Diversity and distribution of bird and butterfly communities in the indigenous farming systems of Sikkim

A Thesis Submitted

To

Sikkim University

In Partial Fulfilment of the Requirement for the

Degree of Doctor of Philosophy

By

Kishor Sharma

Research Supervisor

Dr. Bhoj Kumar Acharya, Associate Professor

Department of Zoology

School of Life Sciences

December 2021

Dedicated to my beloved Parents

Shri. Bidya Sharma & Smt. Tika Devi

For fostering childhood interest in nature, letting me chase my dream, greatly valuing higher education, believing in me and wholeheartedly supporting me to pursue and complete Ph.D. research despite facing many hurdles along the way, for which I will be forever grateful.

6 माइल, सामदुर, तादोंग - 737102
गंगटोक, सिक्किम, भारत
फोन-03592-251212, 251415, 251656
टेलीफैक्स - 251067
वेबसाइट - www.cus.ac.in

6th Mile, Samdur, Tadong-737102
Gangtok, Sikkim, India
Ph. 03592-251212, 251415, 251656
Telefax : 251067
Website : www.cus.ac.in

सिक्किम विश्वविद्यालय
SIKKIM UNIVERSITY
(भारत के संसद के अधिनियम द्वारा वर्ष 2007 में स्थापित और नैक (एनएएसी) द्वारा वर्ष 2015 में प्रत्यायित केंद्रीय विश्वविद्यालय)
(A central university established by an Act of Parliament of India in 2007 and accredited by NAAC in 2015)

CERTIFICATE

This is to certify that the Ph.D. thesis entitled "**Diversity and distribution of bird and butterfly communities in the indigenous farming systems of Sikkim**" submitted to **Sikkim University** in partial fulfilment of the requirement for the **Degree of Doctor of Philosophy in Zoology**, embodies the original research work carried out by **Kishor Sharma** at the Department of Zoology, School of Life Sciences, Sikkim University. It is a bonafide investigation carried out and completed by him under my supervision and guidance. He has followed the rules and regulations prescribed by the University. The results are original and have not been submitted anywhere else for any other degree or diploma.

It is recommended that this Ph.D. thesis be placed before the examiners for evaluation.

Bhoj Acharya

Dr. Bhoj Kumar Acharya

अध्यक्ष
Head
प्राणी विज्ञान विभाग
Department of Zoology
सिक्किम विश्वविद्यालय
Sikkim University

Ph.D. Supervisor
Associate Professor & Head,
Department of Zoology,
School of Life Sciences,
Sikkim University,
Gangtok, Sikkim, India.

6 माइल, सामदुर, तादोंग - 737102
गंगटोक, सिक्किम, भारत
फोन-03592-251212, 251415, 251656
टेलीफैक्स - 251067
वेबसाइट - www.cus.ac.in

6th Mile, Samdur, Tadong-737102
Gangtok, Sikkim, India
Ph. 03592-251212, 251415, 251656
Telefax : 251067
Website : www.cus.ac.in

सिक्किम विश्वविद्यालय
SIKKIM UNIVERSITY
(भारत के संसद के अधिनियम द्वारा वर्ष 2007 में स्थापित और नैक (एनएएसी) द्वारा वर्ष 2015 में प्रत्यायित केंद्रीय विश्वविद्यालय)
(A central university established by an Act of Parliament of India in 2007 and accredited by NAAC in 2015)

Ref No:-SU/DEPT OF ZOO/2021/277 **Date:-25.11.2021**

PLAGIARISM TEST CERTIFICATE

This is to certify that plagiarism check has been carried out for the following Ph.D thesis with the help of **URKUND SOFTWARE** and the result is 1 % tolerance rate, within the permissible limit (below 10% tolerance rate) as per the norm of Sikkim University.

"Diversity and distribution of bird and butterfly communities in the indigenous farming systems of Sikkim"

Submitted by **Mr. Kishor Sharma** under the supervision of **Dr.Bhoj Kumar Acharya**, Associate Professor & Head, Department of Zoology, Sikkim University

Kishorsharma
Kishor Sharma
Roll no: 16PDZ002
Signature of the candidate

Dr.Bhoj Kumar Acharya
Associate Professor & Head
Department of Zoology,
School of Life Sciences,
Sikkim University,
Gangtok

अध्यक्ष
Head
प्राणी विज्ञान विभाग
Department of Zoology
सिक्किम विश्वविद्यालय
Sikkim University

Dr. Shri Ram
Librarian
Central Library
Sikkim University

पुस्तकालयाध्यक्ष
Librarian
केन्द्रीय पुस्तकालय Central Library
सिक्किम विश्वविद्यालय
Sikkim University

DECLARATION OF THE CANDIDATE

I, Kishor Sharma, hereby declare that the present Ph.D. thesis entitled **"Diversity and distribution of bird and butterfly communities in the indigenous farming systems of Sikkim"** submitted in partial fulfilment of the degree of Ph.D. in Zoology, is my original work and has not previously formed the basis for the award of any degree, diploma, associateship, fellowship or any other similar title.

KishorSharma
....

Kishor Sharma
Registration No.:16/Ph.D./ZOO/02
Ph.D. Scholar
Department of Zoology
School of Life Sciences
Sikkim University
Gangtok, Sikkim, India

Place: Gangtok, Sikkim

Date: 13 December, 2021

ACKNOWLEDGEMENTS

I would like to express my sincere gratitude to my Ph.D. Supervisor Dr. Bhoj Kumar Acharya, Associate Professor and Head, Department of Zoology, School of Life Sciences, Sikkim University (SU) for the continuous guidance and support throughout my Ph.D. research, for his patience, motivation and sharing immense knowledge in study design, field sampling, data analysis, and scientific writing. I am grateful for recommending and facilitating my participation in specialized training workshops, conferences and research visit at reputed Universities/Institutes from within the country and abroad. Within few hours of reaching Sikkim, I was in the field with Dr. Acharya, Dr. Basundhara Chettri, and their research team, which provided a great start to my research in Sikkim Himalaya. Over the years, he kept me close by, constantly guided me with warmth, gave multiple opportunities to overcome the shortcomings, and ensured that my Ph.D. research got completed. With a calm nature, kind-heartedness and strong hold on the subject, Dr. Acharya provided the ideal conditions for me to conduct the present ecological study in the Eastern Himalaya. I could not have imagined having a better Supervisor and mentor for my Ph.D. study. I am also thankful to Acharya sir, Basundhara Madam, and the whole family for warmly welcoming me at their homes and providing various supports till now.

Besides my Ph.D. Supervisor, I am also very grateful to all Doctoral Committee Members, Dr. Basundhara Chettri (Former Head & Assistant Professor), Dr. Bisu Singh (Assistant Professor), and Dr. Sudeep Ghatani (Assistant Professor), Department of Zoology, SU for their insightful comments, continued encouragements and supports during the tenure of Ph.D. research. I am also grateful to Dr. K. Birla Singh (Former Head & Associate Professor), Dr. M.P. Thapa (Guest faculty), Department of Zoology, SU for the encouragements and continued support. I record my gratitude to Prof. T.B. Subba (former Vice Chancellor, SU), Dr. Basundhara Chettri and Dr. Acharya for giving me an opportunity to teach the M.Sc. students briefly in 2013 and 2016. It was a great learning experience for me to teach those bright students, wherein I could share subject papers with Dr. Singh and Dr. Ghatani, who were very supportive to me, for which I am grateful to them. I also thank Mr. Bedanta P. Saikia (Technical Assistant) for his various helps during the Ph.D. tenure.

I express my deep gratitude to Dr. Ghanashyam Sharma (Head & Program Manager), The Mountain Institute-India for the research opportunity, guidance, and encouragements and recommending my participation in training workshops especially during the initial phase of my Ph.D. With his positivity, kindness and sound research expertise Dr. Sharma, guided and supported me both within the institute and outside. The warmth and full support received from Sir and Madam Sharma, and their whole family, allowed me to settle down, smoothly conduct field works, and deal with many challenges, for which I am grateful to them.

I also convey my sincere gratitude to the Vice Chancellor, SU; Registrar, SU; Dean, School of Life Sciences, SU; Dean, Student Welfare, SU; Prof. Jyoti Prakash Tamang (Head, Department of Microbiology, SU and former officiating Vice Chancellor, SU); Prof. Dhaniraj Chettri, Prof. Shanti S. Sharma, Dr. Arun Chettri, Dr. S.K. Rai (Department of Botany, SU), Dr. Niladri Bag (Head, Department of Horticulture, SU), Dr. Nagendra Thakur (Department of Microbiology, SU) for being supportive during my Ph.D. I am thankful to Chief Librarian, Information Scientist and other staffs at Central Library, SU for library facilities and help to run plagiarism tests; Medical officer and nurses at SU health centre for taking care of ailments over the years; and drivers for transportation. I also thank Dr. S.K. Gurung, Jt. Registrar, Academics, SU and all other Administrative officials and staffs of SU for their help, especially for scholarship processing.

For research funding at different stages of my Ph.D. research, I am grateful to the Department of Science and Technology, Government of India (Grant no.: SERB/AS/2011/11); University Grants Commission, New Delhi for the Junior Research Fellowship (Grant No.: 3520/(NET-JULY2016); Sikkim University for the Non-NET fellowship; and Department of Education, BTC, Assam for the Cash Award & Incentives to meritorious students. I also Forests, Environment and Wildlife Management Department (FEWMD), Government of Sikkim (GoS) especially PCCF-cum-Secretary; Research Evaluation and Monitoring Committee (REMC) and State Biodiversity Board for research permission (Permit no.: 24/GOS/FEWMD/BD-R-2014(Part-III)/CF(WL)/137;78/GOS/FEWMD/BD-R-2015/CCF(T&HQ)342) to undertake this study. At the FEWMD, GoS, I record my gratitude to Shri. M.L. Srivastava (ACS and PCCF-cum-Secretary); Dr. Thomas Chandy (former PCCF-cum-Secretary); Mrs. Usha Ganguli-Lachungpa, Dr. Bharat K. Pradhan, and Ms. Sunita Khatiwada. I thank Sikkim University and The Mountain Institute-India for providing excellent research facilities.

For offering overwhelming support, hospitality and access to their farms during field sampling, I am grateful to the local communities of East district (Bering-Tareythang, Sang-Khamdong, Pendam, Luing) and south district (Lingee-Payong, Sumbuk) of Sikkim. I am especially thankful to Smt. Tara Devi Acharya and family at Lingee-Payong, Dr. Shiva Kumar Sharma and family at Sumbuk, Shri. Ram C. Tiwari and family at Lingee Karjee, Smt. Krishna Neopaney and family at Sang Busty, Smt. Khinamaya Khatiwada and family at Sumik Khamdong, Shri. Y.P. Bastola and family at Bering Busty, Shri. Sagar Dhakal and family at Pendam. For assisting me with field work and logistics, I am very thankful to Shri. Yamuna P. Bastola, Shri. Sagar Chettri, Shri. Kuber Acharya, Shri. Ram C. Tiwari, Shri. Devi P. Khatiwada and Shri. Ram P. Acharya.

I thank Mr. Mike Meredith, Drs. Kailash S. Gaira, R. Sreekar, Joydeep Bhattacharjee, Naoki Katayama, for their valuable inputs and suggestion on statistical analysis. I extend my gratitude to Dr. Ajith Kumar; Late Dr. K.S. Krishnan; Dr. Rajah Jaypal; Department of Science and Technology; and Society for Conservation Biology for facilitating my attendance in training workshops and research visit at NCBS, IISc Bangalore, SACON Coimbatore, UWICER (Gelephu, Bhutan), and University of Nottingham Malaysia Campus. I thank Prof. Sandeep Tambe (IFS), Prof. Min Bahadur, Prof. S. Jayakumar and Dr. Ratul Baishya for their various help. I am grateful to Dr. Nittie Srivastava ma'am and Shri M.L. Srivastava sir for lending me i-7 laptop, which allowed me to complete data analysis in R software, when hard drive of my laptop had crashed.

I am grateful to Prof. Kiranmay Sarma, Dr. Laxmi Rawat, Dr. Pushpadhar Sarma, Dr. Renu Kathpalia, and Dr. Monish Tourangbam for inspiring and supporting me to pursue Ph.D. research. I also thank all my 'Gurus' from the primary school; Uparkhuti M.E. School; Golmagaon M.E. School; Jawahar Navodaya Vidyalaya, Nalbari; Kirori Mal College, Delhi University; Forest Research Institute Deemed University; The Mountain Institute-India; and SU. I express my gratitude to Shri. Jaganath Kharel, Shri. Bishnu Upadhyay, Shri. Sashi Sharma, Shri. Anil Barman, Shri. Pushpa Sharma, Shri Debo Rabha, Shri Dal B. Chettri, Smt. Yamuna Upadhyay, Smt. Rinzing Yakha Dewan, Shri. P.K. Basnet (Retired IFS, and former PCCF, FEWMD, GoS), Shri. Netra Upadhyay, Shri. Gopal C. Chetry, Shri. Padmalal Chetry, Shri. Tikaram Timsina, and their respective families, as well as all other well-wishers for helping me in my studies.

For the support, stimulating discussions and timely help, I would also like to thank my seniors: Shri. Suratna S.S.S. Sherpa (IFS), Shri. Jaswant N. (IFS), Shri. Kushal Tamang, Smt. Sangeeta Baral, Shri Bhuvan Dhamala; Drs. Kumar Basnet, Anand Sharma, Pankaj Sarma, Ikbal Ahmed, M. Siraj, Dipankar Nath, Gaurango Chakravarty, Binod Bhattarai, Saurabh Bajpai, Satyam Verma, Mangesh Pal, Sabita Devi, and friends: Rakesh Sharma, Sanjib Sharma, Rahul A. Pradhan, Sanjay Khati, Pramod Bhattarai, Rajesh Ngangbam, Mahesh Kalita, Kuljan CK, Bidyut B. Borah, Amar J. Nath, Nayan J. Gogoi, Anup, Lalit, Prakash Chettri, Mahindra Luitel, Kalsyang Nyuma, Sushant Mondal, Drs. Bijoy Chettri, Ekta Choudhury, Amit Kumar, Saurabh Anand, Nima Lepcha. My special thanks to Drs. Sailendra Dewan, Jins V.J., Ms Ananta Rai, Ms Prerna Dhakal, Suman Dahal, Deependra Chamlagai, Jiwan Gurung, Prem Chettri, and other scholars at the department for cooperation and supports during my Ph.D.

I am very grateful to my family for the unconditional love, blessings, constant support and encouragement in my quest for higher education. My late grandmother Ganga Maya Parsai nurtured and supported me during my childhood, and played a key role in initiating me to primary school, and would continue to bless me to study well and dream big till her last breathe in 2011. Her memories continue to inspire me till date. My parents, inspite of their limited education and facing hardships for better part of their life, nurtured me well, always put a lot of faith on me, and provided every supports for my education till Ph.D. My elder brother Shri. Binod K. Sharma, and younger sisters Smt. Lakhi Devi; Ms. Binita Devi have been a great siblings, to whom I could always fall back when faced with difficulties. I am also grateful to Ima, Thulema, Thulaba, Badema, Badaba, Mama, Maiju, Bhinaju, Bhauju, Jwai, Echema and Da Monish, Echeton and Yambung for always being very supportive in my endeavour. I thank my wife Dr. Samjetsabam Bharati Devi for being a great companion for over a decade now, and son Toyesh Sharma for inspiring me to become a better person and keeping me sane, especially while writing thesis. Finally, I thank the Almighty God for giving me a loving family, compassionate 'Gurus', wonderful friends and empathetic well-wishers, and guiding me to navigate the rollercoaster ride in life especially during my Ph.D. research.

***Place:** Gangtok, Sikkim*

***Date:** December 2021* *Kishor Sharma*

CONTENTS

LIST OF FIGURES

LIST OF TABLES

LIST OF PHOTO PLATES

LIST OF APPENDICES

ABBREVIATIONS

AES: agri-environmental schemes

AET: actual evapo-transpiration

AEZs: Agro-ecological zones

AlM: Altitudinal migrant

ANOVA: Analysis of variance

AMSL: Above mean sea level

BEF: Biodiversity-ecosystem function

CAP: Common Agricultural Policy

CCS: Conservation concern species

CITES: Convention on International Trade in Endangered Species of Wild Fauna and Flora

CWM_bodymass: Community-weighted mean of body mass for birds

CWM_Wingspan: Community-weighted mean of wingspan for butterflies

EBA: Endemic bird area

EE: Ecosystem engineering

ES: Ecosystem services

FAS: Farm-based agroforestry systems

FD: Functional diversity

FDis: Functional dispersion

FDiv: Functional divergence

FEve: Functional evenness

FRic: Functional richness

Forest: Natural forests

FG: Forest generalist

FS: Forest specialist

FSB: Forest specialist butterflies

FSCIBi: Forest specialist canopy insectivores

FSFNBi: Forest specialist frugivores and nectarivores

FSMSIBi: Forest specialist midstorey insectivores

FSUIBi: Forest specialist understorey insectivores

GIAHS: Globally important agriculture heritage system programme

GLMM: Generalized linear mixed model

GPS: Geographic positioning system

IBAs: Important bird areas

IFS: Indigenous farming systems

IndVal: Indicator value index

IPC: Insect/ invertebrate pest control

ISFR: India state of forest report

KBR: Kanchendzonga Biosphere Reserve

LCAS: Large cardamom-based agroforestry systems

LULCC: Land use and land cover change

MAP: Mean annual precipitation (rain: in mm)

MAT: Mean annual temperature (temp: in ° C)

MB: Monophagous butterflies

MOAS: Mandarin orange-based agroforestry systems

MOEF & CC: Ministry of Environment, Forests & Climate Change

MRM: Multiple Regressions on distance Matrices

ND: Nutrient deposition

NMDS: Non-metric multidimensional scaling

NPP: Net primary productivity

NTFP: Non-timber forest products

OA: Open-land species

P: Pollination

PAs: Protected areas

PB: Protected butterflies

PEBi: Protected and endemic birds

PM: Passage migrant

SD: Seed dispersal

SV: Summer visitor

TB: Total butterflies

TBi: Total birds

TD: Taxonomic diversity

TGA: Total geographic area

WD: Vertebrate pest control & scavenging

WPA 1972: Indian Wildlife Protection Act 1972

WV: Winter visitor

AICc: Second-order Akaike Information Criterion
AICw: Akaike weights
Δ **AICc**: Delta second-order Akaike Information Criterion
$\boldsymbol{\beta}_{\mathbf{bray}}$: Pair wise Bray-Curtis dissimilarity
$\boldsymbol{\beta}_{\mathbf{bray.bal}}$: Pair wise Bray-Curtis dissimilarity due to balanced variation in abundance
$\boldsymbol{\beta}_{\mathbf{bray.gra}}$: Pair wise Bray-Curtis dissimilarity components due to abundance gradients
$\boldsymbol{\beta}_{\mathbf{BRAY}}$: Multiple-site Bray-Curtis dissimilarity
$\boldsymbol{\beta}_{\mathbf{BRAY.BAL}}$: Multiple-site Bray-Curtis dissimilarity components due to balanced variation in abundance
$\boldsymbol{\beta}_{\mathbf{BRAY.GRA}}$: Multiple-site Bray-Curtis dissimilarity components due to abundance gradients
$\boldsymbol{\beta}_{\mathbf{sor}}$: Pair-wise Sorensen dissimilarity
$\boldsymbol{\beta}_{\mathbf{sim}}$: Pair-wise Sorensen dissimilarity components of turnover
$\boldsymbol{\beta}_{\mathbf{sne}}$: Pair-wise Sorensen dissimilarity components of nestedness-resultant
$\boldsymbol{\beta}_{\mathbf{SOR}}$: Multiple-site Sorensen dissimilarity
$\boldsymbol{\beta}_{\mathbf{SIM}}$: Multiple-site Sorensen dissimilarity components of turnover
$\boldsymbol{\beta}_{\mathbf{SNE}}$: Multiple-site Sorensen dissimilarity components of nestedness-resultant
R: Environmental variables matrix
L: Species abundances matrix
Q: Functional traits matrix
H′: Shannon-wiener diversity index (shannon)
elev: Elevation (in m)
gbh: Girth at breast height (in cm)
mm: Millimetre
cm: Centimetre
m: Metre
km: Kilometre
%: Percentage
°C: Degree Celcius
pcc: Percentage canopy cover (in %)
tba: Tree basal area (in m^{2} ha^{-1})
tden.: Tree density (in stems ha^{-1})
tsr: Tree species richness
Tukey’s HSD: Tukey’s Honest Significant Difference

Chapter 1

GENERAL INTRODUCTION

Streaked Spiderhunter *Arachnothera magna*

1.1. Introduction

The agricultural landscapes are increasingly dominating the tropical region and continue to encroach upon remaining forest lands mostly due to ever increasing global population and subsequent demands for food (Laurance *et al.*, 2014). When compared to protected areas (PAs), agricultural landscape constitutes about three times more land cover (Approximately ~13% vs. ~38%) globally (Watson *et al.*, 2014; World Bank, 2019). Habitat destruction due to conversion of natural forest into agricultural lands is the most important cause for biodiversity loss worldwide. During the period of 1980 to 2000, more than 55% of new agricultural fields were established in primary forests and nearly 28% in secondary forests (Gibbs *et al.*, 2010). The rapid and unsustainable intensification of agriculture has caused landscape simplification leading to strong decline of biodiversity in agroecosystems and the adjacent natural forest ecosystems (Matson *et al.*, 1997; Tscharntke *et al.*, 2005).

1.1.1. Landscape heterogeneity, biodiversity conservation and ecosystem services

Landscape simplification associated with land use change and agricultural intensification has been reported to filter species trait and drive biotic homogenization where the specialist species are gradually replaced by generalist species in the community composition leading to reduction in beta diversity e.g., in birds (Devictor *et al.*, 2008; Şekercioğlu, 2012; Gámez-Virués *et al.*, 2015; Almeida *et al.*, 2016) and butterflies (Börschig *et al.*, 2013; Gámez-Virués *et al.*, 2015; Uchida and Ushimaru, 2015). Apart from affecting the taxonomic diversity (TD) and community composition, landscape simplification also negatively affects the functional diversity (FD: the diversity of traits that determine roles or function of species in an ecosystem; Tilman, 2001) and functional composition (multivariate distribution of traits across co-occurring species; Gravel *et al.*, 2016) in birds (Fischer *et al.*, 2007; Barbaro and Van Halder, 2009; Luck *et al.*, 2013, 2015; Lindenmayer *et al.*, 2015; Bregman *et al.*, 2016; Bovo *et al.*, 2018; Maseko *et al.*, 2019; Matuoka *et al.*, 2020), butterflies (Loos *et al.*, 2014; Hanspach *et al.*, 2015; Perović *et al.*, 2015; Aguirre-Gutiérrez *et al.*, 2017; Goded *et al.*, 2019), and plants/ trees (Diaz *et al.*, 2007; Flynn *et al.*, 2011; Loos *et al.*, 2015). Land-use intensification has also been reported to reduce functional redundancy (the number of species contributing similarly to an ecosystem function) and response diversity (how functionally similar species respond differently to disturbance) in birds (Luck *et al.*, 2013), butterflies (Kühsel and Blüthgen 2015) and plants (Laliberté *et al.*, 2010), and consequently affects the ecosystem resilience. Effect of landscape heterogeneity on biodiversity is scale dependent, alpha diversity increases due to introduction of non-native species at local level, but beta diversity declines at regional level (Ekroos *et al.*, 2010; Hiley *et al.*, 2016). However, studies have also reported the increase in beta diversity as well, for example tree communities in the central Himalaya (Sharma and Vetaas, 2015). Landscape heterogeneity has positive effect on persistence of both birds and butterflies in agroecosystems (Kumar *et al.*, 2009; Fischer *et al.*, 2011; Perović *et al.*, 2015). Interaction between FD and landscape heterogeneity drives the potential for different ecosystem services (ES) by birds including pollination, seed dispersal, natural pest (both invertebrates and vertebrates) control, scavenging, nutrient deposition, and ecosystem engineering (Şekercioğlu, 2006; Barbaro *et al.*, 2014, 2017; Bregman *et al.*, 2016; Cuthbert *et al.*, 2016; Lindell *et al.*, 2016). FD also interacts with landscape heterogeneity to drive ES by butterflies, including pollination, pest control, and cultural services (Losey and Vaughan, 2006; Cardinale *et al.*, 2012; Diffendorfer *et al.*, 2014; Cussera *et al.*, 2016; Jain *et al.*, 2016; Kehimkar, 2016).

1.1.2. Role of vegetation and habitat features on retention of biodiversity

Vegetation plays crucial role in maintaining bird and butterfly communities as it provide habitat for foraging, roosting and breeding in both forests and agroecosystems. Presence of native vegetation and resulting structural complexity within the agricultural landscape is critical for retention of endemic, forest specialist and resident species of birds (Daily *et al.*, 2001; Şekercioğlu *et al.*, 2007; Ranganathan *et al.*, 2008; Fischer *et al.*, 2011; Buechley *et al.*, 2015) and butterflies (Horner-Devine *et al.*, 2003; Ohwaki *et al.*, 2007; Schulze *et al.*, 2010). Such retention and seasonal dynamics are closely linked with the plant phenological behaviour (leaf drop, leaf flushing, flowering and fruiting) and availability of larval host plants and nectar plants specifically for butterflies (Koh and Sodhi, 2004; Kitahara *et al.*, 2008; Jain *et al.*, 2016). Land use change often results in decrease in plant species richness and diversity (Foody and Cutler, 2003). However, Sharma and Vetaas (2015) found consistently richer tree species in farmlands in Central Himalaya compared to forests but with distinct community composition. Agroforestry system involving shade trees also significantly contributes to the integrity of riparian corridors for wildlife conservation along tropical and subtropical belt in the Himalaya (Zomer *et al.*, 2001).

1.1.3. Importance of agro-ecosystems in biodiversity conservation

Agriculture creates novel ecosystems through transformation of landscapes, wherein new configuration of biotic and abiotic factors of the environment are set (Hobbs *et al.*, 2006). To achieve conservation goals within human-modified landscapes, it is necessary to have detailed understanding of functioning of various agricultural ecosystems, land use types, and their appropriate management measures

(Vandermeer and Perfecto, 2007; Flohre *et al.*, 2011). Agroforestry systems (those that combine trees within the cultivation of crops and/or rearing of animals) have particularly high conservation potential, due to their structural complexity, high floristic diversity and close resemblance to natural forest ecosystems (Schroth *et al.*, 2004). Potentiality of agricultural systems for biodiversity conservation and associated ES by depends on its management practices. When compared to conventional agroecosystems, retention of biodiversity and ES is found significantly high in traditionally managed (Mcneely and Schroth, 2006; Tscharntke *et al.*, 2008; Buechley *et al.*, 2015) and organic agroecosystems (Tuck *et al.*, 2014; Katayama *et al.*, 2019). Effective management of such ecosystems along with biodiversity benefits agriculture in terms of productivity, overall sustainability and resilience to climate extremes (Şekercioğlu, 2006; Isbell *et al.*, 2015; Grace *et al.*, 2016). The traditionally managed and organic agroecosystems retains high TD and community composition (including forest specialist, endemic/range-restricted or threatened species) (Mas and Dietsch, 2003, 2004; Harvey and Villalobos, 2007; Bubova *et al.*, 2015; Katayama *et al.*, 2019). These wildlife-friendly agroecosystems also sustains high FD and functional composition, and associated ecosystem functions (Luck *et al.*, 2013; Loos *et al.*, 2014, 2015; Hanspach *et al.*, 2015; Lindenmayer *et al.*, 2015; Barbaro *et al.*, 2017; Goded *et al.*, 2019).

