and ecological perspectives, such as - demand on labour, capital, farm land, and activity for income generation of farmers; biodiversity, soil quality, erosion, water quality, nutrient cycling, and air pollution, ecological resource flow and several other variables. To apply LandSHIFT at a smaller spatial scale than it has been used so far, in managing landscape at micro-landscape level, efficient conservation of protected area or sustainable urban planning would require technical modification. LandSHIFT provides a framework for the combination of biophysical and socio-economic information with geographical information on land use and land cover and integrates the information generated in different modelling approaches.

As the very important aspect of the study to develop the land-use model for ex ante analysis in the field of ecological modelling which is so far lacking in viewpoint of when there is need to invite the scientists/researchers from multiple disciplines to work together. The land-use management in current climate dilemma and sustainable landscape development is the prime concern of policy planners across countries. Therefore, the other major goal of future research should be to provide appropriate solution to the farmers, policymakers, and decision takers while practically evaluating the consequences both explicit and implicit of the proposed solutions in economic and ecological terms. Presently, no practical example exists that could be cited where integrated and interdisciplinary modelling approach has been used in the country. Therefore, current research efforts could be considered as one of the first steps being taken in this direction for the evaluation of landscape sustainability in India. In order to achieve the desired results, the research is focussing on the following specific landscape elements.

Database on land-use/land-cover change and important driving factors in rural agricultural, urban, and protected landscape.

- Documentation of TEK and peoples' experiences about the pattern of climate change and its impact on forest, agriculture, livestock, and humans.
- To plan possible adaptation measures for vulnerability to climate change towards sustainable landscape development and socio-economic upliftment of the rural people.
- Effect of climate on seasonal variability and reliability and climate extremes affecting agriculture production, forestry, and water resources at micro-level.
- Simulation models explaining the changes due to change in climate, urban expansion, and protected areas, and information enabling sustainable development strategies for policy formulation and designing and developing the sound database for future research and policy intervention.
- Catalogue of adapted land-use management systems and socio-economic development.
- Identification of areas/themes for further research.

16.9 Use of LandSHIFT in Indian Context

LandSHIFT is the first large-scale model that puts the emphasis on a detailed representation of the competitions for natural resources between the major land-use sectors such as settlement and industrial, agricultural, and forestry. Technical modification in the model structure to operate at micro-scale in the Indian context is the need of the hour. Estimation of such a model would require a variety of spatially articulated area-specific variables (land characteristics like slope and elevation) and locational characteristics (proximity to amenities) to regulatory features (zoning or planning policies). LandSHIFT has been successfully validated and applied in Indian context.

There are two versions of LandSHIFT that operate at two different scales of spatial resolutions. One version operates for global extent which has a spatial resolution of 5arc (approximately 10 km at the equator). The other version (LandSHIFT.R) has spatial resolution of 1 km which is referred as regional scale model. The model has been authenticated to be used in Indian conditions (Schaldach et al. 2011; Gupta 2009) and has the scope and flexibility to be modified for finer spatial resolution and assimilation of empirical data to study landscape with more specified characteristics. In a pilot study, Schaldach et al. (2011) used the LandSHIFT model to assess the impact of biofuel production on land-use change in India up to the year 2030. The case study aimed at the simulation of land-use change and its relation to other global change processes. It explicitly states the competition among land-use activities such as human settlement, biofuel production, wood production, and food production and the resulting effects on the spatial extent of natural land. It captures land suitability and land constraints, and thus, it delivers an accurate picture of land evolution process and provides a reasonable estimate of land available for various purposes (Couleau et al. 2015) which provide the future possible changes in a more pragmatic manner. Future land-use change in these models is determined by statistical relationships of past trends in land use. LandSHIFT also permits integration of other functional model components representing different aspects of land-use change (Gupta 2009) and integrated analysis of drivers of land-use change. In the model simulation for development scenarios, a comprehensive linkage is made between driving forces and policies that will affect the land use/land cover in the future.