The debate on land-sparing vs. land-sharing approach is ongoing to address the global challenge to balance the biodiversity conservation on one hand and pressure from land use change and agricultural intensification to ensure food security on the other (Chazdon *et al.*, 2009; Chappell and LaValle, 2011). Land-sparing involves separation of agricultural land (usually at high intensity production, with high yielding varieties) and PAs (geographically delimitated and legally protected natural forests), to preserve biodiversity and nature, and the associated ES (Green *et al.*, 2005; Michael *et al.*, 2016; Balmford *et al.*, 2019). Land-sparing approach is most widely adopted worldwide (Gibson *et al.*, 2011; Phalan *et al.*, 2011, 2016) including Himalaya (Manish and Pandit, 2019; Ghosh-Harihar *et al.*, 2019). However, potentiality of PAs, and therefore effectiveness of land-sparing approach to halt continued biodiversity loss has been widely debated (Singh, 1999; Watson *et al.*, 2014; Coad *et al.*, 2019; Velazco *et al.*, 2019). The inadequacy of land-sparing can be due to sparing of incomplete area or lower habitat quality (Balmford *et al.*, 2019). Land-sharing, on the other hand, is integration of food production (usually at low intensity and yields) with biodiversity conservation on the same land (Green *et al.*, 2005; Balmford *et al.*, 2019). For example, agri-environmental schemes (AES) of European Union compensate the farmers for potential loss in income so that detrimental effects of intensive agriculture on biodiversity are mitigated (Michael *et al.*, 2016). Other well-known examples of land-sharing are found in agricultural landscapes dominated by agroecosystems under wildlife-friendly agricultural scheme, traditional agroforestry system or organic farming (Green *et al.*, 2005; Fischer *et al.*, 2008, 2011; Perfecto and Vandermeer, 2011; Tscharntke *et al.*, 2012; Katayama *et al.*, 2019). The studies from sacred groves (Bhagwat *et al.*, 2005a, b; Shrestha *et al.*, 2018) and off-reserve forests (Dahal *et al.*, 2014) have also highlighted the importance of land-sharing framework.

1.1.4. Rationale for the study

The Eastern Himalaya, spreads over a wide spectrum of ecological zones, have a diverse socio-economic potential and biodiversity value. Pastoralism, agro-pastoralism, mixed farming systems, shifting cultivation and commercial cash crops cultivation are the five major farming systems in the Eastern Himalaya (Sharma and

Kerkhoff, 2004). Sikkim, a significant component of the Eastern Himalaya in terms of landscapes and biodiversity, houses one of the world's unique indigenous farming systems (IFS) that have been recognised by the FAO, UNO as an associate site under its Globally Important Agriculture Heritage System (GIAHS) Programme(GIAHS, 2007). The IFS of Sikkim are both traditionally managed and organic (Sharma and Acharya, 2013; Bhutia, 2016; Sharma *et al.*, 2016a). The Sikkim Himalayan mountains are unique due to its very extensive elevational gradient (300-8586 m), high variability in rainfall (from 3800 mm to <500 mm), and temperature (ranging between 28°C to sub-zero), both of which declines along the gradient (Acharya *et al.*, 2011a; Acharya and Vijayan, 2015; ISFR, 2019). It also experiences variation in climate from hot tropical below 900 m to sub-arctic and arctic above 4500 m (Grimmett *et al.*, 2019). Eastern Himalaya is also located at the convergence of the Indian, Indo-Malayan and Indo-Chinese biogeographic regions (Mani, 1974). The resulting complex physiography, bioclimatic zonation has led to the formation of wide range of ecological condition, elevational zonation of life forms and high biodiversity and endemism in the region (Mani, 1972; Pandit, 2017). The Eastern Himalaya with rich biodiversity and endemism within the Himalaya biodiversity hotspot is highly threatened by anthropogenic activities including land use and land cover change (LULCC) and climate change (Laiolo, 2004; Pandit *et al.*, 2007; Chettri *et al.*, 2010b; Pandit, 2017; Banerjee *et al.*, 2019).

The past studies on taxonomic diversity and community composition of birds in human modified landscapes including agroecosystems of India has been undertaken mostly in the Western Ghats region (Daniels *et al.*, 1990; Kunte *et al.*, 1999; Anand *et al.*, 2008; Ranganathan *et al.*, 2008; Karanth *et al.*, 2016; Chang *et al.*, 2018). A few such studies are also available from the Himalaya and North East region of India

(Raman, 2001; Elsen *et al.*, 2016, 2018; Mandal and Raman, 2016; Yashmita-Ulman *et al.*, 2016; Srinivasan *et al.*, 2019). Similarly, researchers have conducted few ecological studies on the butterfly communities in agroecosystems, but mainly in Western Ghats (Kunte, 1997; Kunte *et al.*, 1999; Shahabuddin and Ali, 2001; Dolia *et al.*, 2008). However, except for a recent study on birds (Cottee-Jones *et al.*, 2015), FD and functional composition of birds and butterflies, in both forest and human-modified ecosystems including agroecosystems in India has not been addressed yet. Therefore, exploration of biodiversity in the IFS of Eastern Himalaya is necessary to understand their potentiality in biodiversity conservation, retention of associated ES and to get insights on ecological roles of biodiversity components. Such information would be very important management inputs and aid in long-term conservation of biodiversity in human-modified landscapes.

Though birds and butterflies are the most studied vertebrate and invertebrate groups respectively, studies on the effects of habitat change on the life history traits of birds and butterflies (Koh, 2007; Barbaro and Van Halder, 2009; Börschig *et al.*, 2013; Newbold *et al.*, 2013; Hanspach *et al.*, 2015; Slancarova *et al.*, 2016), are still scarce, especially from the Eastern Himalaya. Therefore, extensive research is necessary to determine the life-history traits that best predict species retention in agroecosystems of Eastern Himalaya.

1.1.5. Birds and butterflies as indicator taxa

To objectively evaluate the suitability of a given taxon as an indicator, seven tests of criteria has been proposed by Pearson (1994): "(i) well known and stable taxonomy; (ii) biology and natural history well understood; (iii) readily surveyed and manipulated; (iv) higher taxa broadly distributed geographically and over a breadth of habitat types; (v) lower taxa specialized and sensitive to habitat changes; (vi) patterns of biodiversity reflected in other related and unrelated taxa; and (vii) potential economic importance". However, the author further called for prioritizing the criteria of potential indicator taxa based on choice among the two categories (monitoring or inventory) of biodiversity studies. The monitoring studies places an emphasis on sensitivity to habitat changes, but inventory studies focuses more on systematics.

Birds are easy to survey, diverse, best studied vertebrate taxa in terms of taxonomy, ecology and life history attributes, and its diet guilds responds differently to various threats (Şekercioğlu, 2012; Whelan *et al.*, 2015; Sreekar *et al.*, 2015) at different spatial scales (Zingg *et al.*, 2018). They also provide multiple ES such as seed dispersal, pollination, pest control, scavenging, ecosystem engineering and nutrient cycling (Şekercioğlu, 2006; Whelan *et al.*, 2015; Şekercioğlu *et al.*, 2016).

Similarly, butterflies are the most vulnerable invertebrate taxa with high sensitivity to climate and habitat changes (Thomas, 2005; Nelson, 2007), with varied response at local and landscape scales (Rundlof *et al.*, 2008; Zingg *et al.*, 2018) due to their short life-cycle, feeding and habitat specificity during their different life-stages, and high diversity and endemism (Dennis, 2010; ILTEO, 2015). They are also easy to survey with most well-documented taxonomy, ecology and life-history traits within the invertebrates (Thomas, 2005) and offer a wide range of ES including pollination, pest control and cultural services (Losey and Vaughan, 2006; Cardinale *et al.*, 2012; Diffendorfer *et al.*, 2014; Cussera *et al.*, 2016).

Due to these features, birds and butterflies are well-known biodiversity indicators of natural and human-modified ecosystems, accepted in India (ILTEO, 2015) and at global level (e.g., EU Commission) approving the status as a surrogate of biodiversity and environmental health (Kremen, 1992; Pollard and Yates, 1993; Schulze *et al.*, 2004a; Brereton *et al.*, 2011; Herrando *et al.*, 2016).

1.1.6. Ecological services of bird communities in agricultural landscapes

Avian mediated ES has been widely recognised in both natural ecosystems as well as agroecosystems (Şekercioğlu, 2006, 2012; Luck *et al.*, 2012; Whelan *et al.*, 2015; Şekercioğlu *et al.*, 2016). Birds provide various ES such as pollination (nectarivorous birds), seed dispersal (frugivorous and granivorous birds), invertebrate/insect pest control (insectivorous birds), vertebrate pest control and scavenging (vertebrate-feeding, and scavenging- omnivorous birds), nutrient deposition (piscivorous, aquatic, communal roosting birds), and ecosystem engineering (primary burrow and cavity-nesting birds), with the former four services under regulating and latter two under supporting types (Şekercioğlu *et al.*, 2004, 2016; Şekercioğlu, 2006).

1.1.6.1. Pollination

Pollination by birds (ornithophily) has been considerably studied in both natural and human-modified ecosystems (Subramanya and Radhamani, 1993; Corlett, 2004; Şekercioğlu, 2006; Şekercioğlu *et al.*, 2016). It has been estimated that more than 900 species of birds pollinate around 500 plant genera globally (Şekercioğlu, 2006). The decline in insect pollinators, has led to recognition of vertebrate pollinators especially birds in the recent decades. Birds such as Sunbirds, Spiderhunters, Flowerpeckers and White-eye are important pollinator birds, crucial for pollinating large number of plant species in both forests and agroecosystems in the region. The ornithophily is vulnerable to land-use change and agricultural intensification (Şekercioğlu, 2012; Cussera *et al.*, 2016; Maseko *et al.*, 2019; Shahabuddin *et al.*, 2021).

1.1.6.2. Seed dispersal

The seed dispersal by bird is also an important ES in both natural forests and agroecosystems (Davidar, 1987; Corlett, 1998; Levey *et al.*, 2005; Lozada *et al.*, 2007; Sethi and Howe, 2009; Wenny *et al.*, 2016). In tropical forests, land use and climate change has threatened population and seed recruitment by different species of hornbills (most of which are globally threatened) and other avian seed dispersers especially forest specialist species (Sethi and Howe, 2009; Naniwadekar *et al.*, 2015; Sreekar *et al.*, 2015; Bregman *et al.*, 2016; Peters *et al.*, 2019; Shahabuddin *et al.*, 2021).

1.1.6.3. Invertebrate pest control

Increasing number of studies from different parts of the world have reported birds as the most efficient predators of arthropods, leading to invertebrate pest control in both natural forests (Letourneau *et al.*, 2009; Singh, 2010; Böhm *et al.*, 2011) and agro-ecosystems (Philpott *et al.*, 2009; Sinu, 2011; Karp *et al.*, 2013; Maas *et al.*, 2013, 2015; Barbaro *et al.*, 2017). This avian-mediated ES is in turn associated with significant reduction in leaf damage and plant mortality, thereby leading to around 60% increase in crop yield and fruit production (Karp *et al.*, 2013; Maas *et al.*, 2013; Whelan *et al.*, 2015). Insect pest control is widely recognized in agroecosystems because more than half of the birds are insectivores and one-third are occasional invertebrate feeders (Whelan *et al.*, 2015). In Sal forest (the major forest type in the tropical belt of Sikkim and elsewhere in the Himalaya), Woodpeckers help control infestation of wood-borer beetles, and promote natural regeneration of forest stands (Singh, 2010). Studies have further reported that pest control by natural enemies depends on different factors, e.g., remnant habitat size and proximity to habitat patch

(Karp *et al.*, 2013; Jordani *et al.*, 2015), tree age and shade tree cover, forest composition and cover, identity and species richness of avian predators (Böhm *et al.*, 2011; Maas *et al.*, 2013, 2015), landscape heterogeneity and structure (Barbaro *et al.*, 2017; Boesing *et al.*, 2017). Among the different types of agroecosystems, organic one shows more pronounced avian-mediated pest control (Crowder *et al.*, 2010; Mangan *et al.*, 2017), and subsequent increase in crop yield (Maas *et al.*, 2013) by promoting evenness in natural enemies (Crowder *et al.*, 2010) and enhanced tree cover (Luck *et al.*, 2012). The insect pest control provider birds (specially forest specialist insectivores) are highly threatened by land use and climate change (Sreekar *et al.*, 2015; Bregman *et al.*, 2016; Shahabuddin *et al.*, 2021).

1.1.6.4. Vertebrate pest control and scavenging

The efficiency of non-insect/vertebrate pest control by birds of prey in agricultural landscapes is being recognised from different parts of the world (Şekercioğlu, 2006; Lindell *et al.*, 2018). Integration of birds of prey into the agroecosystems allow for effective, inexpensive and environment friendly biological control of vertebrate pests resulting into increase in farmers' revenue and safeguard public health through prevention of pest's population outbreak (Lee, 1997; Pande and Dahanukar, 2011; Kan *et al.*, 2014; Kross *et al.*, 2012, 2016). Among the species associated with pest control, owls are agile nocturnal hunter and effectively control rodents outbreaks in agroecosystems, e.g., Barn Owls in oil palm-and cocoa-dominated agricultural landscapes in Malaysia (Lee, 1997), farmlands in Israel (Meyrom *et al.*, 2009), and agricultural landscape in USA (Kross *et al.*, 2016), Indian Eagle Owl (Pande and Dahanukar, 2011), and Spotted Owlet (Vanitha *et al.*, 2014) in farmlands of India. The diurnal counterparts, the raptors, plays complementary role in biological vertebrate pest control in various crop fields (Kross *et al.*, 2012).

The scavenging of carcases by birds is well documented from forests and human-modified ecosystems both globally (Peterson *al.*, 2001; Şekercioğlu, 2006; DeVault *et al.*, 2016) and within India (Markandya *et al.*, 2008). The scavenging and omnivorous birds contribute in disposal of carcasses and wastes, energy cycling and disease control in agricultural and forest ecosystems (Peterson *et al.*, 2001; DeVault *et al.*, 2016). However, these scavenging birds (such as *Gyps* vultures in Asia) are greatly threatened mainly due to poisoning from veterinary drugs namely Diclofenac and Nimesulide (Cuthbert *et al.*, 2016). Although Diclofenac has been banned in India and other countries, *Gyps* vultures and other scavengers are still declining, resulting in diminishing critical scavenging services.

The birds providing vertebrate pest control (e.g.: Raptors and Owls) and scavenging (e.g.: Vultures, Crows) services are revered and protected in the Indian sub-continent due their socio-cultural and religious importance. But recently, they have been greatly threatened due to land-use change and agricultural intensification (through biomagnification of chemical pesticides) and diminishing socio-cultural values (through intentional poisoning, use of banned veterinary drugs).

1.1.6.5. Nutrient cycling

The bird communities play significant role in nutrient cycling (Şekercioğlu, 2006; Fujita and Koike, 2009; Fujita and Kameda, 2016). Prior to the discovery of artificial nitrification, phosphate-rich guano deposits of seabirds were much sought and traded commodity in agricultural sector (Schnug *et al.*, 2018). Seabirds contribute nutrient cycling of nitrogen and phosphorus in the islands and coastal regions, whereas aquatic birds play similar role in riparian and forest ecosystems (Kitchell *et al.*, 1999; Ligeza and Smal, 2003; Fujita and Koike, 2009; Fujita and Kameda, 2016; Otero *et al.*, 2018). The avian-mediated aquatic–terrestrial reciprocal energy flows in heterogeneous landscapes affects food-web dynamics (Nakano and Murakami, 2001; Rundio and Lindley, 2012), and structure and composition of plant communities (Ellis, 2005). This in turn depends on hydro-period and species richness of aquatic birds (Schriever *et al.*, 2014). The LULCC including those resulting from hydro-power dams and pharmaceutical companies in the bio-diverse tropical and sub-tropical riparian corridors has greatly threatened these nutrient depositor birds, like most other fauna and flora taxa in the Himalaya (Zomer *et al.*, 2001; Pandit, 2017).

1.1.6.6. Ecosystem engineering

Similar to many other vertebrate taxa, some groups of burrow- and cavity-excavating birds (e.g., Woodpeckers, Trogons, Beavers, Bee-eaters, Kingfishers, Rollers, Owls, etc.) are also excellent ecosystem engineers. Once these nests abandoned after the breeding season, they are subsequently occupied by other species of birds and mammals, many of which contribute other important ES in the landscapes (Casas-Crivillé and Valera, 2005; Şekercioğlu, 2006; Cockle *et al.*, 2011; Sodhi *et al.*, 2011; Nummi and Holopainen, 2014; Şekercioğlu *et al.*, 2016). Globally, about 1000 species of birds are ecosystem engineers (Şekercioğlu, 2006), which have been found to be highly sensitive to land-use change and agricultural intensification (Şekercioğlu, 2012; Ibarra *et al.*, 2017; Shahabuddin *et al.*, 2021). The ecosystem engineer birds like most other avian-mediated ES providers, is poorly studied in the Himalaya (but see Singh, 2010; Shahabuddin *et al.*, 2021).

1.1.7. Ecosystem disservices by birds

In addition to the wide range of avian-mediated ES, some birds, especially non-native passerines, are known for crop damage and depredation. They damage varieties of crop orchards such as vineyard, apple orchards, almond orchards and sweet cherries (Kross *et al.*, 2012; Luck *et al.*, 2015; Mangan *et al.*, 2017; Lindell *et al*, 2016), and many other food crops and fruits (Dhindsa and Saini, 1994; Kale *et al.*, 2014). However, many studies that assessed cost-benefit trade-off of bird activity in agroecosystems have shown that the benefit of insect pest control outweighs the crop damage, thereby leading to 20-70% increase in crop yields (Kellermann *et al.*, 2008; Maas *et al.*, 2013; Karp *et al.*, 2013; Whelan *et al.*, 2015; Peisley *et al.*, 2015).

1.1.8. Ecological services of butterflies in agricultural landscapes

The butterflies play an important role in the ecosystems. They provide different ES such as pollination, pest control, and cultural services.

1.1.8.1. Pollination

Globally, almost 87.5% species of the flowering plants and more than three-fourth of the food crops depends on animal pollination or in essence the Zoophily (Klein *et al.*, 2007; Ollerton *et al.*, 2011). The land use change through agricultural intensification have severely threatened the insect pollinators including honey bees and butterflies worldwide (Kremen *et al.*, 2007; Potts *et al.*, 2010; Jain *et al.*, 2016; Dainese *et al.*, 2017). Recent studies have highlighted that wild pollinators (other than bees) including butterflies can enhance and stabilize pollination services threatened by land use change and landscape simplification (Kremen *et al.*, 2007; Dainese *et al.*, 2017). They can also enhance fruit set of crops (Garibaldi *et al.*, 2013), yield and quality of oilseeds (Bommarco *et al.*, 2012; Zou *et al.*, 2017). Pollination by butterflies (Psychophily) is important ES in both natural forests and human-modified ecosystems (Balasubramanian, 1990; Corlett, 2004; Cussera *et al.*, 2016; Jain *et al.*, 2016), because butterflies have strong flying capacity and can move pollens to long distances (Andersson *et al.*, 2002). Butterflies usually depends on olfactory/chemical signal to pollinate plants having white/dull coloured, strongly scented flowers (Balasubramanian, 1990; Andersson *et al.*, 2002), whereas they depend on visual signal to pollinate plants with brightly coloured, faintly scented flowers (Borges *et al.*, 2003). The adoption of traditional and organic agroecosystems can protect the wild pollinators and consequently stabilize pollination services (Ohwaki *et al.*, 2007; Rundlof *et al.*, 2008; Hanspach *et al.*, 2015; Goded *et al.*, 2019).

1.1.8.2. Pest control

Except for few species, the Lepidoptera also contribute to herbivore pest control in the natural forests and agroecosystems. Some butterfly species of Lycaenidae family are insectivorous at larval stage, and directly helps in herbivore pest control by feeding on aphids (e.g., Brownies *Miletus* spp.), scale insects (e.g., Apefly *Spalgis epius*, Mottles *Logania* spp., and Forest Pierrot *Taraka hamada*), and ant larvae (e.g., Moth butterfly *Liphyra brassolis*) (Kehimkar, 2016). The butterflies also indirectly contribute to pest control by provisioning of caterpillar biomass for the natural predators such as insectivorous birds, spiders, etc. (Hammond and Miller, 1998; Losey and Vaughan, 2006). The butterfly-mediated pest control service can be sustained and/or enhanced by adopting wildlife-friendly agricultural practices.

1.1.8.3. Cultural services

In recent decades, cultural services provided by butterflies have been widely recognized (Diffendorfer *et al.*, 2014; Kehimkar, 2016). U.S.A. and neighbouring countries (having habitat or annual migration routes of Monarch butterfly) have started providing incentives to the local communities for the cultural values of this iconic and threatened butterfly species to aid its conservation (Diffendorfer *et al.*, 2014). A recent study reported that Monarchs are valued at one-time payment of \$4.78–\$6.64 billion overall by households in U.S.A. (Diffendorfer *et al.*, 2014). Similar incentive-based conservation measures for butterflies are being explored and adopted in different parts of India, e.g., in the form of ecotourism and butterfly parks in Eastern Himalaya and Western Ghats (Kumari *et al.*, 2010; Kehimkar, 2016; Singh, 2017b), and annual migration routes of milkweed butterflies in the Western Ghats (Kehimkar, 2016). Such initiatives will greatly encourage the local communities in different agricultural landscapes of India to adopt wildlife-friendly farming practices, and take proactive role in conservation of butterflies, other biodiversity elements and their habitats.

1.1.9. Ecosystem disservices by butterflies

Studies have also reported ecological disservices by few species of butterflies in the form of crop damage in agroecosystems (Feber *et al.*, 1997; Jainulabdeen and Prasad, 2004; Ali and Rizvi, 2007). Butterflies affect six different species of *Brassica* as well as *Eruca sativa* (Jainulabdeen and Prasad, 2004). The population dynamics of pest butterflies, and consequently crop damage in agroecosystems depends on relative humidity (Jainulabdeen and Prasad, 2004), crop plants' intraspecific variation in leaf nitrogen and water content (Tabashnik, 1982) as well as farming management (Feber, 1997). A comparative study of pest and non-pest butterflies in UK, reported significantly more non-pest butterflies in organic than conventional agroecosystems, and uncropped boundary habitat than crop edge habitat in both type of agroecosystems (Feber *et al.*, 1997).

1.2. OBJECTIVES

The broad aim of this study is to understand the biodiversity conservation potential of the indigenous farming systems of Sikkim in the Eastern Himalaya.

The specific objectives are:

1. To assess the community structure of birds and butterflies in the indigenous farming systems.
2. To understand the functional diversity of birds and butterflies in the indigenous farming systems.
3. To identify indigenous farming systems of high biodiversity conservation value.

1.3. THESIS OUTLINE

The thesis has been organized into a total of eight chapters. The first chapter, titled "General Introduction" gives overview of the present study focussing on the land-use change and simplification, which has resulted into continued biodiversity loss and biotic homogenization, and how organic and/or traditionally managed agro-ecosystems in the heterogeneous landscapes can complement the efforts of PAs in biodiversity conservation. It then explains the research problem, objectives of the research and ends with thesis outline.

The chapter 2, titled "Review of Literature" focuses on the previous studies conducted globally, within India, and finally from the Eastern Himalaya, to understand the potentiality of agro-ecosystems, especially organic and traditionally managed agricultural landscapes, in biodiversity conservation with reference to birds and butterflies, and identifies the research gap in the subject.

The third chapter, titled “Overview of the Study Area”, describes the study region including the climate, vegetation types, agro-ecosystems types, biodiversity, biogeography, and thereafter study ecosystems, and study sites selection.

The next four technical chapters address the three objectives of this thesis. The fourth Chapter describes the patterns and plausible determinants of bird alpha and beta diversity along agroecosystem-forest gradient of Sikkim. Chapter five describes the patterns and plausible determinants of butterfly alpha and beta diversity along agroecosystem-forest gradient of Sikkim. Similarly, chapter six explores the functional diversity and functional composition of birds and butterflies along the agroecosystem-forest gradient of Sikkim. The chapter seven highlights the importance of the agricultural-dominated landscapes of Sikkim for the protection of conservation concern species of birds and butterflies. It also identifies the indicator and ecosystem exclusive species of birds and butterflies for long-term ecological monitoring in the Eastern Himalaya.

The Ph.D. thesis concludes with the final chapter, titled “Synthesis and Recommendations”, which synthesizes the key research findings of the thesis, and give recommendations for conservation of birds and butterflies and the associated ecosystems services in the indigenous and organic agroecosystems of Sikkim, Eastern Himalaya.

Chapter 2

REVIEW OF LITERATURE

Indian Tortoiseshell *Aglais caschmirensis aesis* Fruhstorfer, 1912

2.1. Biodiversity conservation potential of agroecosystems

Land-use change and agricultural intensification induced landscape simplification has resulted in decline in biodiversity and associated ES throughout the world, more adversely in the tropics (Devictor *et al.*, 2008; Gibbs *et al.*, 2010; Börschig *et al.*, 2013; Gámez-Virués *et al.*, 2015; Almeida *et al.*, 2016). Such habitat simplification is mainly attributed to the anthropogenic pressure in order to meet the increased food demand for the rapidly increasing global population which is expected to reach nine billion by 2100 A.D. (Gibbs *et al.*, 2010; Laurance *et al.*, 2014). In order to address the dual global challenges of biodiversity conservation and ensuring food security (Tscharntke *et al.*, 2012), concept of land-sharing or land-sparing is gaining importance (Godfray *et al.*, 2010; Phalan *et al.*, 2011, 2016). The land-sparing is the most preferred conservation strategy globally (Gibson *et al.*, 2011; Phalan *et al.*, 2011). However, the effectiveness of this strategy to halt continued loss of biodiversity and ES from LULCC and climate change has been largely debated (Watson *et al.*, 2014; Coad *et al.*, 2019; Velazco *et al.*, 2019). On the other hand, several studies from agricultural landscapes has led to the recognition of land-sharing approach to play a complementary role in conservation measures (Bhagwat *et al.*, 2008; Chazdon *et al.*, 2009; Chappell and LaValle, 2011; Tscharntke *et al.*, 2012; Michael *et al.*, 2016). Still a few studies have urged for adopting both land-sparing and land-sharing strategies in conservation framework, instead of either one of them (Grass *et al.*, 2019). In the Himalaya, most studies (Manish and Pandit, 2019; Ghosh-Harihar *et al.*, 2019) have called for land-sparing strategy (more focus on establishment of PAs), while others have questioned its adequacy (Singh, 1999) or highlighted the role of agroecosystems in biodiversity conservation, therefore of land-sharing approach (Elsen *et al.*, 2018).

2.2. International status

2.2.1. Birds

Numerous studies indicate that human modified ecosystems such as shaded coffee, shaded cacao, jungle rubber, shaded tea, and other multi-strata agroforestry systems can conserve high bird diversity including forest specialist, endemic and threatened species (Perfecto *et al.*, 2003; Round *et al.*, 2006; Beukema *et al.*, 2007; Harvey and Villalobos, 2007; Lin *et al.*, 2012; Katayama, 2016; Prabowo *et al.*, 2016), in some cases even greater than the adjoining forests (Buechley *et al.*, 2015). Shade coffee agroforestry systems has been proposed as functional surrogate of the tropical forests for biodiversity because their structural complexity and taxonomical diversity provide suitable food sources and niche requirements (Philpott *et al.*, 2008) with high bird diversity (Perfecto *et al.*, 2003; Buechley *et al.*, 2015). Shift in the management practice from traditional shade coffee to intense and alternate management practice has led to drastic loss of associated biodiversity in the neotropics (Perfecto *et al.*, 2003; Philpott *et al.*, 2008). Mas and Dietsch (2004) studied the bird diversity indifferent shade coffee management systems (intensive commercial to traditional, rustic systems) in Mexico and found decline in bird species richness with increasing management intensity. Higher bird diversity (but with modified community assemblage) has been reported in shade Cacao agroforestry systems when it involved diverse shade tree species (especially forest trees) in Costa Rica (Reitsma *et al.*, 2001), Panama (Van Bael *et al.*, 2007) and Indonesia (Clough *et al.*, 2009). The diversity increased further, when coffee plantations were traditionally managed (Harvey and Villalobos, 2007) and when landscape of farms was surrounded by forests instead of agricultural lands (Faria *et al.*, 2006). Beukema *et al.*(2007) reported equivalent bird species richness between jungle rubber agroforestry system and primary forest (though with lower forest species) in Sumatra, Indonesia. Lin *et al.* (2012) found significantly reduced bird species richness from forest to shade tea plantations to open terraced tea plantations in China. The richness and diversity among functional traits differed and insectivorous birds significantly increased from forest and shade to open tea land. In Thailand, the mixed fruit orchard harboured about 75% more bird species compared to forest but the community was dominated by smaller frugivores, nectarivores and widespread generalist species (Round *et al.*, 2006). Similarly, Milder *et al.*(2010) found high diversity of birds, butterflies and trees in agricultural mosaics involving various cultivated systems in Central America. Bird community in wheat fields located along a gradient of landscape structural complexity and farming practice in Germany showed mixed effects of landscape structure and farming practice on diversity, abundance and species richness (Fischer *et al.*, 2011). Farmland and forest bird species in agricultural landscapes were enhanced by landscape complexity owing to the availability of nesting and sheltering sites in non-crop habitats. Similarly, organic farming enhanced the species richness of all groups of birds during the breeding season. Study on the biodiversity conservation values of tropical land-use systems in Cameroon by using geographic range size of birds, butterflies, trees and understorey plant species along a gradient of habitat modification displayed declining species richness with increasing habitat modification between taxon-specific groups of similar geographic range categories (Waltert *et al.*, 2011). Winqvist *et al.* (2011) studied the farmland biodiversity in Europe and reported a great reduction in plant species richness (33 % vs. 16%) and cover (14% vs. 5.5%), and bird species richness (45.5% vs. 34%) and abundance (39% vs. 32%) in conventional fields compared to organic fields due to landscape simplification (20%-100%). Organic farming enhances biodiversity by 30% than conventional farming systems (Tuck *et al.*, 2014). Tanalgo *et al.* (2015) reported decline in diversity of birds along the disturbance gradient of different land-use types (agroforests, ricefields and roads and heavily disturbed areas) and, high diversity of frugivores and insectivores in agroforests and ricefields respectively in Philippines. Katayama (2016) found higher diversity and richness of birds (especially insectivores) in organic apple orchard compared to conventional orchards in Japan.

2.2.2. Butterflies

Many studies have recorded high diversity of butterflies in agro-ecosystems, e.g. in Vietnam (Lien and Yuan, 2003), Costa Rica (Horner-Devine *et al.*, 2003) and Japan (Kitahara, 2008) but there are reports of low butterfly diversity in some regions (Schulze *et al.*, 2004 a, b; Vu, 2009). Butterfly community is negatively affected by habitat loss and modification (Perfecto *et al.*, 2003; Bobo *et al.*, 2006). In Central Sulawesi, Schulze *et al.* (2004 a, b) found a steady decrease of butterfly species diversity from natural forest, to old secondary forest, secondary forests, agroforestry systems and maize field sites but no significant difference between natural and old secondary forests sites. However, in Cameroon, Bobo *et al.*(2006) reported significant decline of butterfly richness and abundance from secondary forests and agroforestry sites towards near primary forests and annual crop sites, and high species turn over along the gradient of land conversion in butterfly community but with loss of range-restricted and forest species. In Southern Brazil, Francesconi *et al.* (2013) compared species richness of fruit-feeding butterflies in six land-use practices in two agricultural landscapes. The study found distinct species assemblage in agricultural practices and forest, but significant difference only between three agricultural land-use practices (Pastures, Cassava and Sugarcane) and the forest habitats (edge and interior). Shaded coffee practices maintaining long-term mixed tree and crop stands had better potential of conserving forest butterfly species compared to monoculture practices. In Mexico, Mas and Dietsch (2003) studied the fruit-feeding butterfly species richness and vegetation structure indifferent shade coffee management systems (intensive commercial to traditional, rustic systems) and observed decline in butterfly species richness with increasing management intensity. Schulze *et al.* (2010) studied the potential of land-use systems for maintaining tropical forest butterfly diversity and highlighted the importance of human-modified habitats for their conservation across all major tropical regions. Bubova *et al.* (2015) reviewed the impacts of land management on European butterflies of conservation concern and provided policy recommendations for management of butterfly habitats. In a large-scale bird and butterfly monitoring study in the north-east Iberian Peninsula, Herrando *et al.* (2016) found significant correlations of species' habitat preferences to population trends: for both birds and butterflies, there was drastic decline of open-habitat species but moderate increase of forest species.

2.3. National status

2.3.1. Birds

India represents one of the largest agroecosystems in the world and is the first country to have separate policy on agroforestry (National Agroforestry Policy, 2014). In India, bird diversity of human modified landscapes including agroecosystems has been fairly studied but mostly in the Western Ghats region (Daniels *et al.*, 1992; Anand *et al.*, 2008, 2010; Ranganathan *et al.*, 2008; Sreekar *et al.*, 2013, 2015; Karanth *et al.*, 2016). Such studies from the Himalaya and North East region of India that commenced about a decade later are comparatively less (Raman, 2001; Elsen *et al.*, 2016, 2018; Mandal and Raman, 2016; Yashmita-Ulman *et al.*, 2016).

Anand *et al.* (2010) reviewed 35 literatures on biodiversity (nine dealt on birds and five on butterflies) in human-modified landscapes in the Western Ghats and highlighted that conserving remnant forests not only secure their ability to harbour biodiversity, but also enhance the effectiveness of biodiversity-friendly farming practices. The role of ancient tropical countryside and agroforestry systems in sustaining and acting as refugia for biodiversity (especially birds) has been highlighted by some studies (Bhagwat *et al.* 2008; Ranganathan *et al.* 2008). Along the land-use gradient in the Western Ghats and Sri Lanka, Goodale *et al.* (2014) reported decline in forest specialist species (but increase in open-landscape species) in case of both total bird community and mixed-species bird flocks. From the same biodiversity hotspot, Sreekar *et al.* (2015) also reported decline in bird diversity (total as well as forest specialist frugivorous and insectivorous birds) and mass-abundance relationships of understorey insectivorous birds. Daniels *et al.* (1992) found positive correlation of bird species diversity with woody plant species diversity and vertical stratification in the tea plantations and other human-modified ecosystems in Western Ghats. Shaded coffee plantations sustain high bird diversity and may play the role of a buffer to the frugivorous and insectivorous species (Bhagwat *et al.*, 2005a, b). In the tea-dominated landscapes of Western Ghats, the natural windbreaks (of native trees) enhanced bird diversity by sustaining significant proportion of resident forest-dependent species (Sreekar *et al.*, 2013). Similarly, studies have highlighted the importance of shade coffee plantations (especially those located in proximity to forests) in retention of significant amount of biodiversity, especially birds (Shahabuddin, 1997; Kunte *et al.*, 1999; Bhagwat *et al.*, 2005a,b, 2008; Raman, 2006; Anand *et al.*, 2008). Anand *et al.* (2008) reported the bird conservation values of coffee plantation (which was most importantly determined by proximity to forests) and highlighted the need for certification of the same. Karanth *et al.* (2016) observed higher estimated richness of birds per agroforests and densities (in three feeding guilds) in coffee agroforestry systems compared to rubber and areca agroforests, which was determined by the tree cover, tree density and rainfall.

A few studies have also been conducted in the agroecosystems of Himalaya and North East India. Elsen *et al.* (2016, 2018) highlighted the importance of agricultural landscapes for bird conservation during both winter and breeding seasons in the Western Himalaya. Studies on effect of shifting cultivation on birds in Mizoram (Raman *et al.*, 1998; Raman, 2001) found increase in species richness (33 to 69 species) and Shannon-Wiener diversity (2.2 to 3.13) of birds along the vegetation succession gradient. Cottee-Jones *et al.* (2015) reported role of isolated *Ficus* trees in conserving frugivorous bird composition and FD in a human-modified landscape in Assam. In a study conducted in three types of agroforestry systems of Assam, Yashmita-Ulman *et al.* (2016) found high bird diversity (including five species of conservations concern) in different types of agroecosystems. Mandal and Raman (2016) found decline in tropical forest birds due to conversion of shifting agriculture to monocultures of oil palm and teak plantations in Mizoram, North East India.

2.3.2. Butterflies

Ecological studies on the butterfly communities in agroecosystems of India are scanty (Kunte *et al.*, 1999; Shahabuddin and Ali, 2001; Dolia *et al.*, 2008). Kunte *et al.* (1999) studied the butterfly assemblage across eight habitat types (evergreen forests, semi-evergreen forests, deciduous forests, shrubs/savannah, grasslands, monoculture plantation, home gardens and paddy fields) and found decline in species richness and beta diversity along land-use intensity gradient. Shahabuddin and Ali

(2001) studied the impacts of land use change on forest butterflies in the Western Ghats by comparing butterfly communities of forest habitats with lime plantations and found comparable butterfly abundance and species richness (but significantly distinct community composition) in these two habitats. Dolia *et al.* (2008) studied the adult butterfly communities in coffee plantations in Western Ghats and found significant negative effect of distance to protected area and percentage canopy cover on abundance and richness of butterflies. The study proposed that coffee plantations can act as a buffer for butterfly fauna within a certain radius of a PA and can complement PA in conservation efforts.

2.4. Status in Sikkim Himalaya

2.4.1. Birds

More than 11000 bird species are reported worldwide (Birdlife International, 2020a) out of which 1335 species occur in India (Praveen *et al.*, 2016, 2020a). Since the pioneering work of Bulger (1869) in the 19th century, the avifauna of Sikkim Himalaya has been well explored and described by numerous world renowned ornithologists, naturalists and ecologists but most of these studies were mainly explorative in nature to describe new species or their life history traits (reviewed in Ali, 1962; Acharya and Vijayan, 2011a). About a century ago, Herbert Stevens in a series of publications during 1923-1925 provided the first comprehensive account on avifauna of Sikkim Himalaya enumerating 549 species and sub-species (Steven, 1923, 1925). The author also made the pioneering observation on the effects of deforestation on avifauna in the eastern Himalaya, and that retaining, even a fragment of native forests in the human-modified landscapes/ cultivation are favourable to avifauna when compared to disastrous loss in area with unchecked total deforestation. Earnst Schafer

(1938-39), during the third Tibet expedition, extensively collected birds skins (>2700) and eggs (2000) along with other faunal and floral materials from Sikkim, the result of which was recently published (Abs *et al.*, 2010a, b). The analysis of these bird' skins revealed that during the one year study period, Schafer and team collected more than 2700 skin which represents 331 avifauna species across the elevation gradient of Sikkim (Abs *et al.*, 2010a, b; Frahnert *et al.*, 2012). Salim Ali provided the most extensive work on avifauna of Sikkim till date, reporting 527 bird species including their distribution and natural history (Ali, 1962). Acharya and Vijayan (2011a) updated the list of avifauna found in Sikkim and prepared a comprehensive checklist with their altitudinal records, which comprises 574 species of birds belonging to 253 genera and 55 families. The increased focus on ornithological research in the recent times has resulted in new sightings of species by various researchers (Acharya *et al.*, 2010; Acharya and Vijayan, 2011a; Rahut *et al.*, 2012; Alström *et al.*, 2016; Choudhury, 2016; Sharma and Bhatt, 2016; Ash *et al.*, 2017; Singh, 2017a; Chettri *et al.*, 2019, Chettri and Ethenpa, 2020; Lepcha *et al.*, 2020), totalling ~580 bird species in Sikkim.

However, detailed ecological studies on the bird communities of Sikkim which picked up only in the last two decades (Chettri *et al.*, 2001, 2005; Acharya, 2008, Acharya *et al.*, 2010, 2011a; Acharya and Vijayan, 2007, 2010, 2017; Ganguli-Lachungpa *et al.*, 2007; Sathyakumar *et al.*, 2011), has mainly focused on the PAs and forest ecosystems. The effects of fragmentation and habitat change on bird community of Kanchendzonga Biosphere Reserve (KBR) were explored by Chettri (2001). Ganguli-Lachungpa *et al.* (2007) compiled endemic and threatened species of birds in the eleven Important Bird Areas (IBAs) of Sikkim. Acharya and Vijayan (2007) reported the range extension of Rusty-bellied Shortwing in Sikkim. Similarly, studies such as status and distribution of endemic and threatened birds of the Eastern Himalaya found in Sikkim (Acharya and Vijayan, 2010), elevational distribution pattern and the underlying causes (Acharya *et al.*, 2011a) and vertical stratification of birds in different vegetation along the elevation and their determinants (Acharya and Vijayan, 2017) were also undertaken. These studies found highest diversity of birds at the mid-elevation at around 2000 m with most species having narrow range sizes. Researchers have undertaken explorations of avifauna of some of the PAs in Sikkim. Birds of Shingba Rhododendron Sanctuary, a high altitude IBA in Sikkim (>3200 m), galliformes of KBR were studied by different researchers (Acharya *et al.*, 2010; Sathyakumar *et al.*, 2011). The literature on birds of Sikkim mentioned above primarily focused on forest ecosystems or PAs. Despite high management diversity, the biodiversity of cultivated systems have not been explored (for any taxa) in Sikkim till date.

2.4.2. Butterflies

More than 18,000 species of butterflies are described worldwide (IUCN SSC Butterfly Specialist Group, 2020) out of which 1328species are reported from India (Varshney and Smetacek, 2015; Kehimkar, 2016; Personal communication with Peter Smetacek), and 690 species and sub-species in Sikkim (Haribal, 1992; Kunte, 2010; Acharya and Vijayan, 2011b, 2015). Since the pioneering work of Hooker (1855) in the middle of 19th century, the rich butterfly fauna of Sikkim (and adjoining Darjeeling region) has been widely studied by many naturalists, entomologists, and ecologists (reviewed in Haribal, 1992; Acharya and Vijayan, 2011b). Based on the extensive field survey, review of past literatures and examination of specimens from natural history museums within and outside India, Haribal (1992) provided the most comprehensive account on butterflies of Sikkim Himalaya including their natural history. The information published in the form of a book described 689 species and sub-species of butterflies. Additionally, *Symbrenthia silana* has also been reported from Sikkim Himalaya (Wynter-Blyth, 1957; Kunte, 2010). Rediscovery (after about 100 years) of four butterfly species were made from Sikkim recently (Kunte, 2010; Rai *et al.*, 2012; Dewan *et al.*, 2018). Researchers continue to discover butterfly species new to science from Sikkim e.g., *Zographetus dzonguensis* (Karmakar *et al.*, 2021), taking total butterfly species and sub-species of the region to more than 690.

Detailed ecological study on butterfly fauna of Sikkim up surged in recent years. Chettri (2010, 2015) studied the butterfly diversity and richness in the KBR of Sikkim and reported 189 species. The study found decline in species from the disturbed warm temperate broadleaf forest to undisturbed cool temperate sub-alpine forest. Similarly, the butterfly species diversity, richness and evenness significantly differed between the forest types and showed negative correlation along altitudinal gradients. Acharya and Vijayan (2011b) conducted ecological study of butterflies at various vegetation zones in the Teesta Valley in Sikkim. Similarly, Acharya and Vijayan (2015) analysed data on 161 butterfly species in different vegetation types along the elevation gradient of 300-4700 m in the Teesta valley of Sikkim. The study reported declining trend in species richness along the elevation gradient with a hump at 1000m. Recently, Dewan *et al.* (2021) reported declining trend of species richness and density of butterflies for total as well as different sub-groups (except Riodinidae and Palaearctic species) along the elevation gradient (300-3300 m) in Rangeet valley of Sikkim. Annual temperature and actual evapo-transpiration (AET) were the most important determinants reflecting the importance of energy and productivity for butterfly distribution in the eastern Himalayan elevational gradient (Acharya and Vijayan, 2015; Dewan *et al.*, 2021). Although the ecological studies on butterflies (like birds) of Sikkim has significantly increased in the past two decades. However, these previous studies mostly focused on natural forests and PAs, and there was no attempt to study butterflies of the human-modified landscapes despite their significant contribution in maintaining rich biodiversity. Therefore, present study will aid in understanding the role of cultivated systems in biodiversity conservation with reference to birds and butterflies in the Eastern Himalaya in India. It will also help to identify the most effective conservation strategy (land-sparing, land-sharing, or combination of both) applicable in the Himalayan landscape, more particularly in the Sikkim Himalaya.

Chapter 3

OVERVIEW OF THE STUDY AREA

A typical agricultural landscape in the low- and mid- hills of Sikkim

3.1. Study region

The study was undertaken in Sikkim (27° 03' to 28° 07' N and 88° 03' to 88° 57' E), Eastern Himalaya, India. Sikkim is an important part of globally significant biodiversity hotspot of Himalaya (Mittermeier *et al.*, 2011), Eastern Himalaya agrobiodiversity hotspot (Sharma *et al.*, 2016a) and Endemic Bird Area (EBA) (Stattersfield *et al.*, 1998). Despite its small geographical area (7096 km^2) and North to South extent (~100 km), the region is endowed with a very high elevation gradient (300 m to 8586 m, i.e., Mt. Khangchendzonga: the third highest mountain peak in the world) (ISFR, 2019; Chettri and Acharya, 2020). It also harbours diverse vegetation types (tropical semi-deciduous and tropical wet forests at lower valleys, temperate broad-leaved and temperate coniferous forests at mid-elevation, and sub-alpine and alpine vegetation at the higher reaches) (Haribal, 1992; Acharya and Sharma, 2013; ISFR, 2019; Chettri and Acharya, 2020).

The region receives the highest rainfall (2700 mm to 3800 mm), within the Himalaya mainly because of south-west Monsoon (Ali, 1962; Acharya and Vijayan, 2015). Along the elevational gradient, Sikkim witnesses high variation in climate from hot tropical at lower valleys (<900 m), followed by subtropical, temperate, sub-arctic and arctic at Greater Himalaya (>4500 m) (Acharya and Sharma, 2013; ISFR, 2019). Along the elevation gradient in Sikkim, there is decline in temperature (from 28°C to sub-zero) monotonically at the rate of −0.62°C per 100 m rise in elevation, as well as in precipitation from ~ 3,800 mm rainfall to <500 mm (Acharya *et al.*, 2011a; Acharya and Vijayan, 2015). Four major seasons are prevalent in the region, viz., pre monsoon (March-May), monsoon (June-August), post monsoon (September-November) and winter (December-February) (Acharya and Vijayan, 2015).

3.1.1. Vegetation

The vegetation of Sikkim has been divided into following six major types based on floristic characteristics and elevation (Haribal, 1992; Acharya and Sharma, 2013):

(a) Tropical semi-deciduous forests (300-900 m) mainly constitute *Shorea robusta*, *Tectona grandis, Terminalia balerica*, etc. with secondary growths of *Barleria* sp., *Tridex* sp., *Polygonum* sp., and also *Musa* sp., *Pandanus* sp. in the inner valleys. These forests are largely altered for agriculture and teak plantation but remnant patches of original forest still exist in few places.

(b) Tropical moist and broad-leaved forests (900-1800 m), includes broad-leaved tree species such as *Schima wallichii*, *Castanopsis* spp., *Litsea* spp., etc. and secondary growths of *Girardinia* spp., *Maesa* spp., *Melastoma* spp., etc. and tree ferns. The majority of these forests are partially disturbed for cardamom plantation but it still harbours high biodiversity.

(c) Temperate broad-leaved forests (1800-2800 m), one of the most undisturbed forests types characterized by closed canopy of broadleaved trees predominated by *Rhododendron* spp. and *Michelia* spp. covered by mosses and other epiphytes with undergrowth shrubs mainly consisting of *Debregeasia* spp., *Urtica* spp., *Mahonia* spp., *Berberis* spp., etc. along with bamboos.

(d) Temperate coniferous forests (2800-3800 m) are dominated by coniferous trees such as *Tsuga dumosa* and *Abies densa* with undergrowths of diverse Rhododendron species, but in some localities *Thuja* spp. also occurs.

(e) Sub-alpine (3800-4500 m) forests are mainly dominated by stunted coniferous species of Junipers and smaller shrubs such as Rhododendron spp. as well as *Abies densa*. The tree line generally lies between 3800-4000 m in Sikkim.

(f) Alpine vegetation (>4500 m) represents the Greater Himalaya with seasonal plants/herbs typical of high altitude cold desert having short life cycle during monsoon season e.g., species of *Meconopsis*, *Potentilla*, *Saussurea*, etc.

The vegetation adjoining the IFS in the low- to mid-hills selected in this study comprises of three major types, viz., (a) Tropical semi-deciduous forests, (b) Tropical moist and broad-leaved forests, and (c) Temperate broad-leaved forests (Acharya and Sharma, 2013; Table 3.1). Natural Forests is the major land use in Sikkim with nearly 82.31% of State's TGA under the administrative control of the Forest Department (ISFR, 2019). The area under forest cover has been consistently increasing and presently stands at 47.11% (ISFR, 2019). The PAs includes seven Wildlife Sanctuaries and one National Park namely the Khangchendzonga National Park (recently inscribed as a UNESCO World Heritage Site under mixed category) constituting 30.77% of TGA of Sikkim (Acharya and Sharma, 2013; ISFR, 2019).

3.1.2. Agroecosystems of Sikkim

The Eastern Himalaya spreads over a wide spectrum of ecological zones representing diverse socio-economic potential and biodiversity value. Sharma and Kerkhoff (2004) described five major agro-ecosystems of the Eastern Himalaya: Pastoralism, agro-pastoralism, mixed farming systems, shifting cultivation and commercial cash crops cultivation. The agro-ecosystems of Sikkim has been classified and described (Sharma and Acharya, 2013; Sharma *et al.*, 2016a) into the following seven major types that stretches across the elevation range of 300 m to >5500 m (Table 3.1):

(a) Terrace-rice farming system is socio-economically significant agroecosystems of Sikkim within the elevation range of 300-1500m. It is found in the flat land riverbanks of Teesta, Rangeet and their tributaries, and the typical terraced slopes in the lower hills. Diverse traditional varieties of rice, the stable food crop of the region are cultivated in this agroecosystem.

(b) Mandarin orange-based agroforestry system (MOAS) is socio-economically well-regarded agroforestry in Sikkim within the elevation range of 600-1500m. *Citrus reticulata* is a high value, comparatively less labour intensive crop planted with N_2-fixing *Albizia* spp. and other agroforestry trees. It also involves several multilayer fruit species as well as mixed intercropping with maize, pulses, ginger and many other crops.

(c) Farm-based agroforestry system (FAS) is the primary agri-silvicultural system of Sikkim within the elevation range of 600-2500 m that comprise of home gardens, traditional beekeeping and livestock as the principal components. It also maintains farm forests (retaining diverse multipurpose trees, bamboo groves, etc.) that support nutrients and organic matter to the farm.

(d) Large cardamom-based agroforestry system (LCAS) is ecologically suitable and economically most remunerative agroforestry in Sikkim. Large cardamom (*Amomum subulatum*) is cultivated within the elevation range of 600-2400 m as an understorey perennial crop grown under the shade of nitrogen-fixing native Himalayan alder *Alnus nepalensis*, and other mixed tree species such as *Albizia* spp., *Terminalia myriocarpa*, *Viburnum cordifolium*, *Nyssa javanica*, *Schima wallichi*, *Exbucklandia populnea*, *Maesa chisia*, etc. Being mostly contiguous with natural forests, LCAS plays dual role of providing habitat for animals as well as develop a mosaic of ecosystem that enhances overall biodiversity (Chettri *et al.*, 2005). Since the old traditional LCAS (where the current study was conducted) mostly declined due to the infestation of viral disease, the cardamom is being planted in other types of agricultural lands such as rice fields, FAS, etc. as an open farming system. Such practice has resulted in decline in vegetation cover and functionality of the system, and might be poor in retention of varied biodiversity components compared to traditional LCAS.

(e) Mixed farming (subsistence agriculture) system is found within the elevation range of 2500-4000 m where the local communities practice subsistence agriculture by cultivating a variety of crops (barley, wheat, millet, etc.), vegetables, fruits and different kinds of medicinal plants. The local communities also collect non-timber forest products (NTFP) for various uses such as food, medicine, fodder and fibre that are locally consumed, and used as commercial commodities.

(f) Agropastoralism is the major agro-ecosystems within the elevation range of 4000-5500 m prevalent in some parts of North and East Sikkim. It involves both animal rearing and cultivation of agri-horticultural crops. Tracking the seasonal fluctuation in grazing pasture, the local indigenous communities move seasonally with their herds of cattle and sheep, upward during summer and downward during winter.

(g) Pastoralism is found in the Tibetan plateau areas in North Sikkim above 5500 m. The indigenous communities are involved in animal rearing mainly Yaks. Dzumsa, the local governance mechanism has helped in sustainably managing the allocation of grazing pasture lands and harvesting of medicinal plants in the area (Acharya and Sharma, 2012).

The IFS selected in this study comprised of three major types, namely FAS, MOAS and LCAS (Table 3.1). These IFS are globally recognised as an associate site under GIAHS by FAO, UNO (GIAHS, 2007). Sikkim is declared as the first fully organic farming state of India (Bhutia, 2015). The state has approximately 74,190.86

ha agricultural land (owned by 64,726 households) that make up 10.45% of the total geographical area of the state (Bhutia, 2015).

Table 3.1. Comparative accounts of elevational distribution of the vegetation and agroecosystem types of Sikkim, Eastern Himalaya, India.

Vegetation types	Elevation (m)	Agroecosystem types (IFS)	Elevation (m)
#Tropical semi-deciduous	300-900	Terrace-rice farming system	300-1500
#Tropical moist and broad-leaved	900-1800	$Mandarin orange-based agroforestry system	600-1500
#Temperate broad- leaved	1800-2800	$Farm-based agroforestry system	600-2500
Temperate coniferous	2800-3800	$Large cardamom-based agroforestry system	600-2500
Sub-alpine	3800-4500	Mixed farming (subsistence agriculture) system	2500-4000
Alpine	>4500	Agropastoralism	4000-5500
		Pastoralism	>5500

#: forest types and $: indigenous farming systems included in the present study.

Based on climatic conditions and soil characteristics, India has been divided into 20 agro-ecological zones (AEZs). Sikkim falls under the Warm pre-humid ecoregion with brown and red hill soils, which covers whole of Sikkim and Arunachal Pradesh and Himalayan foothills of Assam and West Bengal. This AEZ is characterized by high precipitation (2000-4000 mm), lesser potential evapo-transpiration (<1000 mm) and longer growing seasons (>210 days) (Sehgal *et al.*, 1992).

3.1.3. Biogeography of birds and butterflies

The Eastern Himalayan region represents the transition zone between Indian, Indo-Malayan and Indo-Chinese biogeographic regions (Mani, 1974). When compared to other mountain ranges of the world, biogeography of Himalaya is peculiar due to its widest elevation gradient, Tertiary orogeny, the Pleistocene glaciations and continued Post-Pleistocene uplift (Mani, 1974). The Unique biogeography of the Himalaya is dominantly determined by the atmospheric temperature, and monsoon-mediated precipitation, resulting into wide range of ecological conditions, the elevational zonation of life forms, the east-west gradations of ecosystems and distributional patterns (Mani, 1974; Pandit, 2017).