16.10 Integration of Subsystems in Landscape Modelling

Agent-based models have emerged in land and environmental science as a way to better capture complex system characteristics of coupled socioecological systems. ABMs for socio-economic systems have evolved as extensions of other modelling techniques, including analytical and statistical modelling, cellular automata, artificial learning, and others (Tatiana et al. 2013). One advantage of using ABM is that it allows to model these systems not only using traditional mathematic and statistics but also using behavioural information, for which other computational and economic models have been criticized (Heppenstall et al. 2012). There has been extensive research on land at disintegrated level which results in availability of spatially referenced small-scale (parcel/farm) data with immense use of statistical and simulation model. An agent-based model of land-use/cover change (ABM-LUCC) is an effective tool for exploring agent diversity and landscape heterogeneity within a spatially explicit structure (Evans and Kelly 2004). To develop an efficient framework for landscape planning, the following steps need to be considered:

1. Select a computationally feasible model for spatio-temporal land-use change modelling.
2. Robust estimation by controlling parameters which will be context-specific (e.g. urban, protected area, farm).
3. Integrate an Agent-Based Module to the spatially explicit model.
4. Further development of model for scenario analysis with most effective use of computational model for strategy formulation for sustainable development.

Behaviour of a landscape depends on the aggregated behaviour of the agents in the landscape. The agents can be human actors in the landscape in various forms, institutions or environment. Human subsystem can be integrated as actors which gives the likelihood of changes. Landscape decisions are to be driven by the actors influenced by institution and environment. Cellular automata and statistical approaches have many difficulties in dealing with complex behaviour and decision-making, which cannot be explained satisfactorily by transition rules (Parker et al. 2003). Multi-agent models simulate decision-making by individual agents of land-use change explicitly addressing interaction among individuals (Reed et al. 2013). In recent time, agent-based models have emerged as a new tool for tackling the complex and multifaceted challenges of developing integrated scenarios of LULCC. At smaller landscape level, such hybrid model would have particular advantage in participatory context which will lead to bottom-up planning for sustainable landscape management. It will help in linking micro-scale decision to macro-scale phenomena.

Feedbacks from actors' decision-making on land use also need to be incorporated. The actors (farmers as in the example—Fig. 16.3) are continuously influenced by institutions and environment, and in discourse to that consequence, the constraints and opportunities for the land-use change are to be assessed by counting on the feedback from actors and environment.

For integration of models to put human subsystem (actor agents) in the modelling framework, data from household survey are required (e.g. SYPRIA; Schaldach et al. 2011) which can be supported by extensive empirical studies at various locations in any urban, village, or protected area settings. A comprehensive decision-making process should be programmed considering all the agents for land-use change at micro-scale (Fig. 16.4).

16.11 Conclusion

Land-use/land-cover change is a process result of interactions determined by anthropogenic behaviour in human-dominated landscapes. With the range of socio-economic and geo-climatic variation in India, it is far more difficult to model land-use change in national or regional scale. Incorporation of the fine-scale dynamics is difficult due to the high data requirements. However, it is possible to study the importance of these fine-scale dynamics and its translation to the regional level in case-studies, nested within the larger region. Integrated modelling systems should include information from more than one discipline and have the ability to organize the information in a modularized programme structure and link scientific findings with policy analysis.