The Himalaya stretching ~3000 km east to west differs between eastern and western parts in many respects. Firstly, general climatic conditions are mostly semi-oceanic in the East, but continental in the west (Mani, 1974). Secondly, the forest cover and PAs coverage is higher in the Eastern Himalaya than the Western part (Mani, 1974; ISFR, 2019). Thirdly, the biodiversity and endemism is higher, including birds, butterflies and plants in the Eastern Himalaya than the Western Himalaya (Mani, 1974; Behera *et al.*, 2002; Chettri *et al.*, 2010b; Pandit, 2017). Fourthly, the Eastern Himalaya has genera and species dominated by the oriental fauna (Indo-Chinese and Malayan elements), which has spread sparsely to westwards, in contrast to the Palaearctic-Ethiopian genera and species dominant in the Western Himalaya spreading sparsely eastwards.

The Eastern Himalaya is richer in birds than western Himalaya, especially among the tropical and oriental elements (Mani, 1974). In the Eastern Himalayan EBA (Stattersfield *et al.*, 1998), a total of 22 endemic/range-restricted bird species are found out of which 10 species have been reported from Sikkim (Acharya and Vijayan, 2010; Grimmitt *et al.*, 2019). Similarly, Eastern Himalaya is also exceptionally rich in butterflies, dominated by Indo-Chinese and Indo-Malayan forms, although some Sino-Japanese and Palaearctic forms are present in the higher elevations (Mani, 1974; Dewan *et al.*, 2021). The butterfly fauna below 1800 m elevations are truly oriental with flight period of March-November, whereas about 90% of species above 3000 m elevation are truly Palaearctic with flight period of June-August (Haribal, 1992; Kehimkar, 2016). The region below 1800 m elevation (that covers 30% TGA of the state) harbours more than 75% of the total butterfly species of Sikkim with affinity to oriental fauna (Haribal, 1992). Buterflies from Hesperiidae, Lycaenidae, and sub-family Satyrinae of Nymphalidae, e.g., genera *Lethe* are highly diversified in the Eastern Himalaya (Mani, 1974; Haribal, 1992). Some of the species and sub-species, e.g., *Parnassius imperator agustus*, *Parnassius acco hunningtoni*, *Lethe trisigmata, Lethe atkinsonia* are endemic to the Eastern Himalaya (Haribal, 1992).

3.2. Study ecosystems

Four major study ecosystems were selected for the present study along a gradient of shade tree diversity within the elevation of 600-2000 m above mean sea level (AMSL) covering east and south districts in Sikkim. Based on the comprehensive classification of the IFS of Sikkim (Sharma and Acharya, 2013; Sharma *et al.*, 2016a), the present study was designed to cover three representative agroecosystems namely MOAS, FAS, LCAS, along with nearby Forests (which in turn covered Tropical semi-deciduous forest, Tropical moist and broad-leaved forest and Temperate broad- leaved forest) (Table 3.1; Photo plate 3.1) representing agroecosystem-forest gradient.

3.3. Study site selection

Based on availability of different ecosystems, three different sites for each type of three IFS (MOAS, FAS, LCAS) and adjoining Forests ecosystems (Forest: as control) were selected from the two districts (East and South) of Sikkim for bird and butterfly sampling (Fig. 3.1; Table 3.2).

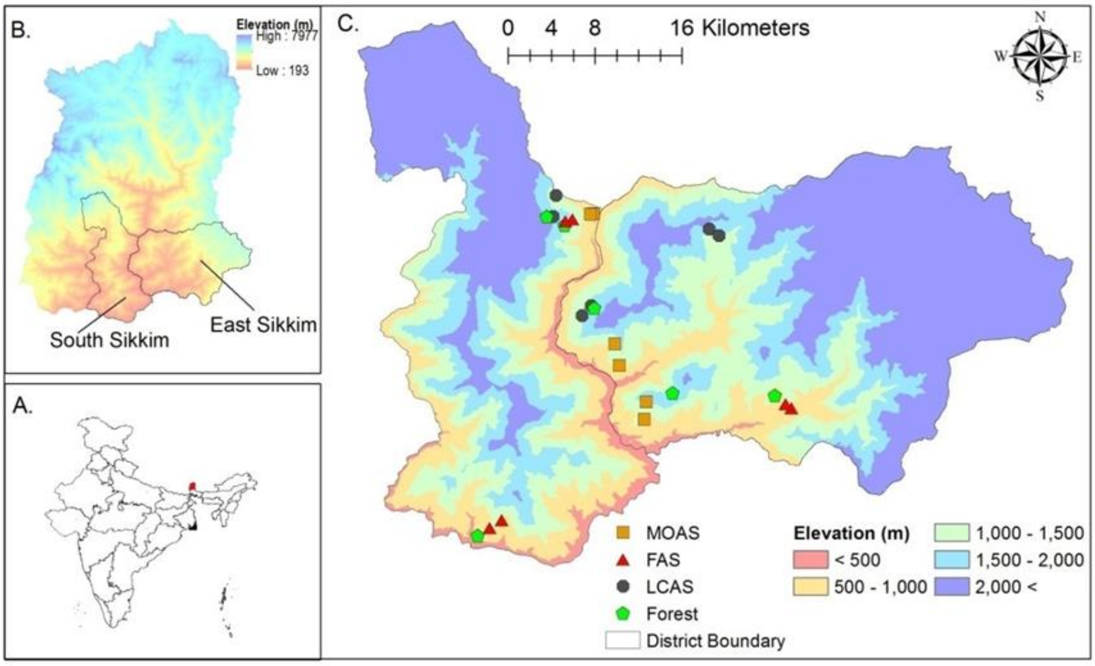


Fig. 3.1. Map of the study area showing the sites along agroecosystem-forest gradient (C) of Sikkim (B), Eastern Himalaya, India (A). MOAS - Mandarin Orange-based Agroforestry Systems, FAS - Farm-based Agroforestry Systems, LCAS - Large Cardamom-based Agroforestry Systems, and Forest refers to Natural forests.

Two transects of 1km length were laid at each site keeping a minimum of 1km distance between two transects thus making six transects per system, totalling 24 transects across the four systems (Table 3.2). Ten permanent points were established along each transect for sampling bird and butterflies maintaining 100 m distance between the two consecutive points. Hence, 240 permanent points were marked covering the various IFS and forest ecosystem. The selected 24 transects (with six per system) were situated within the elevation of 600-2000 m AMSL, with transects belonging to MOAS and FAS having comparatively lower elevation than those from LCAS and Forest (Table 3.2).

Table 3.2. Details of transects laid in different IFS and Forest of Sikkim, Eastern Himalaya, India. Mandarin orange-based agroforestry system (MOAS); Farm-based agroforestry system (FAS); Large cardamom-based agroforestry system (LCAS); and Natural forests (Forest).

Transect	**System**	**Location**	**District**	**Elevation (m)**	**Latitude**	**Longitude**
T1	MOAS	Lingee Suntaley	South	750	27° 22' 24"	88° 28' 58"
T2	MOAS	Lingee Karjee	South	850	27° 22' 23"	88° 28' 29"
T3	MOAS	Lower Pendam	East	1000	27° 12' 08"	88° 31' 21"
T4	MOAS	Sang Sakim	East	1100	27° 14' 49"	88° 30' 00"
T5	MOAS	Sang Bhirkuna	East	1250	27° 15' 54"	88° 29' 46"
T6	MOAS	Upper Pendam	East	1540	27° 13' 00"	88° 31' 29"
T7	FAS	Lower Bering	East	700	27° 12' 36"	88° 39' 34"
T8	FAS	Upper Bering	East	800	27° 12' 48"	88° 39' 15"
T9	FAS	Sumbuk Kamarey	South	980	27° 07' 12"	88° 23' 21"
T10	FAS	Sumbuk	South	1030	27° 06' 49"	88° 22' 40"
T11	FAS	Gumpa Dara Upper Payong	South	1450	27° 22' 09"	88° 27' 29"
T12	FAS	Green Village Upper Payong	South	1550	27° 22' 03"	88° 27' 06"
T13	LCAS	Lower Luing	East	1200	27° 21' 34"	88° 35' 06"
T14	LCAS	Upper Luing	East	1280	27° 21' 14"	88° 35' 40"
T15	LCAS	Lower Khamdong	East	1500	27° 17' 20"	88° 27' 58"
T16	LCAS	Pantharey Upper Payong	South	1610	27° 23' 19"	88° 26' 34"
T17	LCAS	Simkharka Upper Payong	South	1750	27° 22' 17"	88° 26' 24"
T18	LCAS	Upper Khamdong	East	2000	27° 17' 49"	88° 28' 28"
T19	Forest	Sumbuk Forest	South	640	27° 06' 25"	88° 22' 00"
T20	Forest	Tareythang Forest	East	850	27° 13' 13"	88° 38' 40"
T21	Forest	Kaw Forest	South	1597	27° 21' 50"	88° 27' 3"
T22	Forest	Tumin Forest	East	1800	27° 17' 42"	88° 28' 39"
T23	Forest	Sumin Forest	East	1850	27° 13' 25"	88° 32' 58"
T24	Forest	Upper Payong Forest	South	2000	27° 22' 17"	88° 26' 03"

Photo plate 3.1. Different representative indigenous farming systems and adjoining forest ecosystems considered for present study in Sikkim, Eastern Himalaya: Mandarin orange-based agroforestry systems (A); Farm-based agroforestry systems (B); Large cardamom-based agroforestry systems (C); and Natural forests (D).

Chapter 4

ALPHA AND BETA DIVERSITY OF BIRDS ALONG AGROECOSYSTEM-FOREST GRADIENT

White-browed Piculet *Sasia ochracea*

4.1. Introduction

Anthropogenic global change has greatly threatened the biodiversity and ES provisioning, especially in the tropical mountains, including the Himalaya (Newbold *et al.*, 2013, 2015; Pandit, 2017; Peters *et al.*, 2019). Such biodiversity loss in birds results due to diversity decline within a habitat (alpha diversity: especially for forest specialist, endemic and protected species) (Waltert *et al.*, 2004, 2011; Sreekar *et al.*, 2015). Biotic homogenization driven by trait filtering (in favour of generalists) leads to decline in diversity between habitats (beta diversity) and then at the regional level (gamma diversity) (Devictor *et al.*, 2008; Karp *et al.*, 2012; Almeida *et al.*, 2016; Gámez-Virués *et al.*, 2016). All these phenomena ultimately results in disruption of key avian-mediated ES (Şekercioğlu, 2006, 2012; Bregman *et al.*, 2016).

The retention or decline of bird diversity and associated ecosystem functioning in the agricultural landscape is linked with level of intensification, presence/ absence of native shade trees and appropriate management practices in the agroecosystems. Bird diversity, community composition and associated ES declines along the intensification gradient (from low intensive to highly intensive) in agriculture-dominated (Philpott *et al.*, 2008; Doxa *et al.*, 2010; Karp *et al.*, 2012), and forested landscapes (Newbold *et al.*, 2015; Shahabuddin *et al.*, 2021). The polyculture agroecosystems (e.g., shaded/rustic coffee, shaded cacao, jungle rubber, shaded tea, and other multi-strata agroforestry systems) helps in conserving bird diversity including land use sensitive guilds and associated ES instead of their monoculture counterparts (Perfecto *et al.*, 2003, Beukema *et al.*, 2007; Harvey and Villalobos, 2007; Lin *et al.*, 2012; Prabowo *et al.*, 2016; Chang *et al.*, 2019). Finally, agroecosystems adopting wildlife-friendly (i.e., organic, traditional, and high nature value) management practices sustain higher bird diversity (including land use sensitive sub-groups) than conventional farmlands (Philpot *et al.*, 2007; Waltert *et al.*, 2011; Doxa *et al.*, 2012; Goded *et al.*, 2018; Katayama *et al.*, 2019; García-Navas *et al.*, 2020), monoculture plantations (Mandal and Raman, 2016), or even adjoining forest ecosystems (Buechley *et al.*, 2015). The wildlife-friendly agroecosystems also mitigates biotic homogenization in bird communities and prevent disruption of avian-mediated ES provisioning (Bregman *et al.*, 2016; Kross *et al.*, 2016; Şekercioğlu *et al.*, 2016; Lindell *et al.*, 2018), and bolsters landscape-scale ecosystem multifunctionality (van der Plas *et al.*, 2018).

At the large spatial scale, landscape heterogeneity (Fischer *et al.*, 2011; Hiley *et al.*, 2016), natural forest remnant and percentage of primary forest (Anand *et al.*, 2010), proximity to forests/PAs (Anand *et al.*, 2008), structural complexity, high floristic diversity and close resemblance to forest ecosystems (Schroth *et al.*, 2004) also promote high bird diversity. The tree species richness and tree density (Clough *et al.*, 2009, Buechley *et al.*, 2015; Karanth *et al.*, 2016), tree basal area (Lee and Carrol, 2018), canopy cover (Anand *et al.*, 2008; Buechley *et al.*, 2015) are the important determinants at the local scale. The climatic factors such as rainfall and temperature (Acharya *et al.*, 2011a; Echeverri *et al.*, 2019; Srinivasan *et al.*, 2019), as well as elevation (Acharya *et al.*, 2011a) are also potential determinants.

Land coverage under agriculture is ~38% globally, ~60% in India, but only 10.45% in Sikkim (Bhutia, 2015; World Bank, 2019). Conversely, coverage of PAs stands at ~13% globally, 5.8% in India, but 30.77% in Sikkim (Watson *et al.*, 2014; ISFR, 2019). This impressively high PA coverage in Sikkim is, however, poorly represented in the tropical and subtropical belts, which constitutes mosaic landscape (forest patches within agroecosystems)**,** where agroecosystems is a dominant land-use type. Previous studies within forests and PAs along the wide elevation gradient of

Sikkim have reported high diversity of birds (Acharya *et al.*, 2011a; Acharya and Vijayan, 2011, 2017) and other taxa (Chettri *et al.*, 2010; Acharya *et al.*, 2011b; Acharya and Vijayan, 2015; Dewan *et al.*, 2021) below 1800 m elevation. The indigenously managed farming system of Sikkim is unique and has received global recognition. Such diverse systems are presumed to harbour high biodiversity including birds but there was no attempt to study bird communities in the agroecosystems, specifically, the effect of land use on alpha and beta diversity, different land use sensitive guilds and avian-mediated ES providers in the region.

Birds are well-studied bio-indicators to predict the health of ecosystems (Kremen, 1992; Schulze *et al.*, 2004a; ILTEO, 2015; Herrando *et al.*, 2016). They are often used as indicator taxa to assess biodiversity value of forests and human-modified ecosystems. The response of bird communities to land-use change varies at different spatial (e.g., local vs. regional) (Karp *et al.*, 2012) and temporal (breeding vs. dispersal or wintering season) scales (Elsen *et al.*, 2018; Yabuhara *et al.*, 2019) as well as based on their response traits. Bird species with specialized traits such as narrow geographical distributional range (e.g., endemic/range restricted), specialized requirements of habitat (e.g., Forest specialist), diet (e.g., insectivores, frugivores), microhabitat (e.g., forest specialist understorey insectivores), and protected/threatened species are highly sensitive to land use change (Şekercioğlu, 2006, 2012; Sodhi *et al.*, 2011; Sreekar *et al.*, 2015; Chiawo *et al.*, 2018) than their generalist counterparts.

The response of bird communities to land use change linked to agriculture or any other disturbance or along environmental gradients are not properly understood in the eastern Himalaya (but see Acharya *et al.*, 2011; Srinivasan *et al.*, 2019). These effects will be better deciphered when studies considers the response of different land use sensitive, and avian-mediated ES providers across spatio-temporal scales. Apart from evaluating alpha diversity, there is also a need to assess the patterns and drivers of both pair-wise beta diversity and multiple-site beta diversity in the study area (Baiser *et al.*, 2012; Baselga, 2017). The pair-wise beta diversity declines along the land-use/ agricultural intensification gradient, but shows increasing/stable trend in wildlife-friendly agroecosystems/ heterogeneous landscapes, thus allows in detecting taxonomic homogenization and heterogenization, respectively in the community (Baselga, 2010; Baiser *et al.*, 2012; Karp *et al.*, 2012). The multiple-site beta diversity helps to understand overall compositional heterogeneity of communities and identify most effective conservation strategies at landscape level, with dominance of substitution or nestedness components, indicating respectively, the need for focusing on all ecosystems or the richest ecosystem (Dobrovolski *et al.*, 2012; Baselga, 2017).

I hypothesized that organic and traditionally managed agroecosystems of Sikkim, Eastern Himalaya, at varied scale, can support high bird alpha and beta diversity and avian-mediated ES. Therefore, aim of this chapter is: (1) to understand the patterns of alpha diversity for both total bird communities and the different land use sensitive guilds (also for different avian-mediated ES and across seasons) along agroecosystem-forest gradient of Sikkim, Eastern Himalaya; (2) to understand the patterns of bird beta diversity; and (3) to explore plausible determinants of bird diversity (both alpha and beta) in the study area.

4.2. Materials and methods

4.2.1. Study area and site selection

As detailed in chapter 3, the present study covered three representative traditionally managed and organic agroecosystems, namely MOAS, FAS, and LCAS, along with the adjacent natural Forests ecosystems (Forest: as control) within the elevation of 600-2000m in two districts (East and South) of Sikkim, Eastern Himalaya, India. These four ecosystems represent the agroecosystem-forest gradient in the study area differing in shade tree species richness, tree density, tree basal area and percentage canopy cover. A total of 24 transects were sampled for birds across four ecosystems: two transects of 1km length in each site spaced at least 1km apart.

4.2.2. Bird sampling

The birds were sampled following Open-width point count method (Bibby *et al.*, 2000; Raman, 2003; Acharya *et al.*, 2011a; Acharya and Vijayan, 2017) along each of the 24 transects established for this study. Sampling was undertaken by halting at the pre-established permanent point and recording the identity and abundance of birds detected visually and acoustically for a 10-minutes period. I observed birds using a binocular, and if necessary (and possible) photographed them with Digital SLR Camera. The sampling was done on clear days in the morning hours (06:00 hrs to 09:00 hrs). Each point was sampled 6-9 times covering four seasons viz., pre monsoon (March-May), monsoon (June-August), post monsoon (September-November) and winter (December-February) along the temporal range from December 2012 to August 2017. The total sampling effort was 2050 samples during the study period with 540 point counts each in all agroecosystems and 430 point counts in Forests.

4.2.4. Biological variables of birds

The sampled birds were identified to the species level with the help of photo plates and identifying characters provided in the standard field guides and literatures (Ali 1962; Grimmett *et al.*, 2011). The bird taxonomy follows Praveen *et al.* (2016, 2020a), which gives periodically updated account for avifauna of India. Additionally, relevant autecological traits of birds were quantified and classified (Table 4.1) following standard literatures (Ali, 1962; Ali and Ripley, 2002; Şekercioğlu, 2006; Grimmett *et al.*, 2011, 2019; Sodhi *et al.*, 2011; Sreekar *et al.*, 2015; Whelan *et al.*, 2015; Şekercioğlu *et al.*, 2016) supplemented by field observations.

Table 4.1: The autecological traits used for categorization of birds in the present study.

Sl. no.	Trait name	types
1	forest specialization	forest specialist (FS) specialized to forest interior/relatively undisturbed forest habitat and not found in open landscapes; forest generalist (FG) generally prefer forest habitat but also inhabit open landscapes; and open-land species (OA) which are found mainly in open landscapes.
1	primary feeding guild	Insectivore, frugivore, granivore, nectarivore, carnivore, omnivore consuming a majority of arthropods, fruit, seeds, nectar, vertebrates (including scavenging), and general feeders, respectively.
3	preferred foraging stratum	Canopy, midstorey, understorey, ground/terrestrial, water, air.
4	migratory status	Resident, altitudinal migrant (AlM) and long-distance migrant [(summer visitor (SV), winter visitor (WV), passage migrant (PM)]
5	avian-mediated ecosystem services	Seed dispersal (frugivorous and granivorous birds), pollination (nectarivorous-insectivorous birds), invertebrate pest control (insectivorous birds), vertebrate pest control (carnivorous birds), scavenging (carrion-feeding/omnivorous birds), nutrient deposition (piscivorous/aquatic or mountain streams foraging birds), and ecosystem engineering (cavity- and burrow-excavating birds)

The bird were considered as conservation concern species, if they featured in global threatened status of IUCN Red List 2019 (BirdLife International, 2020a), Appendices I and II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) (UNEP-WCMC, 2018), Schedule I of the Indian Wildlife Protection Act 1972 (WPA 1972) (Anonymous, 2010) or were endemic/range-restricted to the Eastern Himalaya (Stattersfield *et al.*, 1998; Acharya and Vijayan, 2010; Grimmett *et al.*, 2019; BirdLife International, 2020b).

4.2.4. Environmental variables

In order to understand the environmental determinants of bird diversity and composition in the present study, different habitat, climatic and topographical factors were quantified. Vegetation sampling was done following quadrat method. Across the 24 transects, 240 quadrats (one quadrat each at 10 permanent sampling points making 10 per transect) of size 20m*20m were laid for tree sampling. The identity and girth at breast height (gbh: at 1.37 m) of all trees (with gbh>20cm) were enumerated. The data so obtained was used to quantify tree species richness (tsr), tree density (tden.: stems ha^{-1}) and tree basal area (tba: m^2 ha^{-1}) for each of the 24 transects. The spherical convex densiometer was used to estimate percentage canopy cover (pcc: %) by taking four readings at each direction at all 10 points in a transect. The 40 readings were averaged to obtain mean pcc values for each transect. For each of the 24 transects, I also quantified mean annual temperature (MAT/temp: °C) and mean annual precipitation (MAP/rain: mm) from the WorldClim database at 1km spatial resolution (<www.worldclim.org>; Fick and Hijmans, 2017) using ArcGIS 10.4. Similarly, elevation (elev: m) and GPS coordinates (latitude: °E; longitude: °N) of each transects were obtained using hand held global positioning system (GPS).

4.2.5. Statistical analysis

All analyses were performed using R version 3.1.4 (R Core Team, 2017) using different packages.

4.2.5.1. Assessment of sampling completeness

To assess the completion of bird sampling, Chao1 (a nonparametric estimator of species richness) value was estimated on the basis of a matrix of abundance data. The

Chao1 was chosen because of its high precision for abundance data (Hortel, 2006). The calculated value of Chao1 was plotted against sampling effort to generate species accumulation curve for total birds using "iNEXT" package in R.

4.2.5.2. Bird alpha diversity and determinants

For the bird community, alpha diversity was measured as species richness (species per point) observed in each point during sampling. I also assessed other community parameters such as abundance per point, total species richness, total abundance, and Shannon-Wiener diversity index (H′) for each of the 24 transects sampled, four ecosystems, and overall data. The observed results of total species richness, Shannon-Wiener diversity, proportion of total species richness for birds in different agroecosystems and Forest of Sikkim Himalaya were compared with past studies from Forests (including in PAs), and agroecosystems and other human-modified ecosystems (Agroecosystems) in the biodiversity hotspots of Himalaya, Indo-Burma, Western Ghats and Sri Lanka and Elsewhere. The results were plotted in the form of boxplot using packages "ggplot2" and "gridExtra" in R.

To account for slightly unequal sampling effort among study systems, I focused further analysis on alpha diversity, abundance per point and H′. One-way analysis of variance (ANOVA) was used to test variation in bird alpha diversity, abundance per point, and H′ among the ecosystems for total bird species (TBi), and different land use sensitive guilds: forest specialist insectivores sub-grouped into forest specialist understorey insectivores (FSUIBi), forest specialist midstorey insectivores (FSMSIBi), forest specialist canopy insectivores (FSCIBi), as well as forest specialist frugivores & nectarivores (FSFNBi), and conservation concern species (protected and endemic species: PEBi). To identify the particular context which actually made the difference in the parameter, post-hoc, pair-wise comparisons were done between the four ecosystems (FAS vs. MOAS; LCAS vs. MOAS; Forest vs. MOAS; LCAS vs. FAS; Forest vs. FAS; Forest vs. LCAS) by incorporating Bonferroni correction to the Type-1 threshold (alpha value) to compensate for multiple testing using “multcomp” package in R. I performed similar tests to assess the pattern in bird diversity based on avian-mediated ES (one-way ANOVA followed by Bonferroni corrected post-hoc, pair-wise test), and seasonal dynamics (two-way ANOVA followed by post-hoc, pair-wise comparisons with Bonferroni correction) among the four ecosystems. To understand the spatio-temporal pattern in bird communities, bird diversity at different spatial (at point, transect and ecosystem level) and temporal (seasons: winter, pre monsoon, monsoon, post monsoon) scales were quantified for the total species, different land use sensitive guilds, and avian-mediated ES providers. Correlation test was used to explore the relationship of bird community parameters with different habitat and environmental variables, and correlation matrix plot was prepared to depict significant ($p<0.05$) correlation using “corrplot” package in R.

4.2.5.3. Bird beta diversity and determinants

Beta diversity in bird community was partitioned into substitution and nestedness components using incidence-based (Sorensen dissimilarity) and abundance-based (Bray-Curtis dissimilarity) indices respectively for both pair-wise dissimilarity and multiple site dissimilarity (Baselga, 2010, 2013a, 2017). Original abundance based community data matrix for Bray-Curtis dissimilarity indices, and only presence/absence matrix for incidence-based Sorensen dissimilarity indices were used to estimate the beta diversity. To assess patterns of beta diversity along the agroecosystem-forest gradient, the pair-wise dissimilarity index for each transects pair was calculated. In particular, total Bray-Curtis dissimilarity (βbray) and its components due to balanced variation in abundance (βbray.bal) and due to abundance gradients (βbray.gra) as well as pair-wise total Sorensen dissimilarity (βsor) and its components of turnover (βsim) and nestedness-resultant (βsne) were estimated. The homogeneity and their significant difference among the ecosystems were assessed using one-way ANOVA. To identify the particular context driving the actual difference in pair-wise beta diversity, Tukey's HSD post-hoc, pair-wise comparisons were also carried out. To account for large heterogeneity and >2 sites in the study, I also quantified multiple site beta diversity (Baselga, 2013b). Therefore, to assess if average multiple site dissimilarity indices (β_{BRAY}, β_{SOR} and their components) differ significantly from random expectation, a resampling procedure (taking 1,000 random samples of 10 sites) was employed to compute average dissimilarity values for total bird communities based on abundance-based and incidence based dissimilarity indices (Baselga, 2010, 2017). Similar analysis was done to the different land use sensitive guilds and avian-mediated ES providers using Bray-Curtis indices.

Next, distance decay analysis was implemented to understand the relationship of bird beta diversity (overall and their components of substitution and nestedness) with the different environmental and habitat variables (Appendix A) for total bird communities (TBi), and land use sensitive guilds (FSUIBi, FSMSIBi, FSCIBi, FSFNBi, and PEBi). The multiple regression models for distance matrices (MRM) was used to assess the relationship between the matrices of overall beta diversity and their components (for both indices) and the Euclidean distance matrices of environmental and habitat variables, and the regression slopes (a), and intercepts (b) were quantified (Lichstein, 2007; Si *et al.*, 2015). Additionally, there might be possibility of spatial auto-correlation and non-independent observations of pair-wise beta diversity inflating the significance tests. Therefore, partial mantel tests (with 999 permutations) were run including geographical distance between transects as a covariate to estimate the *p*-values and the Pearson correlation coefficients (*r*) (Lichstein, 2007). The analyses on beta diversity were performed using packages “ape4”, “betapart", ecodist” and “vegan” in R.

Finally, the bird community composition for total species and the different land use sensitive guilds (also the different ES) across four ecosystems were compared through non-metric multidimensional scaling (NMDS) using Bray-Curtis distance, on which environmental variables were fitted subsequently. To assess the significance of environmental variables, Monte-Carlo randomization test with 999 permutations was used. Performance of NMDS was tested using Kruskal’s stress formula multiplied by 100 (McCune and Grace, 2002). NMDS ordination was applied using the command ‘meta-MDS’ in package “MASS” of R.

4.3. Results

4.3.1. Habitat and environmental variables

Along the agroecosystem-forest gradient in the study, tree species richness ($F_{3, 20} =$ 2.995, p=0.055; Fig. 4.1a) declined near significantly but tree basal area ($F_{3, 20} =$ 4.991, p<0.01; Fig. 4.1c) and percentage canopy cover ($F_{3, 20} =$ 9.835, p<0.001; Fig. 4.1d) significantly increased, whereas, tree density lacked significant difference ($F_{3, 20} =$ 1.38, p=0.278; Fig. 4.1b; Table 4.2). Bonferroni corrected post-hoc, pair-wise comparisons revealed significant difference for tree basal area between Forest and MOAS (estimate = 31.402; p<0.01), whereas, percentage canopy cover significantly differed between Forest and MOAS (estimate = 23.54; p<0.001), Forest and FAS (estimate = 17.18; p<0.01), LCAS and MOAS (estimate = 15.88; p<0.05) (Table 4.2).

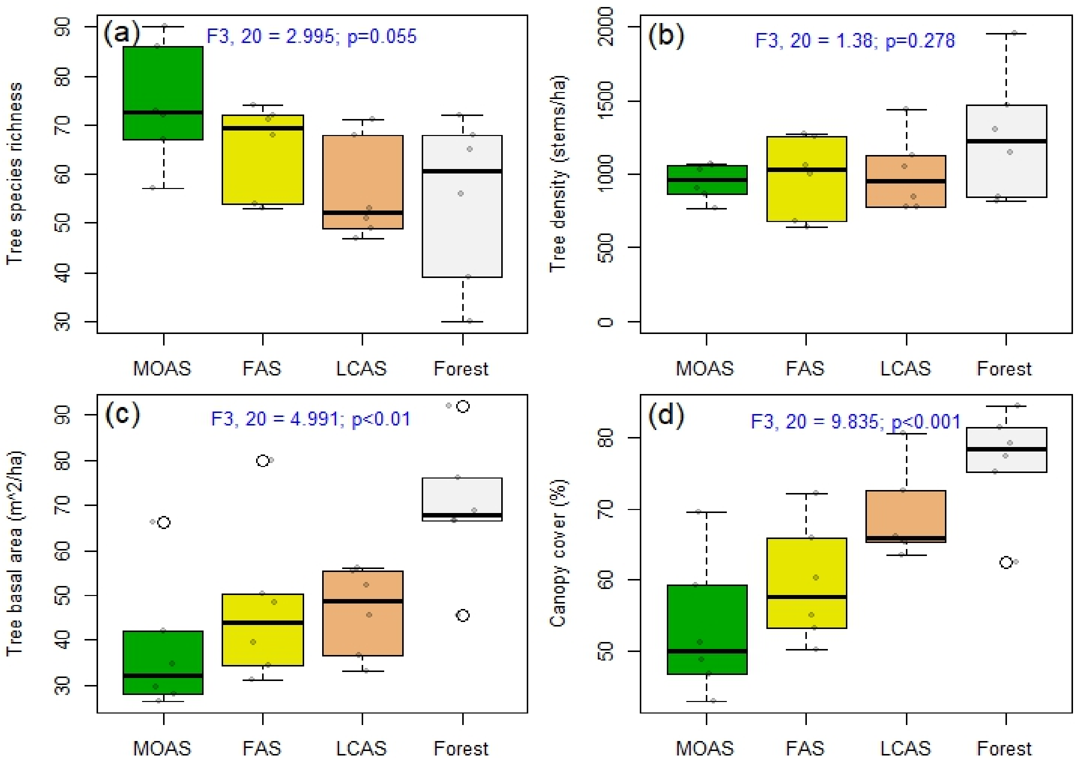


Fig. 4.1. Habitat characteristics along agroecosystem-forest gradients of Sikkim, Eastern Himalaya, India: (a) tree species richness, (b) tree density, (c) tree basal area, (d) percentage canopy cover. For results of ANOVA tests, F- statistic, degree of freedom (3: between group; 20: within group coming from Ecosystems and Transects, respectively), and significance level are also shown.

The environmental variables also varied (marginal significantly: $0.5<p<0.1$ for elevation and MAT, but non-significantly for MAP), along the agroecosystem-forest gradients of Sikkim (Table 4.2). Transects from MOAS and FAS on average were situated at comparatively lower elevation but received higher MAT and MAP when compared to those from LCAS and Forest (Table 4.2; Appendix A). Along the elevation gradient, MAT ($r^2 = -0.89$) and MAP ($r^2 = -0.84$) significantly declined. The tree species richness correlated significantly negatively with tree basal area ($r^2=-0.58$) and pcc ($r^2= -0.68$), but weak positively with tree density ($r^2=0.19$). The tree density significantly increased with elevation (0.41), but declined with MAP ($r^2=-0.57$) and MAT ($r^2=-0.37$). With elevation, there was weak positive correlation for pcc ($r^2=0.31$) and tree basal ($r^2=0.23$), whereas, tree species richness ($r^2= -0.24$) declined.

Table 4.2. Habitat and environmental variables along agroecosystem-forest gradients of Sikkim, Eastern Himalaya, India. Mean values along the gradient not sharing letters are significantly different at the alpha level of p = 0.05 based on Bonferroni corrected post-hoc, pair-wise comparisons.

Variables	$F_{3,20}$	**p-value**	**MOAS**	**FAS**	**LCAS**	**Forest**
Tree species richness	**2.995**	**p=0.055**	74.17±4.98a	65.33±3.83a	56.50±4.21a	55.00±6.93a
Tree density (stems ha^{-1})	1.38	p=0.278	948.33±50.26a	983.75±111.45a	1002.08±106.33a	1253.75±173.98a
Tree basal area (m^2 ha^{-1})	**4.991**	**p<0.01**	37.88±6.14a	47.28±7.22ab	46.38±4.01ab	69.28±6.17b
pcc (%)	**9.835**	**p<0.001**	53.11±3.95a	59.47±3.39ab	68.98±2.66bc	76.65±3.10c
Elevation (m)	2.399	p=0.098	1081.67±116.77a	1085.0±140.59a	1556.67±121.62a	1456.17±232.56 a
MAT (^{0} C)	2.81	p=0.066	19.96±0.39a	20.21±0.77a	17.33±0.42a	18.36±1.32a
MAP (mm)	1.699	p=0.199	3101.67±131.76a	3242.25±310.66a	2561.83±54.45a	2918.50±295.22a

pcc: Percentage canopy cover; MAT: Mean annual temperature; MAP: Mean annual precipitation

4.3.1. Bird species richness and diversity

I recorded 19354 individuals (in 8189 detections) of 221 resident and migrant species of birds in the present study of which 212 species were recorded in different IFS against 147 species in Forests (Table 4.3; Appendix B). The bird community was represented by 51 avian families, and three of them (Muscicapidae: 34 species; followed by Leiothrichidae: 19 species; and Phylloscopidae: 14 species), together accounted for about 30% of the total avifauna species observed in the present study (Appendix B). Among the four ecosystems, I observed highest total abundance and species richness of birds in LCAS, whereas, family richness in FAS (Table 4.3). Based on habitat specialization, bird communities were dominated by Forest specialist (125 species) compared to Forest generalist and grassland-openland birds (58 and 38 species, respectively). Based on migratory status, altitudinal migrants were the most speciose (107 species), followed by residents (84 species), and breeding (SV) and non-breeding migrants (WV & PM) were represented by 19 and 11 species respectively (Table 4.3).

Table 4.3. Patterns of alpha diversity and other community parameters of birds along agroecosystem-forest gradient of Sikkim, Eastern Himalaya, India. The values were pooled for each ecosystem and also overall. The value of Chao1 is mean ± standard deviation.

Community parameters	**Ecosystem**				**Total**
	MOAS	**FAS**	**LCAS**	**Forest**	
Total species richness (Sobs)	132	157	160	147	221
Chao1	133.47±1.68	162.04±3.51	162.14±2.1	151.59±3.58	227.48±4.34
Sobs:Chao1	98.90%	96.89%	98.68%	96.97%	97.15%
Total abundance	5104	5038	5582	3630	19354
Family richness	44	46	41	43	51
Habitat specialization					
Forest specialist	**73**	**83**	**97**	**88**	**125**
Forest generalist	36	40	41	40	58
Open-land species	23	34	22	19	38
Migratory status					
Resident	54	62	58	59	84
Altitudinal migrant	66	74	85	76	107
Breeding migrant	7	13	13	10	19
Non-breeding migrant	5	8	4	2	11
Ecosystem services					
Invertebrate pest control	100	116	123	108	164
Pollination	13	12	15	14	16
Seed dispersal	13	21	20	18	29
Waste disposal	13	14	9	14	19
Nutrient deposition	12	16	9	10	18
Ecosystem engineering	17	25	23	22	35
Land use sensitive guilds					
FSFNBi	9	12	13	14	18
FSUIBi	23	20	28	21	35
FSMSIBi	10	15	17	14	20
FSCIBi	26	29	32	31	38
PEBi	9	14	9	14	18

Ecosystems: MOAS - Mandarin orange-based agroforestry systems, FAS - farm-based agroforestry systems, LCAS - large cardamom-based agroforestry systems, Forest - Natural forests; different land use sensitive guilds: FSUIBi - forest specialist understorey insectivores, FSMSIBi - forest specialist midstorey insectivores, FSCIBi - forest specialist canopy insectivores, FSFNBi - forest specialist frugivores and nectarivores, and PEBi - protected and endemic species; IUCN Red list: NT - near-threatened, VU - vulnerable; CITES - Convention on International Trade in Endangered Species of Wild Fauna and Flora; WPA 1972 - Indian Wildlife Protection Act 1972; Sobs:chao1: ratio of observed total species richness to estimated richness based on chao1.

The bird community classified based on land use sensitive guilds revealed that 8.1% of the total avifauna (18 species) were conservation concern species (Table 4.3). Similarly, other land use sensitive sub-groups contributed 49.3% of the total avifauna, of which the most species-rich were FSCIBi, followed by FSUIBi, FSMSIBi, and FSFNBi, representing 38, 35, 20, 18 species respectively (Table 4.3). About three-fourth of the total avifauna pool were represented by species responsible for invertebrate pest control (164 species), whereas pollination, seed dispersal, waste disposal, nutrient deposition, and ecosystem engineering service providers were represented by 16, 29, 19, 18 and 35 species respectively (Table 4.3).

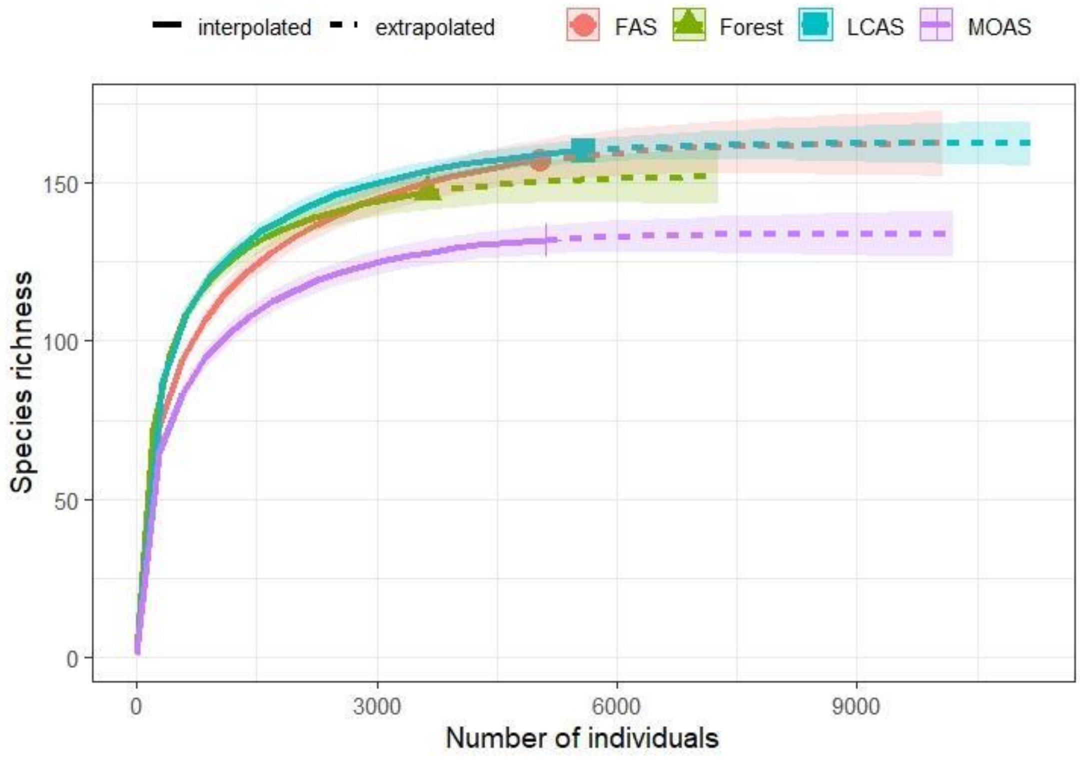


Fig. 4.2. Species accumulation curve for bird diversity (Species richness; q=0) along agroecosystem-forest gradient of Sikkim, Eastern Himalaya. Mandarin orange-based agroforestry systems (MOAS), farm-based agroforestry systems (FAS), large cardamom-based agroforestry systems (LCAS), and Natural forests (Forest).

The sampling effort for birds in the present study was adequate because individual-based rarefaction curve (including extrapolation) reached asymptote in all ecosystems (Fig. 4.2). The estimated species richness (chao1) of total birds was highest in LCAS, followed by FAS, Forest and least in MOAS (Fig. 4.2; Table 4.3). The ratio of observed species richness to estimated richness based on chao1 (TBi:chao1) values of ≥97% across the agroecosystems, Forest, and overall data, further points towards adequate sampling effort (Table 4.3).

4.3.2. Patterns of bird alpha diversity

For total birds along the agroecosystem-forest gradient, LCAS had significantly highest species per point (Fig. 4.3a), abundance per point ($p<0.01$; Fig. 4.3.g) and Shannon-Wiener diversity (Fig. 4.3m). Bonferroni corrected post-hoc tests also showed significant differences ($0.001<p<0.05$) in TBi' species per point (all IFS including LCAS vs. Forest), abundance per point (Forest vs. LCAS) and Shannon-Wiener diversity (LCAS vs. MOAS and FAS) (Fig. 4.2a, g, m). For the different land use sensitive guilds, alpha diversity was significantly highest for FSCIBi (Fig. 4.3d), FSFNBi (Fig. 4.3e), PEBi (Fig. 4.3f) in LCAS, followed by Forest, FSUIBi (Fig. 4.3b) in LCAS, followed by MOAS, however, FSMSIBi in Forest (Fig. 4.3c) followed by FAS. Bonferroni corrected post-hoc tests also showed significant difference ($0.001<p<0.05$) in alpha diversity of FSUIBi (LCAS vs. other ecosystems), FSMSIBi (LCAS vs. MOAS), FSCIBi (LCAS and Forest vs. each of MOAS and FAS), FSFNBi (LCAS vs. FAS and MOAS; Forest vs. FAS), PEBi (LCAS and Forest vs. MOAS) (Fig. 4.3b-f). Similar pattern with significant difference was found for abundance per point of FSCIBi, FSFNBi and PEBi (Fig. 4.3j-l) and Shannon-Wiener diversity of FSCIBi (Fig. 4.3p). The abundance per point of these 3 land use sensitive guilds were significantly higher ($0.001<p<0.05$) in LCAS (for FSCIBi, FSFNBi and PEBi) and Forest (for FSCIBi and PEBi), than MOAS and FAS (Fig. 4.3).

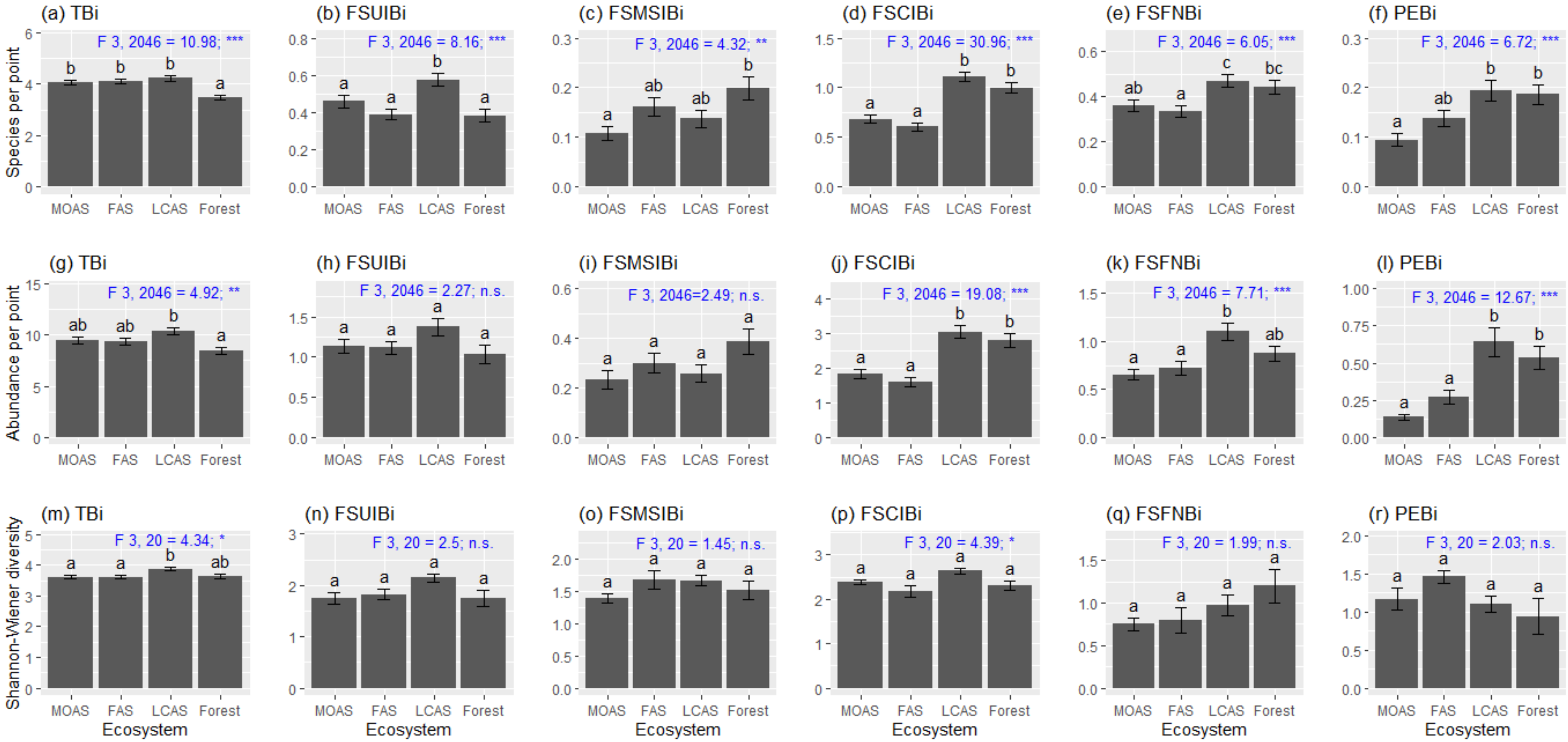


Fig. 4.3. Species richness per point, abundance per point and Shannon-Wiener diversity for total birds, and different land use sensitive guilds along agroecosystem-forest gradient of Sikkim, Eastern Himalaya: total birds (TBi), forest specialist understorey insectivores (FSUIBi), forest specialist midstorey insectivores (FSMSIBi), forest specialist canopy insectivores (FSCIBi), forest specialist frugivores and nectarivores (FSFNBi), and protected and endemic species (PEBi). For one-way ANOVA test, F-statistic, degree of freedom (between group, within group, respectively), significance level (***: p<0.001; **: p<0.01; *: p<0.05; n.s.: non-significant) are also shown. Bars not sharing letters are significantly different at the alpha level of p = 0.05; Bonferroni corrected post-hoc test.

Two-way ANOVA along with Bonferroni corrected post hoc tests also revealed marked variation in the patterns of species per point (please see Table 4.4) and abundance per point (see Table 4.5) of total birds and different land use sensitive guilds (Fig. 4.4) across seasons and ecosystems. TBi' species per point significantly differed across (1) ecosystems (only during winter season) with higher values in three IFS than Forest; and (2) seasons with higher values during post monsoon compared to pre monsoon (in all ecosystems), winter (in LCAS and Forest) and monsoon (in LCAS). TBi' abundance per point of also significantly varied across (1) ecosystems (only during winter season) with higher values in LCAS compared to FAS and Forest; and (2) seasons (only in Forest) with higher values in post monsoon than pre monsoon (Table 4.4-4.5; Fig. 4.4A, G). FSUIBi' Species per point varied significantly across (1) seasons, with ~2 times higher values during post monsoon and winter than both pre monsoon and monsoon (in MOAS), and during post monsoon than monsoon (in FAS), as well as (2) ecosystems, with higher values in LCAS compared to both FAS (during monsoon), and Forest (during winter). On the other hand, abundance per point of FSUIBi significantly differed only across seasons with ~3 times higher value during post monsoon than pre monsoon in Forest (Table 4.4-4.5; Fig. 4.4B, H). Across ecosystems, FSMSIBi' species and abundance per point were both significantly higher (all $0.001<p<0.05$) in Forest (than LCAS and MOAS); and FAS (than LCAS) during monsoon (but no significant differences in other seasons). Between seasons, FSMSIBi showed significantly higher mean values of (1) species per point during monsoon than pre monsoon (in Forest) and winter (in Forest and FAS), and during post monsoon than winter (in FAS), and (2) abundance per point during monsoon than pre monsoon (Tables 4.3-4.4; Fig. 4.4C, I). FSCIBi' species per point varied significantly across (1) ecosystems, with higher values in Forest compared to MOAS (during pre monsoon to post monsoon) and FAS (during all seasons except pre monsoon), also higher in LCAS compared to MOAS and FAS (during pre monsoon to post monsoon) (all $0.001<p<0.05$); and also (2) seasons, with higher values during post monsoon compared to pre monsoon (in MOAS), monsoon (in LCAS), and also higher during winter than pre monsoon (in MOAS). FSCIBi' abundance per point varied significantly across (1) ecosystems with higher values in Forest compared to MOAS (during all seasons except post monsoon), FAS (winter, monsoon), LCAS (monsoon), as well as higher in LCAS than MOAS (in winter and post monsoon), FAS (winter and post monsoon monsoon) and Forest (in winter); and (2) seasons, with higher values being during post monsoon than pre monsoon (in MOAS), monsoon than pre monsoon (in Forest), as well as higher in winter compared to pre monsoon (in MOAS, LCAS, Forest) and monsoon (in LCAS) (all $0.001<p<0.05$) (Tables 4.3-4.4; Fig. 4.4D, J). Across seasons, FSFNBi' species per point differed significantly in FAS with higher values during pre monsoon (than post monsoon and winter), and monsoon (than post monsoon); and in LCAS with higher values during monsoon (than winter, pre monsoon) and post monsoon (than winter). FSFNBi' abundance per point was also significantly higher in post monsoon than winter and pre monsoon in LCAS (all $0.001<p<0.05$). Both the indices differed significantly among the ecosystems with higher values in LCAS than FAS and/or MOAS during monsoon and post monsoon (all $0.001<p<0.05$) (Tables 4.3-4.4; Fig. 4.4E, K). For PEBi', only ecosystems had significant effects with LCAS showing higher species per point than MOAS (in pre monsoon and monsoon), and also abundance per point than FAS and/or MOAS (during pre monsoon and winter) (Tables 4.3-4.4; Fig. 4.4F, L).

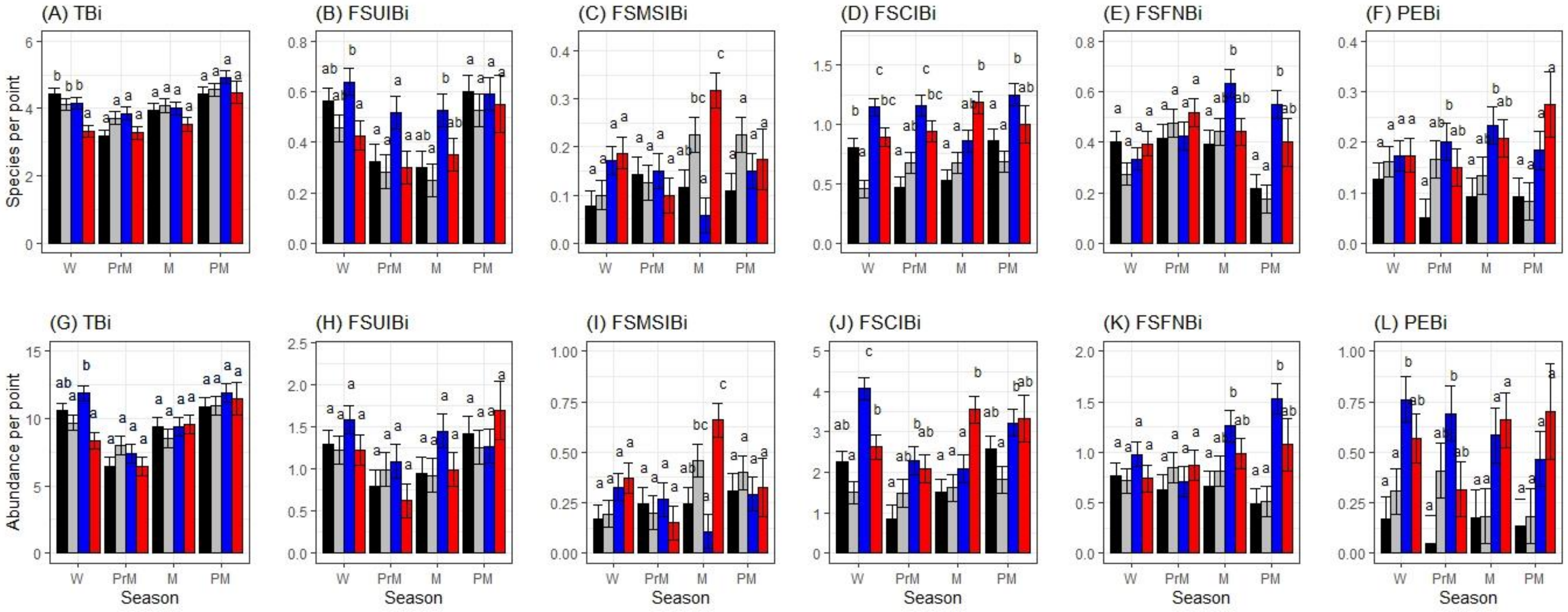


Fig. 4.4. Seasonal dynamics of bird communities in terms of species richness per point and abundance per point for total birds, and different land-use sensitive guilds along agroecosystem-forest gradient of Sikkim, Eastern Himalaya: total birds (TBi), forest specialist understorey insectivores (FSUIBi), forest specialist midstorey insectivores (FSMSIBi), forest specialist canopy insectivores (FSCIBi), forest specialist frugivores and nectarivores (FSFNBi) and protected and endemic species (PEBi). Season: Winter (W), pre monsoon (PrM), monsoon (M) and post monsoon (PM). The four land use types are: MOAS (black), FAS (grey), LCAS (blue), and Forest (red). For the results of post-hoc, pair-wise comparisons with Bonferroni correction, bars not sharing letters represents significant difference at the alpha level of p = 0.05.

Table 4.4: Results of two-way ANOVA, summary statistics and Bonferroni corrected post-hoc, pair-wise comparisons of species per point for total birds and five land-use sensitive guilds in Sikkim, Eastern Himalaya, India. Mean±SE values not sharing letters across ecosystems (horizontally: a, b, c) and seasons (vertically: X, Y, Z) are significantly different (higher: green colour; lower: red colur) at the alpha level of P = 0.05; Bonferroni corrected. df; degree of freedom; sum sq: Sum of square.