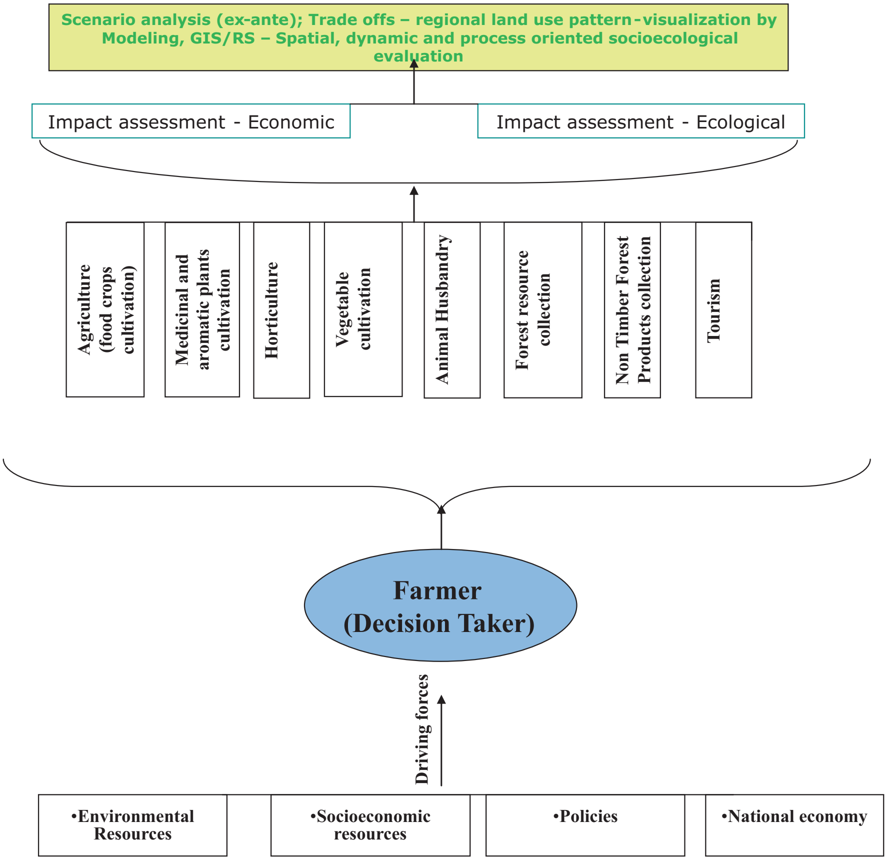

Fig. 16.3 Theoretical background Actor-agent (farmers) in rural landscape in India. (Nautiyal 2011)

At continental and regional scale, LandSHIFT is an integrated modelling system which provides framework for the combination of bio-physical and socio-economic information with geographical information on land use and land cover and integrates this information, generated with different modelling approaches. It facilitates a flexible exchange of functional modules and easy extension with new modules from the rapidly advancing field of land-use research and hence meets the challenge of semantical and technical integration of different modelling approaches within a common conceptual and software framework (Koch et al. 2008). The LandSHIFT model operates with three modules, viz. Bio-physical, Socio-economic, and Land-Use Change module. It also offers flexibility with regard to input data and scenario formulation. Further modification of the model to make it apt for applying in landscape level or farm level analysis will augment its suitability in micro-scale

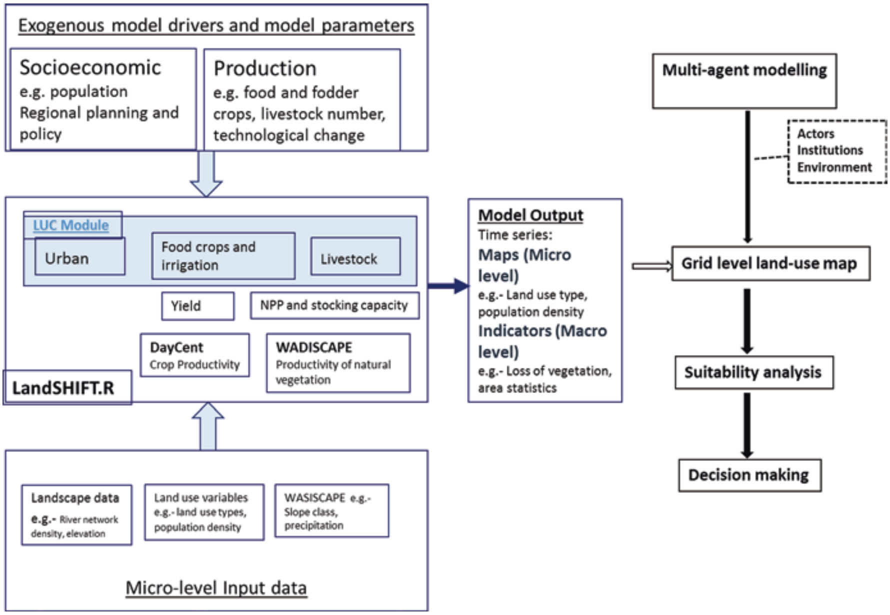

Fig. 16.4 Integrated landscape modelling framework. (Modified from LandShift. R Framework, Koch et al. 2008)

landscape planning in Indian context. Exploration of the scope for integration of Agent-Based Modelling would be significant in the context of representing human decision-making in socioecological system models.