Bird guilds	**Factor**	**sum sq.**	**df**	**Estimate**	**P-value**	**Season**	**MOAS**	**FAS**	**LCAS**	**Forest**
TBi	Ecosystem (E)	110	3	**8.022**	p<0.001	**Winter**	4.44±0.17 b; Y	4.12±0.18 b; XY	4.16±0.17 b; X	3.33±0.16 a; X
	Season (S)	282	3	**20.539**	p<0.001	Pre-monsoon	3.18±0.13 a; X	3.72±0.20 a; X	3.85±0.16 a; X	3.28±0.15 a; X
	E x S	76	9	1.855	p=0.055	Monsoon	3.96±0.16 a; Y	4.08±0.17 a; XY	4.01±0.19 a; X	3.53±0.17 a; XY
	Residuals	9300	2034			Post monsoon	4.44±0.24 a; Y	4.56±0.20 a; Y	4.93±0.26 a; Y	4.47±0.48 a; Y
FSUIBi	Ecosystem (E)	11.4	3	**7.405**	p<0.001	**Winter**	0.56±0.06 ab; Y	0.46±0.05 ab; XY	0.64±0.06 b; X	0.43±0.06 a; X
	Season (S)	18.0	3	**11.692**	p<0.001	Pre-monsoon	0.33±0.05 a; X	0.28±0.06 a; XY	0.52±0.07 a; X	0.30±0.05 a; X
	E x S	3.3	9	0.713	p=0.697	**Monsoon**	0.30±0.06 ab; X	0.25±0.06 a; X	0.53±0.07 b; X	0.35±0.06 ab; X
	Residuals	1043.6	2034			Post monsoon	0.60±0.07 a; Y	0.53±0.07 a; Y	0.59±0.08 a; X	0.55±0.15 a; X
FSMSIBi	Ecosystem (E)	2.2	3	**4.556**	p<0.01	Winter	0.08±0.02 a; X	0.10±0.02 a; X	0.17±0.03 a; X	0.19±0.04 a; X
	Season (S)	0.9	3	1.866	p=0.133	Pre-monsoon	0.14±0.03 a; X	0.13±0.03 a; XY	0.15±0.03 a; X	0.10±0.03 a; X
	E x S	5.0	9	**3.471**	p<0.001	**Monsoon**	0.12±0.03 ab; X	0.23±0.05 bc; Y	0.06±0.02 a; X	0.32±0.05 c; Y
	Residuals	326.8	2034			Post monsoon	0.11±0.03 a; X	0.23±0.05 a; Y	0.15±0.04 a; X	0.18±0.08 a; XY
FSCIBi	Ecosystem (E)	95.7	3	**32.253**	p<0.001	**Winter**	0.81±0.08 b; YZ	0.46±0.06 a; X	1.14±0.09 c; XY	0.89±0.09 bc; X
	Season (S)	5.1	3	1.709	p=0.163	**Pre-monsoon**	0.47±0.06 a; X	0.68±0.09 ab; X	1.16±0.10 c; XY	0.94±0.10 bc; X
	E x S	30.2	9	**3.389**	p<0.001	**Monsoon**	0.53±0.08 a; XY	0.68±0.07 a; X	0.86±0.09 ab; X	1.18±0.10 b; X
	Residuals	2011.3	2034			**Post monsoon**	0.87±0.09 a; Z	0.68±0.08 a; X	1.25±0.11 b; Y	1.0±0.17 ab; X
FSFNBi	Ecosystem (E)	6.0	3	**5.688**	p<0.001	Winter	0.40±0.05 a; X	0.27±0.04 a; XY	0.33±0.04 a; X	0.39±0.05 a; X
	Season (S)	8.8	3	**8.326**	p<0.001	Pre-monsoon	0.42±0.05 a; X	0.48±0.06 a; Z	0.43±0.05 a; XY	0.52±0.06 a; X
	E x S	11.2	9	**3.539**	p<0.001	**Monsoon**	0.39±0.06 a; X	0.44±0.06 ab; YZ	0.63±0.07 b; Z	0.44±0.05 ab; X
	Residuals	713.7	2034			**Post monsoon**	0.22±0.04 a; X	0.18±0.04 a; X	0.55±0.06 b; YZ	0.40±0.09 ab; X
PEBi	Ecosystem (E)	3.4	3	**6.713**	p<0.001	Winter	0.13±0.03 a; X	0.16±0.03 a; X	0.17±0.04 a; X	0.17±0.03 a; X
	Season (S)	0.2	3	0.466	p=0.706	**Pre-monsoon**	0.05±0.02 a; X	0.17±0.04 ab; X	0.20±0.05 b; X	0.15±0.04 ab; X
	E x S	1.6	9	1.072	p=0.380	**Monsoon**	0.09±0.03 a; X	0.13±0.04 ab; X	0.23±0.05 b; X	0.21±0.04 ab; X
	Residuals	338.6	2034			Post monsoon	0.09±0.03 a; X	0.08±0.03 a; X	0.18±0.05 a; X	0.28±0.08 a; X

Table 4.5: Results of two-way ANOVA, summary statistics and Bonferroni corrected post-hoc, pair-wise comparisons of abundance per point for total birds and five land-use sensitive guilds in Sikkim, Eastern Himalaya, India. Mean±SE values not sharing letters across ecosystems (horizontally: a, b, c) and seasons (vertically: X, Y, Z) are significantly different (higher: green colur; lower: red colour) at the alpha level of P = 0.05; Bonferroni corrected. df; degree of freedom; sum sq: Sum of square.

Bird guilds	Factor	sum sq.	df	Estimate	P-value	Season	MOAS	FAS	LCAS	Forest
TBi	Ecosystem (E)	603	3	**3.54**	**p<0.05**	**Winter**	10.57±0.56 ab; Y	9.68±0.57 a; XY	11.87±0.79 b; Z	8.37±0.67 a; XY
	Season (S)	4440	3	**26.066**	**p<0.001**	Pre-monsoon	6.42±0.43 a; X	8.01±0.61 a; X	7.41±0.47 a; X	6.43±0.43 a; X
	E x S	852	9	1.668	p=0.091	Monsoon	9.40±0.71 a; Y	8.53±0.47 a; XY	9.38±0.67 a; XY	9.53±0.68 a; Y
	Residuals	115499	2034			Post monsoon	10.85±0.82 a; Y	10.92±0.62 a; Y	11.93±0.82 a; YZ	11.47±1.55 a; Y
FSUIBi	Ecosystem (E)	29	3	1.98	p=0.115	Winter	1.29±0.18 a; X	1.22±0.16 a; X	1.59±0.22 a; X	1.22±0.23 a; XY
	Season (S)	79	3	**5.391**	**p<0.01**	Pre-monsoon	0.79±0.12 a; X	0.99±0.23 a; X	1.09±0.18 a; X	0.63±0.12 a; X
	E x S	33	9	0.75	p=0.663	Monsoon	0.94±0.15 a; X	0.93±0.13 a; X	1.45±0.23 a; X	0.99±0.20 a; XY
	Residuals	9895	2034			Post monsoon	1.43±0.22 a; X	1.23±0.16 a; X	1.27±0.18 a; X	1.70±0.49 a; Y
FSMSIBi	Ecosystem (E)	6.9	3	**2.757**	**p<0.05**	Winter	0.17±0.06 a; X	0.19±0.05 a; X	0.33±0.07 a; X	0.37±0.09 a; XY
	Season (S)	6.8	3	**2.705**	**p<0.05**	Pre-monsoon	0.24±0.06 a; X	0.20±0.06 a; X	0.27±0.07 a; X	0.15±0.05 a; X
	E x S	20.8	9	**2.762**	**p<0.01**	**Monsoon**	0.24±0.08 ab; X	0.46±0.11 bc; X	0.11±0.05 a; X	0.66±0.14 c; Y
	Residuals	1704.9	2034			Post monsoon	0.31±0.12 a; X	0.40±0.09 a; X	0.29±0.08 a; X	0.33±0.15 a; XY
FSCIBi	Ecosystem (E)	829	3	**20.445**	**p<0.001**	**Winter**	2.26±0.24 ab; Y	1.51±0.26 a; X	4.08±0.47 c; Y	2.63±0.34 b; XY
	Season (S)	301	3	**7.428**	**p<0.001**	**Pre-monsoon**	0.85±0.14 a; X	1.48±0.23 ab; X	2.29±0.24 b; X	2.09±0.27 ab; X
	E x S	411	9	**3.374**	**p<0.001**	**Monsoon**	1.50±0.29 a; XY	1.62±0.27 a; X	2.09±0.30 a; X	3.55±0.41 b; Y
	Residuals	27504	2034			**Post monsoon**	2.57±0.35 ab; Y	1.82±0.24 a; X	3.23±0.33 b; XY	3.32±0.72 ab; XY
FSFNBi	Ecosystem (E)	64	3	**7.766**	**p<0.001**	Winter	0.77±0.11 a; X	0.72±0.14 a; X	0.98±0.17 a; X	0.74±0.12 a; X
	Season (S)	8	3	0.983	p=0.3998	Pre-monsoon	0.63±0.09 a; X	0.85±0.12 a; X	0.71±0.10 a; X	0.87±0.14 a; X
	E x S	57	9	**2.304**	**p<0.05**	**Monsoon**	0.67±0.11 a; X	0.82±0.13 ab; X	1.23±0.19 b; XY	0.98±0.14 ab; X
	Residuals	5588	2034			**Post monsoon**	0.49±0.13 a; X	0.52±0.16 a; X	1.53 ±0.22 b; Y	1.08±0.32 ab; X
PEBi	Ecosystem (E)	84	3	**12.487**	**p<0.001**	**Winter**	0.17±0.04 a; X	0.31±0.08 a; X	0.76±0.22 b; X	0.57±0.14 ab; X
	Season (S)	5	3	0.783	p=0.503	**Pre-monsoon**	0.05±0.02 a; X	0.41±0.13 ab; X	0.69±0.21 b; X	0.32±0.10 ab; X
	E x S	17	9	0.849	p=0.571	Monsoon	0.18±0.06 a; X	0.18±0.05 a; X	0.58±0.14 a; X	0.66±0.15 a; X
	Residuals	4569	2034			Post monsoon	0.13±0.05 a; X	0.18±0.09 a; X	0.47±0.16 a; X	0.70±0.29 a; X

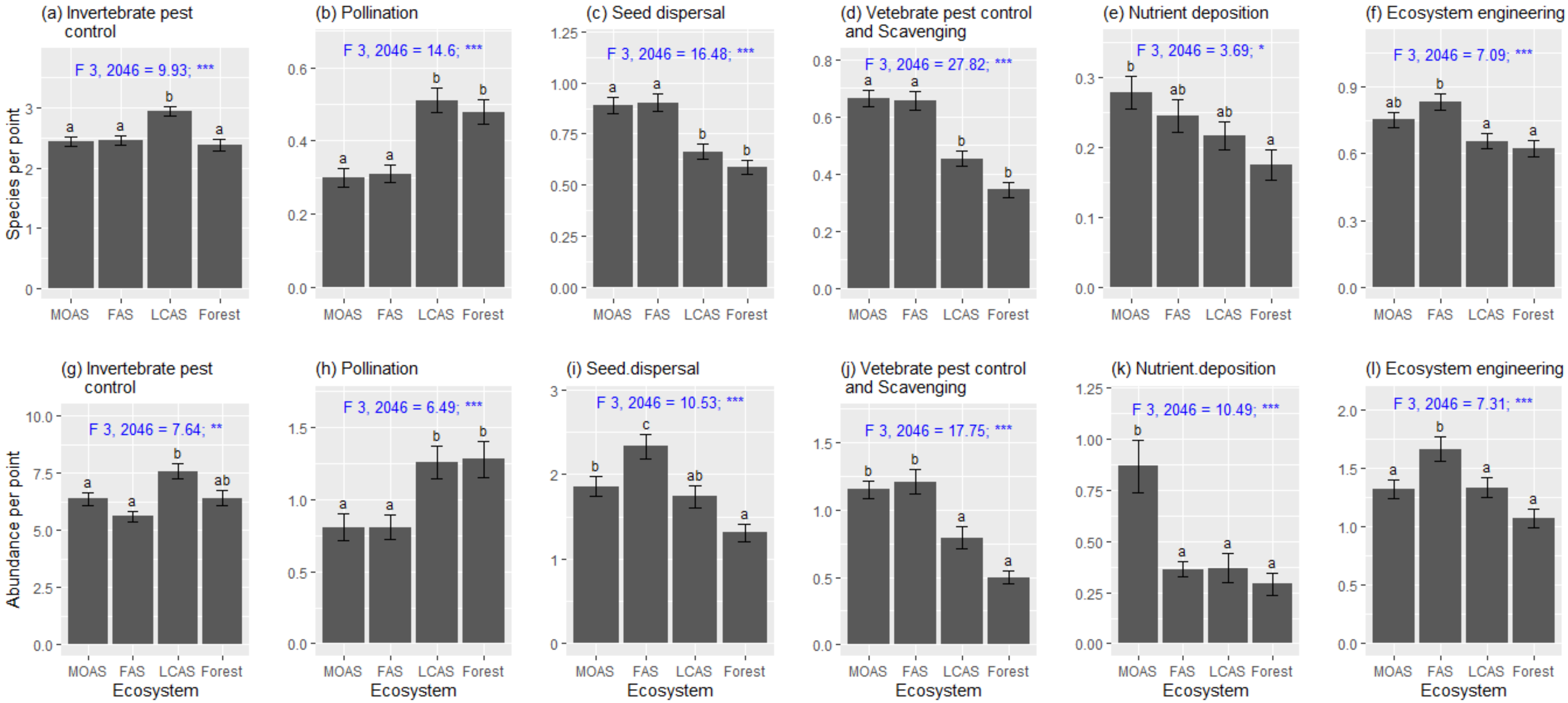


Fig. 4.5. Species richness per point and abundance per point of birds for different avian-mediated ecosystem services along agroecosystem-forest gradient of Sikkim, Eastern Himalaya. Results of ANOVA test including degree of freedom (3: between group; 2046: within group resulting from Ecosystems and total point counts, respectively),and significance level (***: p<0.001; **: p<0.01; *: p<0.05) are also shown. For Bonferroni corrected post-hoc, pair-wise comparisons, bars not sharing letters are significantly different at the alpha level of P = 0.05.

Total bird community (in all ecosystems) and the land use sensitive guilds: FSUIBi (in MOAS and Forest), FSMSIBi (in FAS, Forest), FSCIBi (in MOAS, LCAS and FAS), FSFNBi (in LCAS and Forest), and PEBi (in Forest) largely tracked the seasons. These four sub-groups (in select ecosystems), like the total birds showed increase in species per point and/or abundance per point from the onset of breeding to dispersal seasons, followed by slight decline or remained similar in winter (relative to dispersal season). Total birds were largely more speciose and abundant in LCAS and/or Forest during monsoon and/or post monsoon but in MOAS and FAS during winter and/or pre monsoon. Along the agroecosystem-forest gradient, species richness and/or abundance per point of FSUIBi (in winter), FSMSIBi (in monsoon), FSCIBi (in all seasons), and PEBi (during breeding, dispersal and winter) were higher in LCAS and/or Forest. However, there were instances of higher or comparable values (than Forests and/or LCAS) in MOAS and FAS for FSUIBi (during winter and post monsoon), and for FSMSIBi (during breeding and dispersal season) and FSFNBi (during winter and pre monsoon).

For the birds providing different avian-mediated ES, patterns of alpha diversity (Fig. 4.5a-f) and abundance per point (Fig. 4.5g-l) differed markedly along agroecosystem-forest gradients, with pollination and invertebrate pest control increasing (higher values in Forest and/or LCAS), but seed dispersal, waste disposal, nutrient deposition, ecosystem engineering declining (higher values in MOAS and/or FAS). For invertebrate pest control, LCAS had significantly higher species per point (Fig. 4.5a) compared to MOAS, FAS and Forest, as well as abundance per point (Fig. 4.5g) than MOAS and FAS (all $0.001<p<0.05$: Bonferroni corrected post-hoc tests) but was significantly similar between LCAS and Forest. For pollination, species per point (Fig. 4.5b) and abundance per point (Fig. 4.5h) were significantly higher in

LCAS and Forest than MOAS and FAS (all $0.001<p<0.05$). By contrast, for seed dispersal, I observed significantly higher alpha diversity (Fig. 4.5c) in FAS and MOAS compared with LCAS and Forest, whereas, abundance per point (Fig. 4.5i) in FAS compared with Forest and MOAS, as well as in MOAS than Forest (all $0.001<p<0.05$). For waste disposal, alpha diversity (Fig. 4.5d) as well as abundance per point (Fig. 4.5j) were significantly higher in MOAS and FAS than Forest and LCAS (all $0.001<p<0.05$). For nutrient deposition, MOAS had significantly higher alpha diversity (Fig. 4.5e) than Forest, and also abundance per point (Fig. 4.5k) than Forest and other IFS (all $0.001<p<0.05$). For ecosystem engineering, FAS had significantly higher alpha diversity (Fig. 4.5f) than Forest and LCAS, and also abundance per point (Fig. 4.5l) than Forest, LCAS and MOAS (all $0.001<p<0.05$).

4.3.4. Determinants of bird alpha diversity

For the total bird community, species per point and abundance per point lacked significant correlation with any of the habitat and environmental variables. However, H′ correlated significant positively ($p<0.05$) with elevation but negatively with MAT, whereas, total abundance showed significant negative relationship with tree basal area and density (Fig. 4.6). The different land use sensitive sub-groups (based on species per point, abundance per point and/or Shannon-Wiener diversity/ species richness) showed clear and significant (positive/negative) relationships with various variables (Fig. 4.6). The pcc showed significant positive relationship for FSMSIBi, but negative relationship for PEBi. The tree basal area correlated significant positively for FSFNBi and FSMSIBi. There was significant negative effect of tree species richness on FSMSIBi and of tree density on FSUIBi. The elevation correlated significant positively for FSCIBi, FSFNBi but negatively for FSMSIBi, PEBi. The MAT correlated significant negatively for FSCIBi and FSFNBi but positively for FSMSIBi and PEBi. The correlation with MAP was significant negative for FSUIBi, FSFNBi and FSCIBi, whereas, positive for FSMSIBi and PEBi (Fig. 4.6).

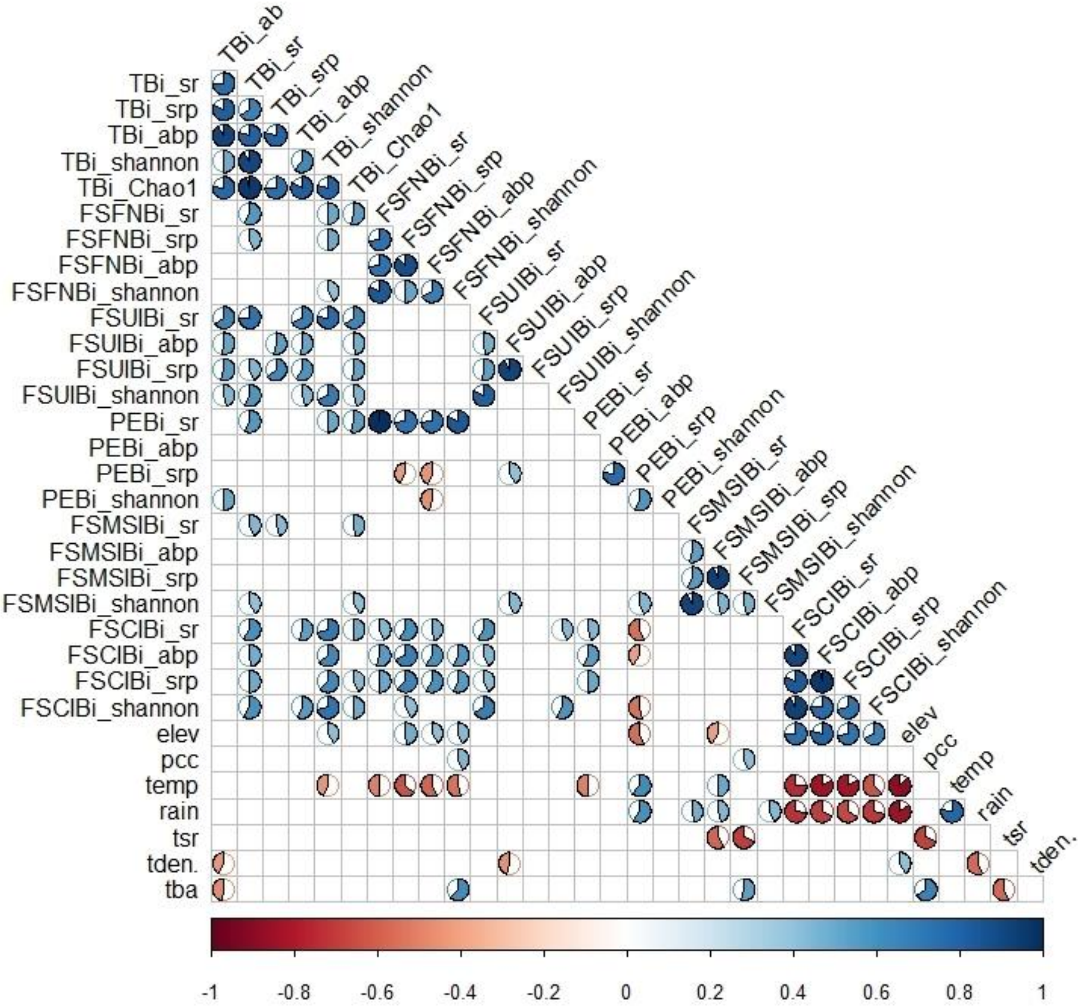


Fig. 4.6. Correlation matrix showing significant ($p<0.05$) and strong ($r>|0.45|$) correlation coefficients between the bird diversity measures and predictor variables along agroecosystem-forest gradient of Sikkim, Eastern Himalaya. The size and colour (-1: dark red to +1: dark blue) of the pie in the matrix code for correlation strength and direction, respectively, whereas, empty cells denotes lack of significant correlations. The habitat and environmental variables are: elevation (elev), mean annual precipitation (rain), mean annual temperature (temp), percentage canopy cover (pcc), tree species richness (tsr), tree density (tden.), and tree basal area (tba); the bird diversity parameters has two parts separated by underscore, first part in upper-case letters represents the bird guilds: total birds (TBi), forest specialist understorey insectivores (FSUIBi), forest specialist midstorey insectivores (FSMSIBi), forest specialist canopy insectivores (FSCIBi), forest specialist frugivores and nectarivores (FSFNBi), and protected and endemic birds (PEBi); and second part represents their diversity parameters, species richness per point (srp), abundance per point (abp), species richness (sr), abundance (ab), Shannon-Wiener diversity (shannon).

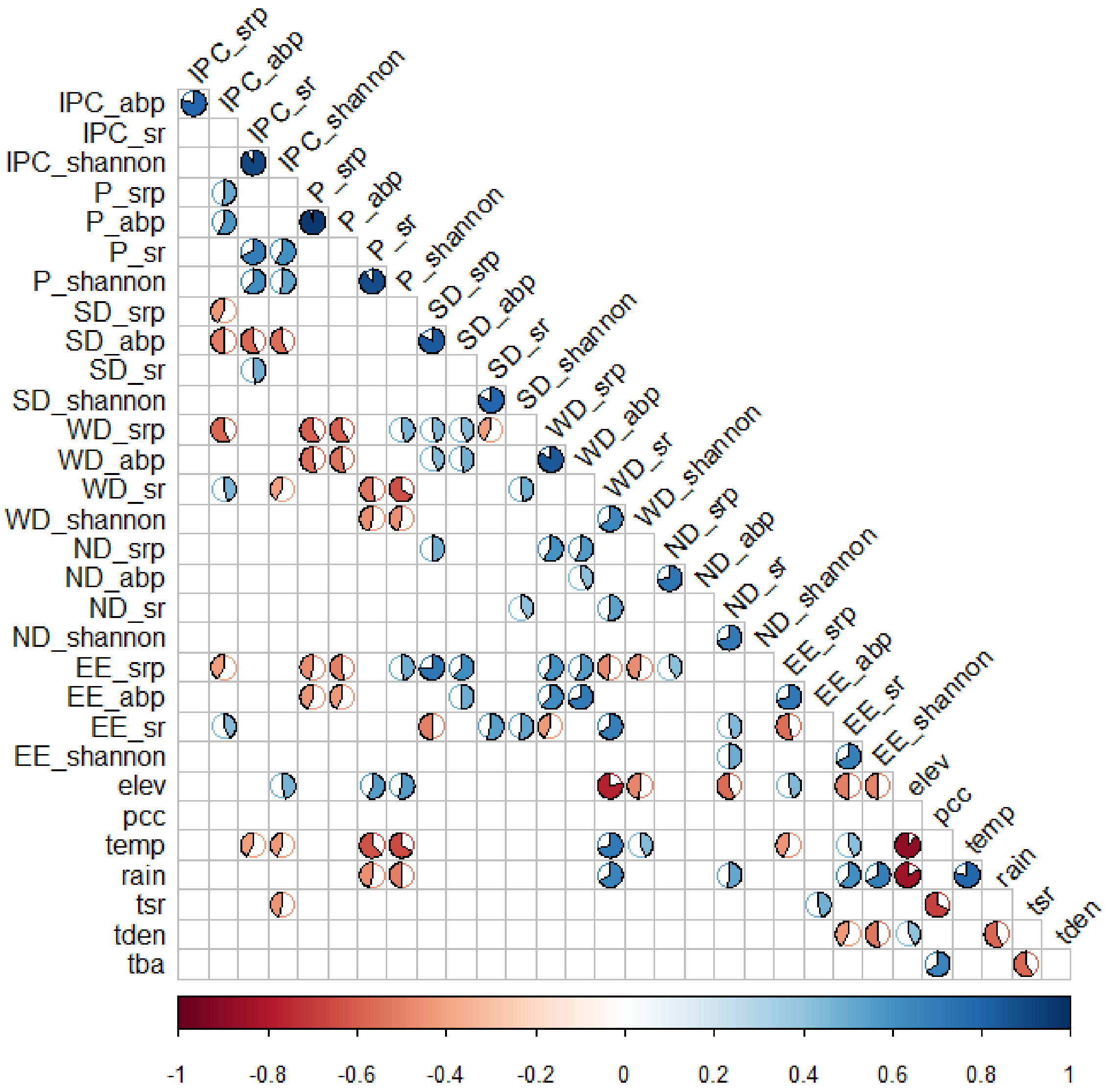


Fig. 4.7. Correlation matrix showing significant (p<0.05) and strong (r>|0.40|) correlation coefficients between the bird diversity measures for different avian-mediated ecosystem service providers and predictor variables along agroecosystem-forest gradient of Sikkim, Eastern Himalaya. The size and colour (-1: dark red to +1: dark blue) of the pie in the matrix code for correlation strength and direction, respectively, whereas, empty cells denotes lack of significant correlations. The habitat and environmental variables are: elevation (elev), mean annual precipitation (rain), mean annual temperature (temp), percentage canopy cover (pcc), tree species richness (tsr), tree density (tden.), and tree basal area (tba); the bird diversity parameters has two parts separated by underscore, first part in upper-case letters represents the avian-mediated ecosystem services: pollination (P), seed dispersal (SD), insect pest control (IPC), vertebrate pest control & scavenging (WD), nutrient deposition (ND), ecosystem engineering (EE); and second part represents their diversity parameters: species richness per point (srp), abundance per point (abp), species richness (sr), Shannon-Wiener diversity (shannon).