References

Agarwal C, Glen GM, Grove JM, Evans TP, Schewik CM (2002) A review and assessment of land-use change models: dynamics of space, time, and human choice. In: GTR NE-297. U.S.D.A. Forest Service, Northeastern Research Station, Newton Square, p 61

Alberti MJ (2003) Integrating humans into ecology: opportunities and challenges for studying urban ecosystems. Bioscience 53(12):1169–1179

Alcamo J, Schaldach R (2006) Landshift: global modelling to assess land use change. In: Tochermann K, Scharl A (eds) EnviroInfo 2006: managing environmental knowledge. Proceedings of the 20th international conference on informatics for environmental protection. Shaker Verlag GmbH, Aachen, pp 223–230

Alcamo J, Schaldach R, Koch J, Kolking C, Lapola D, Priess J (2011) Evaluation of an integrated land use change model including a scenario analysis of land use change for continental Africa. Environ Model Softw 26:1017–1027

Bharath HA, Ramachandra TV (2013) Measuring urban sprawl in tier II cities of Karnataka, India. In: Proceedings of IEEE GHTC SAS 2013, conference. Technopark, Trivandrum, pp 23–24

Clifford NJ (2008) Models in geography revisited. Geoforum 39(2):675–686

Couleau A, Laborde D, Tokgoz S (2015) The MIRAGE-Biof model in the Govila Project: Implementation strategy. Online Access: http://www.govila.tu-darmstadt.de/media/govila/publikationen_4/IFPRI_-_Technical_Paper_I-2.pdf

Crossman BBA (2009) Identifying cost-effective hotspots for restoring natural capital and enhancing landscape multifunctionality. Ecol Econ 68:654–668

Curran PJ, Atkinson PM (1998) Remote sensing and geo-statistics. Prog Phys Geogr 22(1):61–78

d'Aquino P, Page C. Le, Bousquet F, Bah A (2002) Self-designed role-playing game and multi-agent system to empower a local decision-making process on land use management: The self-cormas experiment in Senegal. Presented at the International Society of Ecological Economics (ISEE), "Models, Role Games and Negotiations," Sousse, Tunisia, March 6–9

DFID (1999) Sustainable livelihoods guidance sheets, Department for International Development, London. Online Access: www.livelihoods.org/info/info_guidanceSheets.html#6

Elias P, Dekolo S, Babatola O (2012) Land use change modelling in developing countries: issues and prospects. Int J of Geogr Geol (1):23–41

Evans TP, Kelly H (2004) Multi-scale analysis of a household level agent-based model of land-cover change. Environ Manag 72(2):57–72

Fazal S (2000) Urban expansion and loss of agricultural land – a GIS based study of Saharanpur City, India. Environ Urban 12(2):133–149

Forman RTT, Godron M (1986) Landscape ecology. Wiley, New York, p 620

Garrigues S (2004) Spatial heterogeneity from remote sensing observations of land surface: Characterization and effects on biophysical variable estimates. Thesis, Ecole Nationale Supérieure Agronomique de Rennes, France, p 360

Geist HJ, Lambin EF (2001) In What drives tropical deforestation? A meta-analysis of proximate and underlying causes of deforestation based on subnational scale case study evidence. LUCC Report Series No. 4. University of Louvain, Louvainla-Neuve, Belgium, p 136

GLP (2005) Global land project science plan and implementation strategy. IGBP Report No. 53/IHDP Report No. 19. 64 p

Grimm NB, Foster D, Groffman P, Grove JM, Hopkinson CS, Nadelhoffer KJ, Diane E, Peters DPC (2008) The changing landscape: ecosystem responses to urbanization and pollution across climatic and societal gradients. Front Ecol Environ 6(5):264–272

Gupta S (2009) A modeling analysis of changing global land use as affected by changing demands for wood products, Thesis, Universität Kassel, Kassel. Zur Erlangung des akademischen Grades des Doktors der Ingenieurwissenschaften (Dr. -Ing.) im Fachbereich Elektrotechnik/Informatik der Universität Kassel. http://pubman.mpdl.mpg.de/pubman/item/escidoc:2389822/component/escidoc:2389821/Gupta_Thesis.pdf