Similarly, for atleast one of the avian-mediated ES providers, H′ and/or species richness showed significant correlations with all variables except tree basal area and percentage canopy cover (Fig. 4.7). There was significant negative relationship for insect pest control with tree species richness, and for ecosystem engineering service providers with tree density. The pollination and insect pest control service providers correlated significant positively with elevation but negatively with MAP and/or MAT. By contrast, vertebrate pest control and scavenging, nutrient deposition, and/or ecosystem engineering service providers correlated significant negatively with elevation but positively with MAT and/or MAP (Fig. 4.7).

4.3.5. Patterns and determinants of bird beta diversity

Eighty-nine species (40.3%, 16901 individuals) were common to four ecosystems, whereas, 61 species (27.6%, 284 individuals) were observed exclusively in single ecosystem type, nine in Forests vs. 52 species in different IFS types. For total birds, pair-wise beta diversity based on abundance-based dissimilarity (βbray) was significantly ($F_{3,\ 20}$=9.72, p<0.001) highest in Forests (βbray= 0.3774), followed by FAS which declined towards MOAS (βbray= 0.2347) through LCAS. Tukey's HSD post-hoc tests further showed that βbray of total birds was significantly higher in Forest than MOAS (p<0.001) and LCAS (p<0.05); and also in FAS than MOAS (p<0.05) (Fig. 4.8a). Similarly, the pair-wise beta diversity based on incidence-based dissimilarity ($F_{3,\ 20}$=6.04, p<0.01) was significantly highest in Forests (βsor=0.2998), which declined towards MOAS (βsor=0.2015) through FAS and LCAS. Tukey's HSD post-hoc test further revealed significantly higher βsor of total birds in Forest than MOAS (p<0.01) and LCAS (p<0.05) (Fig. 4.8b). I found very high overall multiple-site beta diversity (β_{BRAY} = 0.881; β_{SOR} = 0.845) dominantly contributed by the substitution components ($\beta_{BRAY.BAL}$ = 0.857; β_{SIM}= 0.808) than nestedness components ($\beta_{BRAY.GRA}$= 0.024; β_{SIM} = 0.037) for both dissimilarity indices.

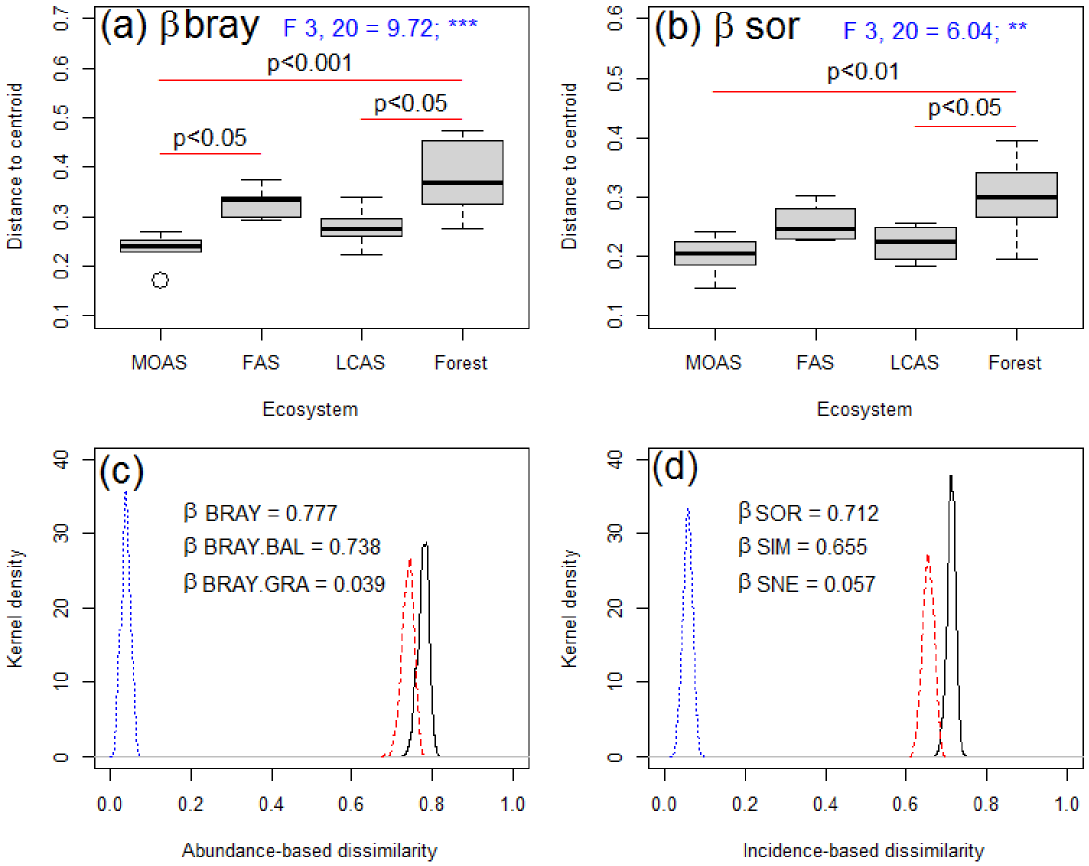


Fig. 4.8. Pair-wise and multiple-site beta diversity for total bird communities along agroecosystem-forest gradient of Sikkim, Eastern Himalaya. The pair-wise beta diversity, (a) βbray and (b) βsor are shown by distance to centroid; and multiple-site beta diversity (computed by resampling across 1000 samples of 10 transects in different ecosystems) shown by kernel density for (c) Bray-Curtis index, solid line: β_{BRAY}, dashed line: $\beta_{BRAY.BAL}$, and dotted line: $\beta_{BRAY.GRA}$; and (d) Sorensen index, solid line: β_{SOR}; dashed line: β_{SIM}; and dotted line: β_{SNE}. MOAS- Mandarin orange-based agroforestry systems, FAS- farm-based agroforestry systems, LCAS- large cardamom-based agroforestry systems, and Forest- Natural forests. Results of ANOVA test including degree of freedom (3: between group, 20: within group resulting from Ecosystems and Transects, respectively), and significance level (**: $p<0.01$; *: $p<0.05$), and Tukey's HSD post-hoc pair-wise comparisons are also shown.

The contribution of substitution components to overall multiple-site beta diversity was 97.28%, and 95.62%, respectively in case of abundance-basedand incidence-based dissimilarity indices. Simulations of multiple-site beta diversity and their components based on the two multiple-site dissimilarity indices resulted in slight decline in overall ($\beta_{BRAY} = 0.778$; $\beta_{SOR} = 0.712$), substitution components ($\beta_{BRAY.BAL} = 0.739$; $\beta_{SIM}= 0.655$) but increase in nestedness components ($\beta_{BRAY.GRA}= 0.039$; $\beta_{SNE} =$

0.057) when compared to original values. The contribution of nestedness components to overall multiple-site beta diversity slightly increased after simulation based on Bray-Curtis (2.72% to 5.01%), and Sorensen (4.38% to 8.01%) dissimilarity indices. However, simulated multiple-site beta diversity was still very highly dominated by substitution components, in case of both abundance-based (Fig. 4.8c) and incidence-based (Fig. 4.8d) dissimilarity indices, contributing 94.99%, and 91.99% respectively to overall multiple-site beta diversity.

For the different land use sensitive guilds, patterns of abundance-based pair-wise beta diversity (βbray) differed markedly along agroecosystem-forest gradient of Sikkim. One-way ANOVA followed by Tukey's HSD post-hoc, pair-wise comparisons showed that βbray of FSUIBi did not differ between MOAS and LCAS, but was significantly higher in Forest and FAS (Fig. 4.9a). For FSMSIBi, βbray did not differ between FAS and LCAS, but was marginal significantly ($0.05<p<0.1$) higher in Forest and MOAS (Fig. 4.9b). FSCIBi showed significant difference in βbray for across the ecosystems ($F_{3,20}=4.02$, $p<0.05$; Fig. 4.9c), with Forest and FAS having near significantly higher mean values compared to MOAS ($0.05<p<0.1$). For FSFNBi, βbray significantly varied across the ecosystems ($F_{3,20}=4.02$, $p<0.05$; Fig. 4.9c), with mean value in FAS being near significantly higher than in LCAS ($0.05<p<0.1$), but its mean value did not differ from MOAS and Forest. For PEBi, βbray also had near significant ($F_{3,20}=3.09$, $p=0.051$) difference among the ecosystems, with Forest having higher mean value compared to LCAS and MOAS ($0.05<p<0.1$), but did not differ from FAS (Fig. 4.9e). The βbray values were higher in Forest (followed by FAS) for forest specialist understorey insectivores, protected and endemic species, forest specialist midstorey insectivores, or in FAS (followed by

Forest) for forest specialist canopy insectivores, forest specialist frugivores & nectarivores birds, all of which declined in MOAS and LCAS.

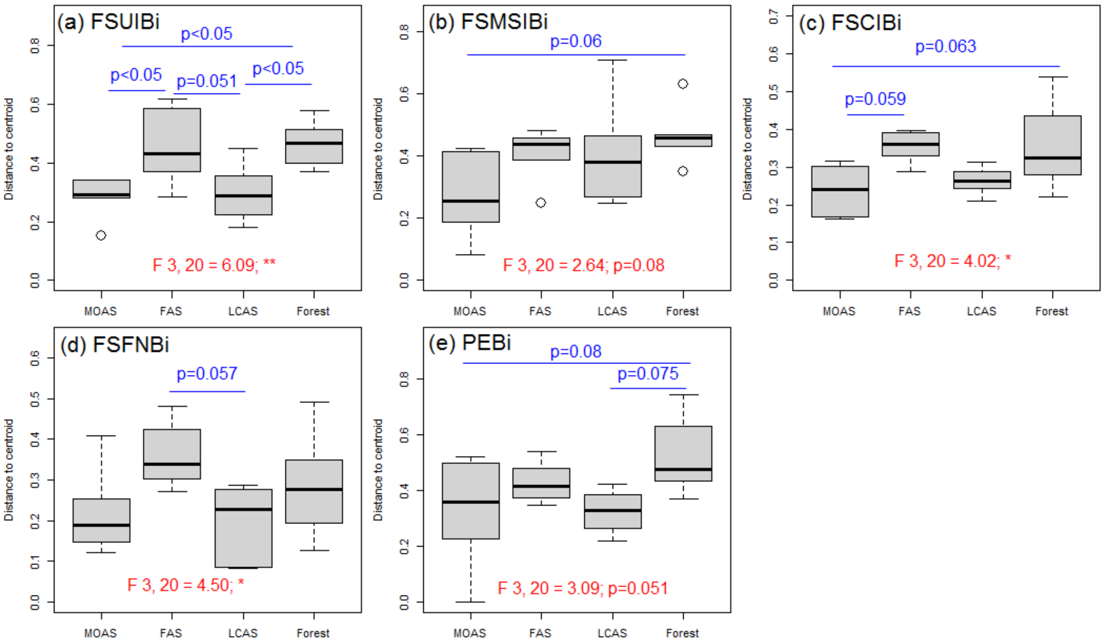


Fig. 4.9. Pair-wise beta diversity shown by distance to centroid for different land use sensitive guilds of bird communities along agroecosystem-forest gradient of Sikkim, Eastern Himalaya. The pair-wise beta diversity (βbray) was quantified following abundance-based Bray-Curtis dissimilarity index. Mandarin orange-based agroforestry systems (MOAS), farm-based agroforestry systems (FAS), large cardamom-based agroforestry systems (LCAS), and Natural forests (Forest). Results of one-way ANOVA test including degree of freedom (3: between group, 20: within group resulting from Ecosystems and Transects, respectively), and significance level (**: p<0.01; *: p<0.05; n.s.: non-significant), as well as Tukey's HSD post-hoc pair-wise comparisons are also provided.

Likewise for different avian-mediated ES providers, pair-wise beta diversity (βbray) for Invertebrate pest control (Fig. 4.10a), Pollination (Fig. 4.10b), Seed dispersal (Fig. 4.10c), and Vertebrate pest control and Scavenging (Fig. 4.10d) was highest in Forests (followed by FAS) that declined in MOAS and LCAS, however, the difference was significant only in case of Invertebrate pest control ($F_{3,\ 20} = 8.55$; p<0.001). Tukey's HSD post-hoc test further showed that βbray for Invertebrate pest control providers was significantly higher in Forest compared to MOAS (p<0.001) and LCAS (p<0.01), as well as in FAS than MOAS (p<0.05).

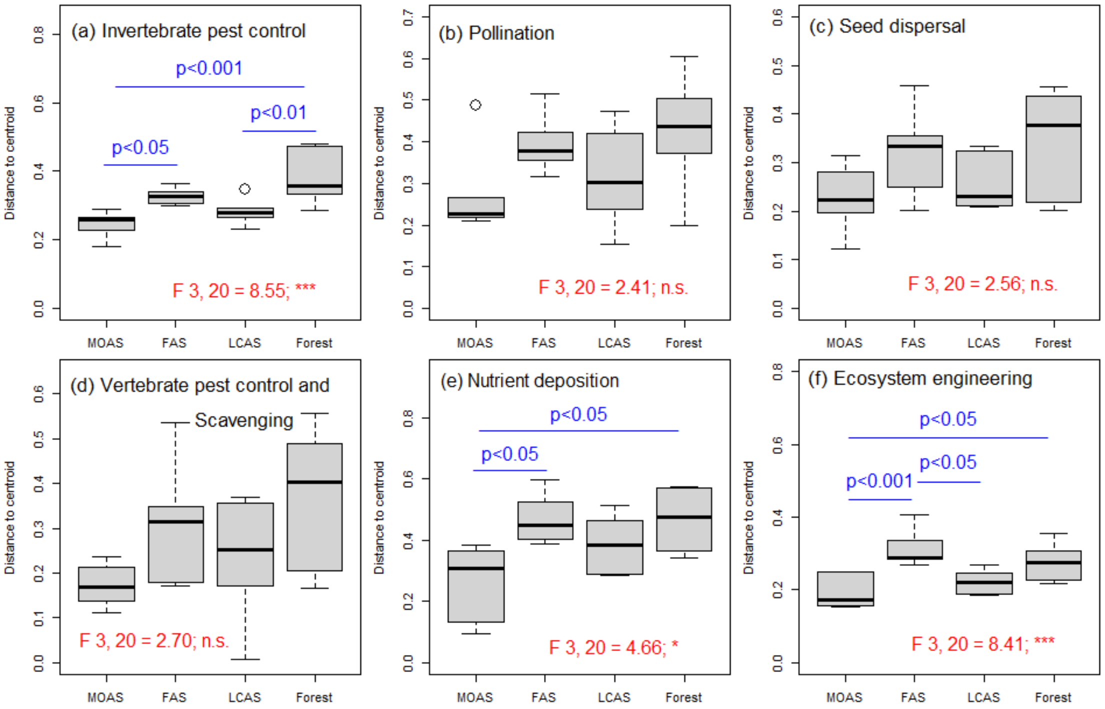


Fig. 4.10. Pair-wise beta diversity shown by distance to centroid for the different avian-mediated ecosystem service providers along agroecosystem-forest gradient of Sikkim, Eastern Himalaya. I quantified pair-wise beta diversity (βbray) following abundance-based Bray-Curtis dissimilarity index. Mandarin orange-based agroforestry systems (MOAS), farm-based agroforestry systems (FAS), large cardamom-based agroforestry systems (LCAS), and Natural forests (Forest). Results of one-way ANOVA test including degree of freedom (3: between group, 20: within group resulting from Ecosystems and Transects, respectively), and significance level (***: p<0.001; *: p<0.05; n.s.: non-significant), and Tukey's HSD post-hoc pair-wise comparisons are also provided.

On the other hand, the βbray was significantly highest in FAS (followed by Forests) for Nutrient deposition ($F_{3, 20}$ = 4.66; p<0.05; Fig. 4.10e) and Ecosystem engineering ($F_{3, 20}$ = 8.41; p<0.001; Fig. 4.10f), which also declined in MOAS and LCAS. Based on Tukey's post-hoc test, there was significant difference in βbray for birds providing Nutrient deposition (Fig. 4.10e) between FAS and MOAS (p<0.05), and between Forest and MOAS (p<0.05), and for Ecosystem Engineering (Fig. 4.10f) in case of FAS vs. MOAS, Forest vs. MOAS, and LCAS vs. FAS (0.001<p<0.05).

In the present study, I observed dominant role of substitution component ($\beta_{\mathrm{BRAY.BAL}}$) in determining the overall multiple-site beta diversity (β_{BRAY}) for 4 out of

6 avian-mediated ES providers. For both original and simulated multiple-site dissimilarity indices, the contribution of $\beta_{BRAY.BAL}$ to β_{BRAY} ranged from very high for Invertebrate pest control (R1: original = 96.25%; simulated = 93.19%) to high for Pollination, Seed dispersal, Ecosystem engineering (R1: original = >90%; simulated = >82%). In contrast, I observed comparatively more important role of nestedness component ($\beta_{BRAY.GRA}$) in structuring β_{BRAY} for birds responsible for Waste disposal (R2: original = 14.27%; simulated = 22.78%) and Nutrient deposition (R2: original = 23.94%; simulated = 39.07%), for both original and simulated multiple-site Bray-Curtis dissimilarity indices, however R1 was still >60 for them (Table 4.6).

Table 4.6. Multiple-site beta diversity for different avian-mediated ecosystem services (ES) providers along the agroecosystems-forest gradient of Sikkim, Eastern Himalaya. Overall multiple-site beta diversity ($\beta_{BRAY.BAL}$), and components of substitution ($\beta_{BRAY.BAL}$) and Nestedness ($\beta_{BRAY.GRA}$) were quantified following abundance-based Bray-Curtis dissimilarity index. Original followed by simulated (±S.D.) values are given for each types of ES providers. Simulations were run by resampling across 1000 samples of 10 transects in different ecosystems.

ES	**Beta diversity**	$\beta_{BRAY.BAL}$	**±S.D.**	$\beta_{BRAY.GRA}$	**±S.D.**	β_{BRAY}	**±S.D.**	**R1 (%)**	**R2 (%)**
Invertebrate pest control	Original	**0.848**	-	0.033	-	**0.881**	-	96.25	3.75
	Simulated	0.725	±0.015	**0.053**	±0.010	0.778	±0.011	93.19	6.81
Pollination	Original	**0.802**	-	0.087	-	**0.888**	-	90.32	9.80
	Simulated	0.654	±0.041	**0.137**	±0.030	0.791	±0.023	82.68	17.32
Seed dispersal	Original	**0.822**	-	0.060	-	**0.882**	-	93.20	6.80
	Simulated	0.684	±0.028	**0.095**	±0.021	0.779	±0.019	87.80	12.20
Vertebrate pest control and scavenging	Original	**0.662**	-	0.208	-	**0.869**	-	76.18	23.94
	Simulated	0.457	±0.090	**0.293**	±0.069	0.750	±0.040	60.93	39.07
Nutrient deposition	Original	**0.782**	-	0.130	-	**0.911**	-	85.84	14.27
	Simulated	0.652	±0.062	**0.192**	±0.055	0.843	±0.019	77.34	22.78
Ecosystem engineering	Original	**0.791**	-	0.067	-	**0.858**	-	92.19	7.81
	Simulated	0.629	±0.033	**0.103**	±0.026	0.732	±0.025	85.93	14.07

R1: ratio of $\beta_{BRAY.BAL}$ to β_{BRAY}; R2: ratio of $\beta_{BRAY.GRA}$ to $\beta_{BRAY;}$ S.D.: standard deviation.

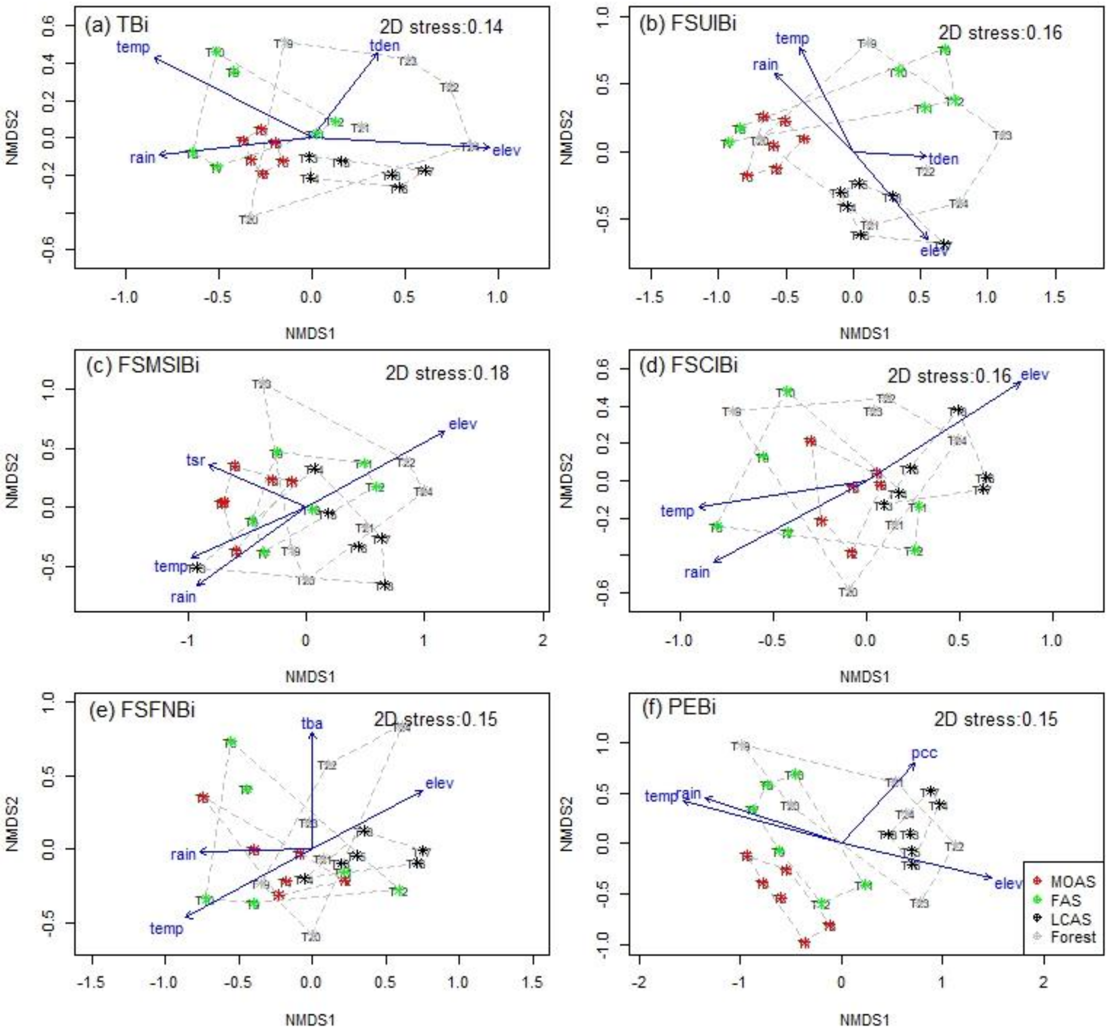


Fig. 4.11. NMDS of bird community composition for total species (a) and different land use sensitive guilds (b-f) along the agroecosystem-forest gradient of Sikkim, Eastern Himalaya. Environmental parameters (vectors as arrow) are overlaid on sites with convex hull volume. Information for ordination statistics are given in Appendices C-D. TBi- total birds, FSUIBi- forest specialist understorey insectivores, FSMSIBi- forest specialist midstorey insectivores, FSCIBi- forest specialist canopy insectivores, FSFNBi- forest specialist frugivores and nectarivores, PEBi- protected and endemic birds.

Based on NMDS, community composition were significantly determined by ecosystems for total birds (Fig. 4.11a), and three land use sensitive guilds: forest specialist insectivores specialized to understorey (4.11b), midstorey (Fig. 4.11c) and canopy (Fig. 4.11d), but not for forest specialist frugivores & nectarivores (Fig. 4.11e), and protected & endemic species (Fig. 4.11f).

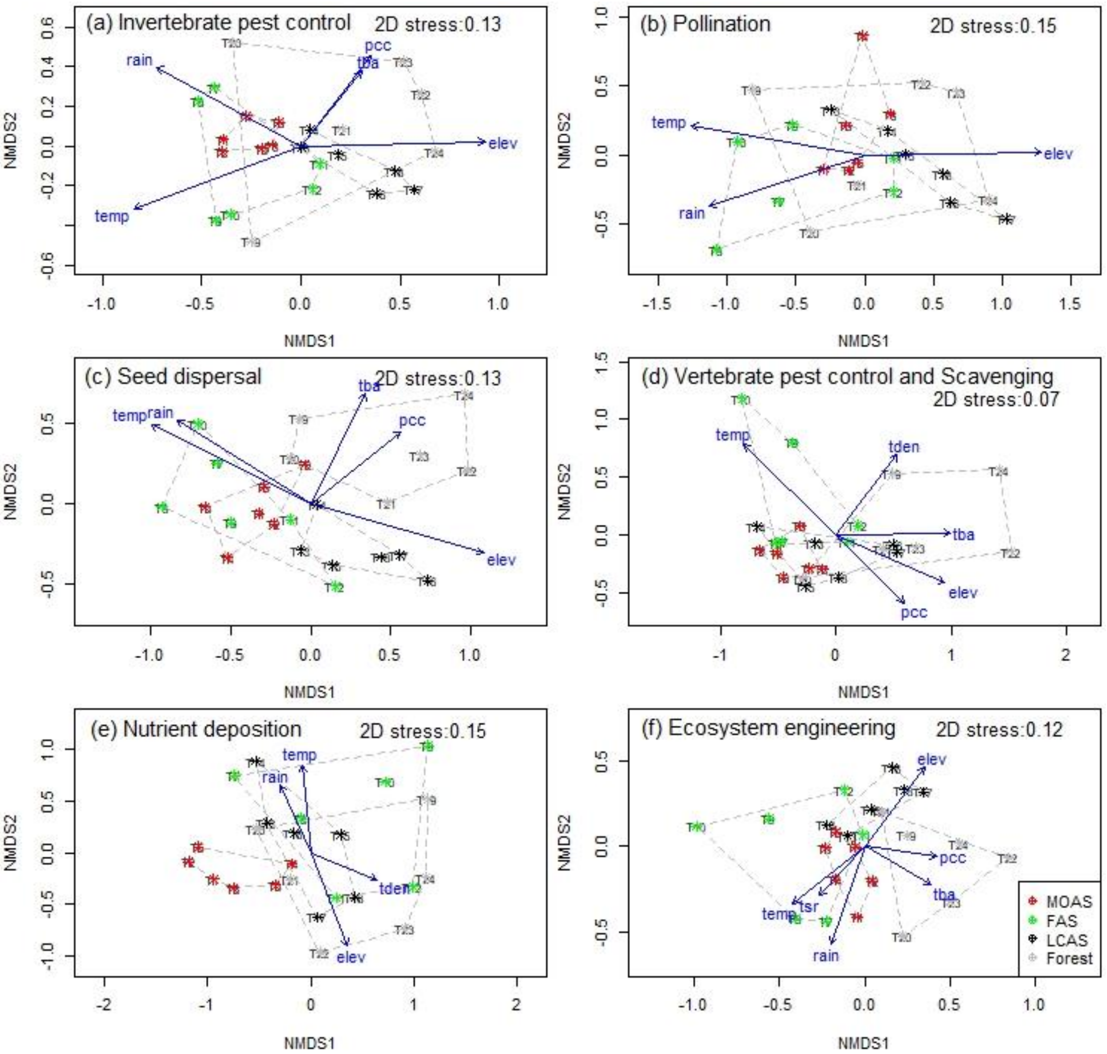


Fig. 4.12. NMDS of bird community composition classified based on avian-mediated ecosystem services along agroecosystem-forest gradient of Sikkim, Eastern Himalaya. Significant environmental parameters (vectors as arrow) are overlaid on sites with convex hull volume. Further information for ordination statistics are given in Appendices D-E.