Haase G, Richter H (1983) Current trends in landscape research. GeoJournal 7:107–119

Haines-Young R, Chopping M (1996) Quantifying landscape structure: a review of landscape indices and their application to forested landscapes. Prog Phys Geogr 20(4):418–445

Hasegawa T, Fujimori S, Ito A, Takahashi K, Masui T (2017) Global land-use allocation model linked to an integrated assessment model. Sci Total Environ 580:787–796

Henseler M, Alexander W, Herrmann S, Krimly T, Dabbert S (2009) Modeling the impact of global change on regional agricultural land use through an activity-based non-linear programming approach. Agric Syst 100:31–42

Heppenstall AJ, Crooks AT, See LM, Batty M (eds) (2012) Agent-based models of geographical systems. Springer, New York, 760p

Huigen M (2003) Agent based modeling in land-use and land-cover change studies. IIASA Interim Report. IIASA, Laxenburg, Austria: IR-03-044. http://pure.iiasa.ac.at/id/eprint/7039/

IEEP, Alterra, Ecologic, PBL and UNEP-WCMC (2009) Scenarios and models for exploring future trends of biodiversity and ecosystem services changes. Final report to the European Commission, DG Environment on Contract ENV.G.1/ETU/2008/0090r. Institute for European Environmental Policy, Alterra Wageningen UR, Ecologic, Netherlands Environmental Assessment Agency, United Nations Environment Programme World Conservation Monitoring Centre

Kickert RN, Tonella G, Simonov A, Krupa SV (1999) Predictive modeling of effects under global change. Environ Pollut 100:87–132

Koch J, Schaldach R, Kochy M (2008) Modeling the impacts of grazing land management on land-use change for the Jordan River region. Glob Planet Chang 64:177–187

Kumar A, Pandey AC, Hoda N, Jeyaseelan AT (2011) Evaluation of urban sprawl pattern in tribal dominated cities of Jharkhand state, India. Int J Remote Sens 32(22):7651–7675

Lambin EF (1997) Modelling and monitoring land-cover change processes in tropical regions. Prog Phys Geogr 21(3):375–393

Laniak GF, Olchin G, Goodall J, Voinov A, Hill M, Glynn P, Whelan G, Geller G, Quinn N, Blind M, Peckham S, Reaney S, Gaber N, Kennedy R, Hughes A (2013) Integrated environmental modeling: a vision and roadmap for the future. Environ Model Softw 39:3–23

Lapola DM, Schaldach R, Alcamo J, Bondeau A, Koch J, Koelking C, Priess JA (2010) Indirect land-use changes can overcome carbon savings from biofuels in Brazil. Proc Natl Acad Sci 107:3388–3393

Mallupattu PK, Reddy S, Reddy J (2013) Analysis of land use/land cover changes using remote sensing data and GIS at an Urban Area, Tirupati, India. Sci World J 2013:1–6. https://www.hindawi.com/journals/tswj/2013/268623/abs/

Manson SM (2005) Agent-based modeling and genetic programming for modeling land change in the Southern Yucatan Peninsular Region of Mexico. Agric Ecosyst Environ 111(1):47–62

Mathews RB, Gilbert NG, Roach A, Polhill JG, Gotts NM (2007) Agent-based land-use models: a review of applications. Landsc Ecol 22(10):1447–1459

Mauldin TE, Plantinga AJ, Alig RJ (1999) Determinants of land use in Maine with projections to 2050. North J Appl For 16:82–88

McGregor D, Simon D (eds) (2012) The peri-urban interface: approaches to sustainable natural and human resource use, 1st edn. Routledge, London, p 272

MEA (2005) Ecosystems & human Well-being: synthesis (millennium ecosystem assessment). Island Press, Washington, DC

Menon S, Bawa KS (1997) Applications of geographic information systems, remote-sensing, and a landscape ecology approach to biodiversity conservation in the Western Ghats. Curr Sci 73(2):134–145

Moghadam HS, Helbich M (2013) Spatiotemporal urbanization processes in the megacity of Mumbai, India: a Markov chains-cellular automata urban growth model. Appl Geogr 40:140–149

Narain V (2009) Growing city, shrinking hinterland: land acquisition, transition and conflict in peri-urban Gurgaon, India. Environ Urban 21(2):501–512

Nautiyal S (2011) Can conservation and development interventions in the Indian Central Himalaya ensure environmental sustainability? A socioecological evaluation. Sustain Sci 6(2):151–167

Nautiyal S, Rao KS, Kaechele H, Zander P (2010) Conceptual model development for landscape management in the mountains of the Indian Himalayan region: an approach for sustainable socio-ecological development. Landsc Online 18:1–19

Paassen V, Roetter RP, Keulen HV, Hoanhe CT (2007) Can computer models stimulate learning about sustainable land use? Experience with LUPAS in the humid (sub) tropics of Asia. Agric Syst 94:874–887

Pandey B, Seto KC (2015) Urbanization and agricultural land loss in India: comparing satellite estimates with census data. J Environ Manag 148:53–66

Parker DC, Manson SM, Janssen MA, Hoffmann MJ, Deadman P (2003) MultiAgent systems for the simulation of land-use and land-cover change: a review. Ann Assoc Am Geogr 93(2):314–337

Pearson LJ, Park S, Harman B, Heyenga S (2010) Sustainable land use scenario framework: framework and outcomes from peri-urban south-east Queensland, Australia. Landsc Urban Plan 96:88–97

Peter E, Dekolo D, Babatola O (2014) Land use change modelling in developing countries: issues and prospects. Int J Geogr Geol 1(1):23–41

Peterson GD, Cumming GS, Carpenter SR (2003) Scenario planning: a tool for conservation in an uncertain world. Conserv Biol 17(2):358–366

Ramachandra TV, Aithal BH, Vinay S (2013) Land use land cover dynamics in a rapidly urbanising landscape. SCIT J 13:1–12

Reed MS, Arblaster K, Bullock C, Burton RJF, Davies AL, Holden J, Hubacek K, May R, Mitchley J, Morris J, Nainggolan D (2009) Using scenarios to explore UK upland futures. Futures 41(9):619–630

Reed MS, Kenter J, Bonn A, Broad K, Burt TP, Fazey IR, Fraser EDG, Hubacek K, Nainggolan D, Quinn CH, Stringer LC (2013) Participatory scenario development for environmental management: a methodological framework illustrated with experience from the UK uplands. J Environ Manag 128:345–362

Ritzema H, Froebrich J, Raju R, Sreenivas C, Kselik R (2010) Using participatory modelling to compensate for data scarcity in environmental planning: a case study from India. Environ Model Softw 25:1450–1458

Roettera RP, Hoanhb CT, Laborteb AG, Keulenc H, Van IMK, Dreisere C, Diepena CA, De Ridderc N, Laar HH (2005) Integration of Systems Network (SysNet) tools for regional land use scenario analysis in Asia. Environ Model Softw 20:291–307

Roy PS, Behera MD (2005) Assessment of biological richness in different altitudinal zones in the Eastern Himalayas, Arunachal Pradesh, India. Curr Sci 88(2):321–330

Roy PS, Giriraj A (2008) Land use and land cover analysis in Indian context. J Appl Sci 8:1346–1353

Schaldach R, Priess JA (2008) Integrated models of the land system: a review of modelling approaches on the regional to global scale. Living Rev Landsc Res 2:1–34

Schaldach R, Alcamo J, Heistermann M (2006) The multiple-scale land use change model LandShift: A scenario analysis of land use change and environmental consequences in Africa. In: Voinov A, Jakeman AJ, Rizzoli AE (eds) Proceedings of the iEMSs third biennial meeting 'summit on environmental modelling and software', 9th July 2006, Burlington, USA

Schaldach R, Alcamo J, Koch J, Kolking C, Lapola DM, Schungel J (2011) An integrated approach to modelling land-use change on continental and global scales. Environ Model Softw 26:1041–1051

Schaldach R, Koch J, aus der Beek T, Kynast E, Florke M (2012) Current and future irrigation water requirements in pan-Europe: a comparative analysis of influencing factors. Glob Planet Chang 95:33–45

Sudhira HS, Ramachandra TV, Jagadish KS (2004) Urban sprawl: metrics, dynamics and modelling using GIS. Int J Appl Earth Obs Geoinf 5:29–39

Swetnam RB (2010) Mapping socioeconomic scenarios of land cover change: a GIS model to enable ecosystem service modelling. Environ Manag 92(3):563–574

Tatiana F, Verburg PH, Parker DC, Stannard CA (2013) Spatial agent-based models for socio-ecological systems: challenges and prospects. Environ Model Softw 24:1–7

Tucker GM, McConville AJ, McCoy, Alterra P, Wageningen UR, Braat L et al (2009). Scenarios and models for exploring future trends of biodiversity and ecosystem services changes; Contract ENV.G.1/ETU/2008/0090r., Brussels

Turner MG, Gardner RH, O'Neill RV (2001) Landscape ecology in theory and practice: pattern and process. Sringer, New York, p 401

Upadhyay TP, Solberg B, Sankhaya PL (2006) Use of models to analyse land-use changes, forest/soil degradation and carbon sequestration with special reference to Himalayan region: a review and analysis. Forest Policy Econ 9(4):349–371

Verburg PH (2006) Simulating feedbacks in land use and land cover change models. Landsc Ecol 21:1171–1183

Verburg PH, Schot PP, Dijst MJ, Veldkamp A (2004) Land use change modelling: current practice and research priorities. Geo J 61(4):309–324

Verburg PH, Tabeau A, Hatna E (2013) Assessing spatial uncertainties of land allocation using a scenario approach and sensitivity analysis: a study for land use in Europe. J Environ Manag 127:132–144

Way Forward

17

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The International Society for Tropical Ecology (ISTE) organised the 14th Tropical Ecology Congress (TEC) at Jawaharlal Nehru University, New Delhi, in December 2014 on the theme of “Tropical Ecosystems in a Changing World”. During the congress, ecologists from across the world critically evaluated the status of the current research vis-à-vis global change and deliberated on developing a broad road map for future research on tropical ecosystems. The present volume could be considered as a small step forward towards this research agenda in the tropics. Though the rationale behind bringing forth this volume is manifold, the major one is to present the contemporary insights on the impacts of global change factors, especially that of climate change, on tropical forests in India, Nepal and Indonesia and more so in the Himalaya. We hope that important scientific information presented in the book will serve not only as a baseline for finding solutions to a range of policy-related questions—such as what are the impacts of climate change, biological invasion, biodiversity loss and forest degradation on the structural and functional integrity of studied tropical ecosystems—but also help in identifying the knowledge gaps that impinge on the sustainable management of ecosystems in the tropics.

The present CO_2 concentration in the atmosphere has touched the highest level throughout human history. If current trends in emissions continue, CO_2 concentration is expected to reach nearly twice (~500 ppm) the preindustrial level by 2050. It may result in an increase in the mean global atmospheric temperature by 3–7 °C, and resultant change in climate will likely have huge ecological and socio-economic costs to societies across the world. Already, the impacts of global warming on ecosystems are perceptible and have been elucidated well in the most recent IPCC special report released in October 2018. In the tropics, the global warming scenarios predict high levels of threat to biodiversity hotspots, coral reefs, fresh water and

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glaciers in the immediate future. Thus global change which not only includes climate change but also anthropogenic shifts such as growing populations, resource exploitation, altered disturbance regimes, biological invasion, habitat loss and extinctions will keep impacting the tropical ecosystems simultaneously. Agricultural intensification and extensification will continue to be the dominant driver of global change in the tropics for the next several decades. The combined impact of climate and land-use changes will keep on reinforcing other drivers of global change and can transform the basic structure and functioning of forests which are constantly being overexploited leading to potentially irreversible consequences for the environment and human well-being in developing tropics.

Though the scientific efforts made so far have greatly enhanced our understanding on the structural and functional attributes of tropical ecosystems, future research may focus on adopting an integrated approach that will help reduce risks of that of multiple global change factors in general and climate change and deforestation to natural and human-managed ecosystems in particular. For instance, research results that help avoid deforestation, support restoration of degraded forests, strengthen agroforestry, sustain biodiversity management, conserve wetland ecosystems, take advantage of traditional knowledge systems, and explore the critical thresholds in the wake of projected climate change have been advocated (IPCC 2018). Generating long-term data sets on various structural and functional attributes of tropical ecosystems and promoting use of appropriate technologies and modelling in ecological research are yet other dimensions of future research for developing suitable conservation strategies to respond to the emerging challenges.

Over the whole world, there is an explicit recognition that conservation of natural ecosystems and sustainable livelihoods are mutually reinforcing. However, finding solutions that help maintain a critical balance between the two is an enormous challenge. This situation gets all the more complex when the level of understanding on the dynamics of ecosystems and social systems at landscape scale is still insufficient. Ecologists, sociologists and economists have much to gain from each others' understandings, but so far examples of taking advantages from such collaborations are negligible. There is plenty of scope for taking up transdisciplinary research in tropics that will help prove the merits of considering ecology in decision-making and utilizing its knowledge for socio-economic development and environmental protection. In this regard, it is important to take note of emerging understanding wherein explicit analogies have been drawn between processes of loss of biodiversity and traditional ecological knowledge (TEK): as the former weaken the stability of ecosystems, the latter have similar implications for socio-cultural systems. Research in the last few decades has highlighted the potential contribution of TEK in building flexibility in socioecological systems, thereby enabling local communities to secure their livelihood and manage periodic disturbances in ecosystems. Since TEK and socioecological systems evolve synchronously, it can help local societies in achieving multiple conservation objectives for sustainable management of natural resources and hence playing a vital role in adapting to global change successfully. TEK can facilitate multidisciplinary, transdisciplinary and participatory research approaches that not only help in its preservation and protection but also in minimizing erosion of the local knowledge base. As the loss of TEK and biodiversity is not commonly exclusive, it is imperative to make efforts not only to protect biodiversity but also the cultural diversity for sustainable management of tropical ecosystems. Therefore, research on erosion and change of TEK in different socio-ecological milieus is the need of the hour.

In addition to the aforementioned priority areas of research, the ISTE in collaboration with other conservation organisations and initiatives mentioned in the introductory chapter also supports global change research on the priority ecosystems such as tropical wetlands and mountains. For example, though covering only 6% of the global surface, the social, environmental and cultural value of marine, inland and man-made wetlands is widely recognized. They are not only vital for water purification but also most productive ecosystems and provide habitat to a range of unique floral and faunal species. Wetlands supply ecosystem services which include food, water and recreation, among many others. Tropical wetlands still support the livelihoods of a large number of indigenous societies which have managed them sustainably until very recently. However, several global change drivers such as such pollution, encroachments, land-use change, weed infestation, siltation and in most recent times climate change are responsible for an alarming rate of loss of wetlands across the globe. Climate change will have huge impacts on the structural and functional performance of wetland ecosystems, but so far these systems have not been given the desired level of research importance from this standpoint. Conservation of wetlands is critical for responding to the challenges of climate change, and therefore enhanced scientific understanding is needed at all scales for protection, management and restoration of wetlands especially in tropics. Similarly the mountain ecosystems support nearly 14% of global population, 25% of the terrestrial biodiversity and half of the biodiversity hotspots, and nurturing rich cultural diversity also becomes important from global change research viewpoint. Mountains are important sources of many ecosystem goods and services, particularly fresh water for nearly half of the global population. Considering the fact that mountain ecosystems are largely fragile which are afflicted with chronic disturbances and hence degrading fast under the influence of global change, future research can focus on generating knowledge to help bring in “mountain perspectives” into sustainable development approaches factoring in climate change.

Social and ecological issues are inextricably linked in the tropics. Therefore, useful research should be interdisciplinary and transdisciplinary. Ecological research in the tropics should be practical. For this ecologists must continue to engage with local communities as well as the decision-makers to enhance cooperation among the stakeholders across tropics. This will help in developing and refining a more targeted research agenda about priority areas of ecological research for the region listed above. Researchers may also place emphasis on region- and ecosystem-specific research priorities that contribute towards achieving the Sustainable Development Goals (SDGs). Given limited resources for research and extraordinary challenges to society, it is important for researchers, managers, officials and rural people to work together to develop and test conservation approaches. The evolution of ecology as a multidisciplinary subject during recent decades certainly helps researchers in this regard. We need research that contributes to better protection, management and restoration of tropical ecosystems.

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