Similar results were obtained when bird communities were analysed based on different avian-mediated ES. The bird community assemblages were significantly determined by ecosystems for all six ES providers (Fig. 4.12a-f). Forest and LCAS transects were characterized by higher elevation, percentage canopy cover, tree basal area and density and but lower MAT, MAP and tree species richness, In contrast, MOAS and FAS transects were characterized by higher MAT, MAP and tree species richness, but lower elevation, percentage canopy cover, tree basal area and density (Fig. 4.11-4.12) (also see Appendices C-E).

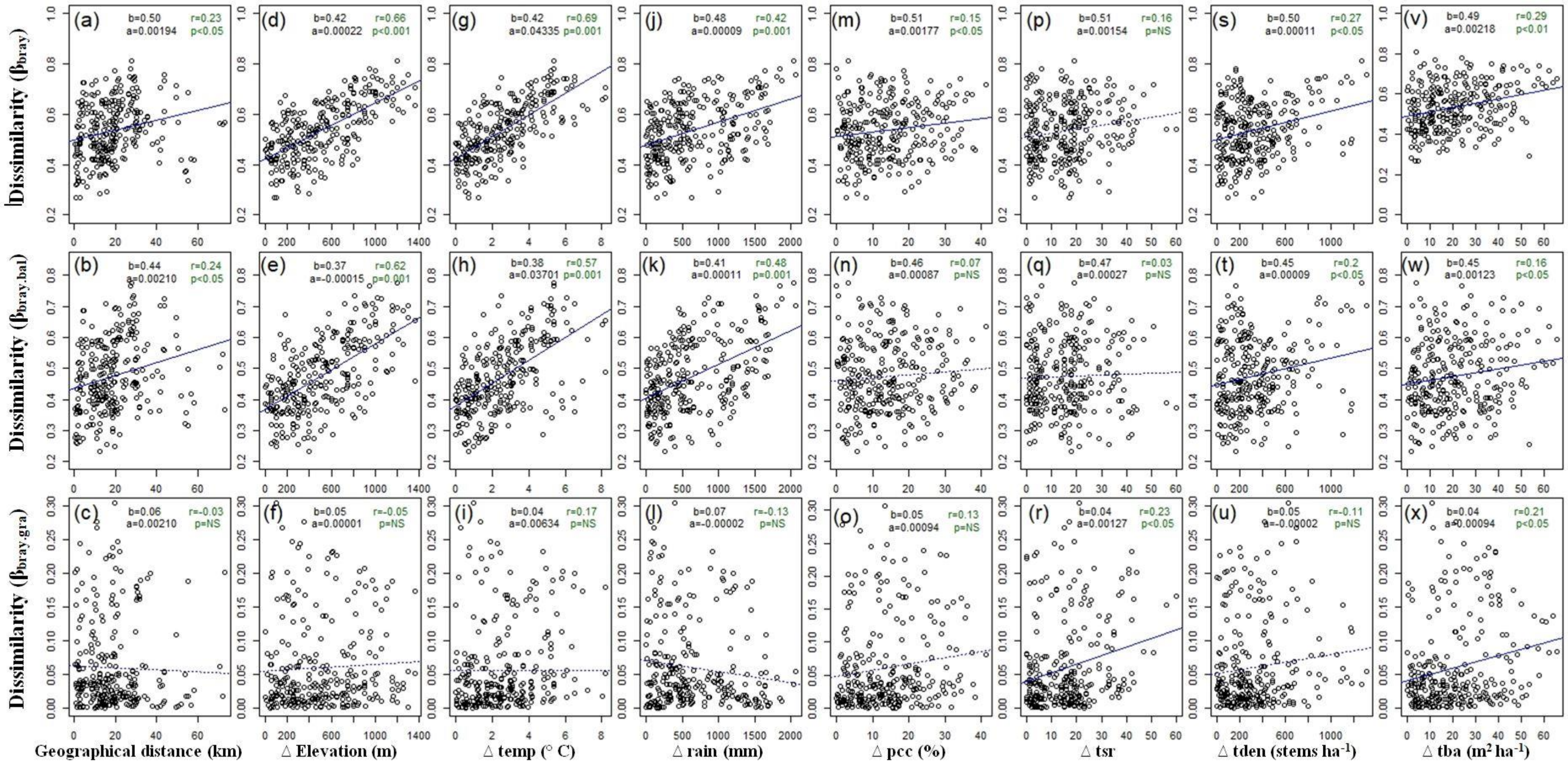


Fig. 4.13. Relationship of overall pair-wise beta diversity (βbray) and its components of substitution (βbray.bal) and nestedness (βbray.gra) for total bird community with the different environmental variables along agroecosystem-forest gradient of Sikkim, Eastern Himalaya: geographical distance (a-c), Δ Elevation (d-f), Δ mean annual temperature (Δ temp:g-i), Δmean annual precipitation (Δrain:j-l), Δpercentage canopy cover (Δpcc: m-o), Δtree species richness (Δtsr: p-r), Δtree density (Δtden: s-u), Δtree basal area (Δtba:v-x). Slopes (*a*) and intercepts (*b*) from MRM, and p-value (*p*) and Pearson correlation coefficient (*r*) of partial mantel tests are given.

Based on distance decay relationship, overall pair-wise beta diversity and its components (mainly substitution component) showed significant relationship with habitat and environmental variable difference/variation (Δ) among the pair of transects (Fig. 4.13). Due to the additive partitioning method, the regression slopes and intercepts of βbray was equal to the sum of βbray.bal and βbray.gra, and likewise those of βsor equalled the sum of βsim and βsne (not presented since they gave qualitatively similar results) (Baselga, 2013a; Si *et al.*, 2015). For total bird communities, overall pair-wise beta diversity (βbray) and its substitution component (βbray.bal) increased significantly with geographical distance between transects (Fig. 4.13a,b), ΔElevation (Fig. 4.13d,e), ΔMAT (Fig. 4.13g,h), ΔMAP (Fig. 4.13j,k), Δtree density (Fig. 4.13p,q), and Δtree basal area (Fig. 4.13s,t). Additionally, there was a significant positive relationship of βbray with Δpcc (Fig. 4.13m), and of nestedness component (βbray.gra) with only Δtree species richness (Fig. 4.13r).

4.4. Discussion

4.4.1. Bird species richness and diversity

In the present study, total species richness and Shannon-Wiener diversity for both the total birds as well as the three land use sensitive guilds were higher (in LCAS) or comparable (in FAS and MOAS) to Forests. The present observation of high bird species richness (Fig. 4.14a), Shannon-Wiener diversity (Fig. 4.14b), and proportion of total bird species of the landscape retained (Fig. 4.14c) in agroecosystems or forest is comparable to or greater than the previous studies from the biodiversity hotspots of the Himalaya (e.g., Chettri *et al.*, 2001; Acharya *et al.*, 2010; Elsen *et al.*, 2018), Indo-Burma(e.g., Mandal and Raman, 2016), Western Ghats and Sri Lanka (e.g., Daniels *et al.*, 1990; Kunte *et al.*, 1999; Karanth *et al.*, 2016), and elsewhere (e.g., Daily *et al.*, 2001; Mas and Dietsch, 2004; Buechley *et al.*, 2015; Prabowo *et al.*, 2016; Yabuhara *et al.*, 2019) (Fig.4.14; Appendix F).

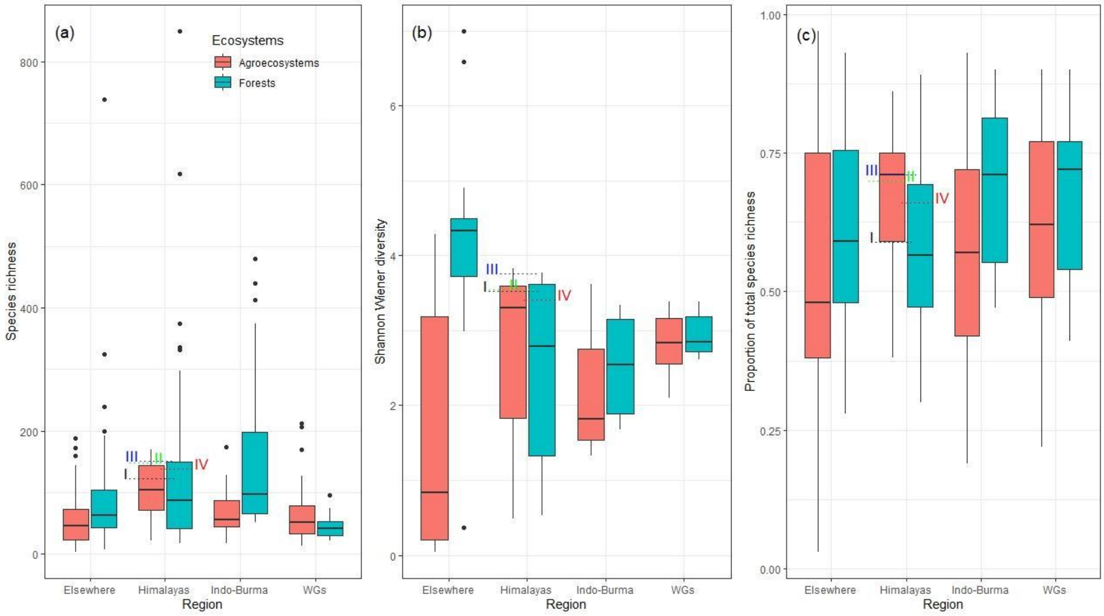


Fig. 4.14. Comparison of total species richness (a), Shannon-Wiener diversity (b), proportion of total species richness(c) of birds in different ecosystems (I: MOAS; II: FAS; III: LCAS; IV: Forest) of Sikkim, Eastern Himalaya, India with those reported from the biodiversity hotspots of Himalaya, Indo-Burma, Western Ghats and Sri Lanka (WGs) and Elsewhere.

Results of the meta- analysis of past studies on birds in the Himalaya, Indo-Burma, Western Ghats and Sri Lanka, and Elsewhere (including two-way ANOVA, and Bonferroni corrected post hoc, pair-wise comparisons) are summarised in Table 4.7 and Fig. 4.14. The agroecosystems and other human-modified ecosystems in the Himalaya (and Western Ghats), retained comparatively higher median bird species richness (Fig. 4.14a) than in Forests and PAs, whereas, opposite trend was found for the Indo-Burma and Elsewhere. However based on Bonferroni corrected post-hoc test, average total species richness of birds was significantly higher in the Forests than agroecosystems (all $p<0.05$) in Elsewhere, Indo-Burma, the Himalaya (Table 4.7). The total species richness of birds for Agroecosystems and human-modified ecosystems was significantly higher in the Himalaya (by ~2 times) compared to elsewhere ($p<0.05$), however, for Forests and PAs, values were significantly higher (all $p<0.05$) in the Himalaya (by up to ~2.4 times) and Indo-Burma (by upto ~1.8 times) than Western Ghats & Sri Lanka, and elsewhere (Table 4.7; Fig. 4.14a). Total bird species richness observed in the three IFS (MOAS:132; FAS:157; LCAS:160 species) of Sikkim was comparatively higher than the mean values reported in agroecosystems from all the regions, which ranged from upto ~1.5 and ~2.4 times more than Himalaya, Indo-Burma, respectively, and upto ~3 and 3.4 times respectively more than WGs, elsewhere. On the other hand, total bird species richness of Forest (147 species) in the present study was ~2.2 and ~1.7 times respectively, higher than mean values reported in Forests and PAs of WGs and elsewhere but was slightly (~10%) lower compared to Himalaya and Indo-Burma (Table 4.7).

Across ecosystems, the agroecosystems and other human-modified ecosystems had comparatively higher median (Fig. 4.14b) and mean (Table 4.7) values of Shannon-Wiener diversity when compared to Forests and PAs in the Himalaya, whereas, opposite was true for the Indo-Burma, WGs, and Elsewhere. However, the difference between ecosystems was significant only for elsewhere, which showed ~2.4 times higher average Shannon-Wiener diversity than agroecosystems ($p<0.05$: Bonferroni corrected post-hoc test). Across regions, Shannon-Wiener diversity in Forest ecosystems was significantly highest for elsewhere compared to the Himalaya and WGs biodiversity hotspots (all $p<0.05$: Bonferroni corrected post-hoc test), but was not different from Indo-Burma. In contrast, Shannon-Wiener diversity in agroecosystems did not significantly differ between the regions (Table 4.7). Shannon-Wiener diversity in the three IFS (MOAS: 3.6; FAS: 3.62; LCAS: 3.83) of Sikkim was upto ~1.4 and ~2.2 times higher than the mean values in agroecosystems reported from the Himalaya, Indo-Burma, WGs biodiversity hotspots, and elsewhere. On the other hand, Shannon-Wiener diversity of Forest (4.8) in the present study was ~1.2 (in WGs) and ~1.4 times higher than mean values reported in Forests and PAs from from the biodiversity hotspots (i.e., the Himalaya, Indo-Burma), but was lower (by ~20%) compared to elsewhere (Table 4.7).

Among ecosystems, comparatively higher median values for proportion of total bird species richness was retained in Agroecosystems than Forests and PAs in the Himalaya (Fig. 4.14c), contrasting with opposite patterns in other regions. However, the mean values were significantly different between ecosystems (all $p<0.05$: Bonferroni corrected post-hoc tests) only in the Indo-Burma and Elsewhere with higher value in Forest compared to agroecosystems. Across regions, proportion of total species richness of birds for Agroecosystems was significantly higher in the Himalaya compared to elsewhere ($p<0.05$: Bonferroni corrected post-hoc tests) but did not varied between Himalaya and other two biodiversity hotspots (all $p>0.05$). Highest proportion of total bird for Forest ecosystems was found in the Indo-Burma than all other regions, but the difference was not significant (Table 4.7). Proportion of total bird species richness in the three IFS (MOAS: 0.6; FAS: 0.71; LCAS: 0.72) of Sikkim was upto ~1.2 and ~1.3 times higher comprared to the mean values reported in agroecosystems from all regions except Himalaya. In contrast, Shannon-Wiener diversity of Forest (0.67) in the present study was ~1.1 (than elsewhere) to ~1.2 times (than Himalaya) higher compared to mean values reported in Forests and PAs from two regions, but was similar for Indo-Burma and WGs.

Table 4.7: Results of two-way ANOVA, summary statistics and Bonferroni corrected post-hoc, pair-wise comparisons of total species richness, Shannon-Wiener diversity and proportion of total species richness of birds in the biodiversity hotspots of the Himalaya (including from present study), Indo-Burma, Western Ghats and Sri Lanka (WGs) and Elsewhere. Mean ±SE values with different letters across regions (vertically: a, b) and ecosystems (horizontally: X, Y), and are significantly different at the alpha level of P = 0.05; Bonferroni corrected. Df: Degree of freedom; sum sq.: Sum of squares.

Diversity	**Two-way ANOVA test**						**Ecosystems**	
parameter	**Factor**	**df**	**Sum sq.**	**Estimate**	**P-value**	**Regions**	**Agroecosystems**	**Forests**
Total species richness	Regions (R)	3	255235	**12.38**	p<0.001	**Himalaya**	**104.5±17.67** b; X	**155.4±15.13** b; Y
	Ecosystems (E)	1	128046	**18.63**	p<0.001	**Indo Burma**	66.4±15.67 ab; X	**162.0±16.92** b; Y
	R:E	3	88856	**4.31**	p<0.01	Western Ghats & Sri Lanka	46.8±19.02 ab; X	67.5±13.63 a; X
	Residuals	366	2515088			**Elsewhere**	53.2±7.36 a; X	87.5 ±8.89 a; Y
						<u>Present study</u>: MOAS	132	147 (in Forest)
						FAS	157	
						LCAS	160	
Shannon Wiener diversity	Regions (R)	3	4.16	0.972	p=0.409	Himalaya	2.67±0.38 a; X	2.49±0.28 a; X
	Ecosystems (E)	1	20.96	**14.704**	p<0.001	Indo Burma	2.10±0.35 a; X	2.51± 0.42 a; X
	R:E	3	33.10	**7.740**	p<0.001	Western Ghats & Sri Lanka	**2.82±0.36** a; X	**2.95±0.45** ab; X
	Residuals	93	132.56			**Elsewhere**	1.72±0.26 a; X	**4.23**±0.32 b; Y
						<u>Present study</u>: MOAS	3.60	3.48 (in Forest)
						FAS	3.62	
						LCAS	3.83	
Proportion of total species richness	Regions (R)	3	0.243	2.089	p=0.1020	Himalaya	**0.67±0.05** b; X	0.58±0.05 a; X
	Ecosystems (E)	1	0.214	**5.504**	p<0.05	**Indo Burma**	0.55±0.04 ab; X	**0.69± 0.06** a; Y
	R:E	3	0.246	2.111	p=0.099	Western Ghats & Sri Lanka	0.62± 0.04 ab; X	0.67±0.06 a; X
	Residuals	267	10.357	**12.38**	p<0.001	**Elsewhere**	0.53±0.02 a; X	**0.61±0.02** a; Y
						<u>Present study</u>: MOAS	0.60	0.67 (in Forest)
						FAS	0.71	
						LCAS	0.72	

The bird’ species richness, Shannon-Wiener diversity and proportion of the total species richness in the IFS (specially in LCAS, FAS) and Forest observed in the present study were mostly above the median and mean values reported from the past studies for all three indices. These results signify the high potentiality of agroecosystems and adjoining forests (outside the PA network) of the Himalaya (Eastern Himalaya in particular) for sustaining high bird diversity, therefore their effectiveness in complementing the conservation efforts in PAs. Hence, conservation framework of land-sharing can complement the land-sparing strategies found effective in recent studies on birds and plants (Ghosh-Harihar *et al.*, 2019; Manish and Pandit, 2019) in the Himalaya that emphasized on establishment of more PAs and extension of boundaries of the existing PAs.

For the different migratory guilds, the overall bird communities were highly dominated by resident and altitudinal migrant (191species), that made up ~86% of total species-pool, whereas, the remaining ~14% pool was constituted by summer visitor and winter visitor/passage migrants (30 species), similar to the past reports in the Himalaya (Acharya *et al.*, 2010; Dahal *et al.*, 2014). Species richness of resident birds (in LCAS, FAS) and altitudinal migrants (in FAS), was comparable to Forest. However, IFS supported comparatively higher winter visitors & passage migrants (8 species in FAS vs. 2 species in Forest), summer visitors (13 species in LCAS and FAS vs. 10 species in Forest), altitudinal migrants (83 species in LCAS vs. 76 species in Forest). This signifies the potential of IFS (especially LCAS and FAS) in supporting different migrant birds, specifically during breeding (for SV), dispersal and winter (for AlM, WV and PM) seasons. In addition to resident birds, altitudinal migrants dominate the bird communities in the low- to mid-elevation area. These birds undertake seasonal migration across the elevation gradient linked to seasonal dynamics in climate and resource availability (i.e., food and suitable habitat), and take refuge in different human-modified ecosystems including agroecosystems (Acharya *et al.*, 2010; Acharya and Vijayan, 2011a; Dahal *et al.*, 2014; Grimmett *et al.*, 2019). Breeding range of only 28 altitudinal species extended above 2000 m, which breed either in the sub-alpine/ rhododendron shrubberies above the treeline (e.g., *Aethopyga ignicauda*, and *Phoenicurus frontalis*), or in the sub-alpine and temperate belts(e.g., *Zoothera dauma*, *Phylloscopus whistleri*, and *Phylloscopus chloronotus*), but wintering in broad-leaved temperate, subtropical and tropical belts (Grimmett *et al.*, 2011, 2019). These birds were better sustained in the LCAS and Forest (19-23 species) than in other IFS types (13 species each). However, breeding range of most of the altitudinal migrants (79 species) such as *Aethopyga gouldiae, Arachnothera magna, Pycnonotus striatus, Phylloscopus castaniceps, Tesia olivea* also extended to subtropical and tropical zone (i.e., <2000 m), thus explaining their presence during breeding season in the IFS (53-62 species) and adjoining Forest (57 species). The mosaic landscape also sustained SV birds including eight Cuckoo species (e.g., *Cacomantis passerines*, *Cuculus* spp., *Hierococcyx* spp., *Surniculus lugubris*), *Monticola cinclorhynchos*, *Geokichla citrina*, *Merops leschenaulti* that breeds in the Himalaya, as well as non-breeding migrants (e.g., *Monticola solitaries, Phoenicurus coeruleocephala*), undertaking latitudinal migration (Grimmett *et al.*, 2019).

Among the three habitat specialization guilds, species richness of forest specialist increased, but openland species declined (whereas, forest generalist lacked clear trend) along the agroecosystem-forest gradient. The present finding follows the general trends seen along the land-use intensity gradient (Goodale *et al.*, 2014; Menon *et al.*, 2019). Such pattern relates to differences in the habitat requirements and diet breadth (narrow for FS vs. wide for FG and OA) for different guild categories, changes in diet and nesting resources, and biotic homogenization in bird communities along the disturbance gradient (Ranganathan *et al.*, 2008; Tscharntke *et al.*, 2008; Şekercioğlu, 2012; Dahal *et al.*, 2014; Morante-Filho *et al.*, 2016; Elsen *et al.*, 2018; Menon *et al.*, 2019; Shahabuddin *et al.*, 2021).

The conservation concern species (PEBi; 18 species), along with other land use sensitive guilds (particularly FSCIBi: 38 species; FSUIBi: 35 species; FSMSIBi: 20 species; and FSFNBi: 18 species), made up more than half of the total avifauna species in the present study. The PEBi alone represented 8.14 % of the total avifauna. These PEBi species include three out of 10 bird species endemic/range-restricted to Eastern Himalaya (Acharya and Vijayan, 2010), and four out of 37 IUCN Red List Threatened and Near-threatened species (BirdLife International 2020a) found in Sikkim. They also represent 12 out of 158 CITES (Appendix I and II) and 10 out of 102 WPA 1972 (Schedule I) bird species of India (Anonymous, 2010; UNEP-WCMC, 2018; Praveen *et al.*, 2020b). Thus, the IFS and adjoining forest ecosystems of Sikkim Himalaya sustained high diversity of land use sensitive birds including protected and endemic species further signifying their conservation potential. The forest specialist, protected and endemic birds of the Himalaya are threatened due to land use change (Pandit *et al.*, 2007; Shahabuddin *et al.*, 2021), however the IFS and adjoining forests of Sikkim, could sustain these land use sensitive guilds.

4.4.2. Alpha diversity of total birds and land use sensitive sub-groups

In the present study, along the agroecosystem-forest gradient, LCAS had significantly highest species per point, abundance per point and Shannon-Wiener diversity for total birds, which declined in Forest relative to other IFS types. However, the land use sensitive guilds (based on alpha diversity, abundance per point and/or

Shannon-Wiener diversity) were most resilient in LCAS and/or Forest that increased along the agroecosystems-forest gradient for FSUIBi (also during breeding and winter), FSMSIBi (also during monsoon and winter), FSCIBi (also during breeding, dispersal and winter), FSFNBi (also during breeding and dispersal seasons), and PEBi (pre monsoon, dispersal, winter).

The three forest specialist insectivores bird sub-groups viz., FSUIBi, FSMSIBi, and FSCIBi were best sustained (richness, abundance/diversity) in LCAS and/or Forest and increased along the agroecosystem-forest gradients. In the present study, FSUIBi species e.g., *Niltava grandis, Suthora nipalensis, Actinodura nipalensis, Yuhina bakeri* and *Leiothrix lutea;* FSCIBi such as *Phylloscopus reguloides, Phylloscopus poliogenys,* and *Pteruthius aeralatus;* as well as FSMSIBi like *Dendrocopos darjellensis, Yungipicus canicapillus, Picumnus innominatus, Certhia discolor, Certhia nipalensis* were highly abundant or only observed in LCAS and Forest. The land use and climate change will also have negative consequences for forest-dependent insectivores (Ibarra *et al.*, 2017; Sreekar *et al.*, 2015; Shahabuddin *et al.*, 2021) viz., FSUIBi and FSMSIBi (specially during breeding and winter) and FSCIBi (during all seasons). However, the FAS and MOAS played a complementary role as they also harboured many unique forest specialist insectivores including breeding and conservation concern species specialized for canopy (*Merops leschenaultia, Aegithalos iouschistos, Melanochlora sultanea*), midstorey (*Micropternus brachyurus, Gecinulus grantia*), and understorey (*Tesia cyaniventer, Pomatorhinus ruficollis, Erythrogenys erythrogenys*) strata. Likewise, many species of the FSUIBi (*Abroscopus superciliaris, Garrulax monileger Pterorhinus pectoralis, Garrulax leucolophus*), FSMSIBi (*Chrysocolaptes guttacristatus, Phaenicophaeus tristis, Sitta frontalis*), and FSCIBi (*Clamator coromandus, Cutia nipalensis,* *Phylloscopus cantator*) were very abundant in these two IFS especially during the breeding and dispersal seasons.

These three forest specialist insectivore sub-groups were well sustained in habitats that retained large trees and dense canopy cover and dense undergrowths, which were mostly found in LCAS and Forest. The resulting vertical and horizontal complexity (Acharya *et al.*, 2011a; Acharya and Vijayan, 2017) provided most conducive habitat and resources across seasons. The FSUIBi and FSCIBi preferred sites located at higher elevation (and low MAT/MAP), whereas opposite was true for FSMSIBi. The contrasting response shown by these sub-groups could result from the difference in prevailing microhabitat and microclimatic regime at different vertical strata, and specialization of these forest-dependent insectivores (canopy and understorey vs. midstorey). Therefore, further habitat loss and landscape simplification in the mosaic landscape needs to be halted (Pandit, 2007) to prevent local extirpation of these forest-dependent insectivores (especially breeding, threatened and range-restricted species) in the region.

The FSFNBi (alpha diversity and abundance per point) was best retained in Forest and LCAS and increased along agroecosystem-forest gradient (especially during breeding and dispersal seasons). FSFNBi communities require large trees with higher gbh (i.e., higher tree basal area), and closed canopy (i.e., higher pcc), which provide resources for breeding. The flowering epiphytes including orchids, trees such as *Leucosceptum canum, Rhododendron* spp. producing flower-nectar and large fruiting trees like Figs, *Castanopsis* spp., *Bridelia retusa* producing berries/fruits providing foraging resources to FSFNBi communities were more species-rich and abundant in LCAS and Forest. This study revealed high vulnerability of FSFNBi communities to land use and climate change, due to their low tolerance to landscape simplification (i.e., to declining tree basal area and pcc) and warming (i.e., to increasing MAT), and preference for higher elevation. Strong negative effects of land use and climate change on forest-dependent large-bodied frugivores (Sreekar *et al.*, 2015; Bovo *et al.*, 2017) such as *Treron apicauda, T. sphenurus,* and small bodied nectarivores (Bennet *et al.*, 2014) e.g., *Aethopyga gouldiae, A. ignicauda, A. nipalensis,* and *A. saturata* will affect the ability of these birds to properly track resources, and thus disrupt pollination and seed dispersal services in the Eastern Himalaya.

The dominance of insectivores, frugivores and nectarivores (representing >80% species pool) in the total bird communities, as well as land use sensitive sub-groups in the present study, corroborates the global pattern seen in the tropical forests and agroforests (Şekercioğlu, 2012). Studies have reported pronounced seasonality in insectivorous (especially altitudinal migrant, summer visitor; and canopy species), frugivorous and nectarivorous birds for tracking respectively insects' larva/adult, fruits/berries and flower-nectar resources in diverse ecosystems (Şekercioğlu, 2012; Mulwa *et al.*, 2012; Bennet *et al.*, 2014; Katuwal *et al.*, 2016a; Gleditsch *et al.*, 2017). The anthropogenic land use and climate change has particularly threatened the forest-dependent species of insectivores (particularly, FSCIBi and FSUIBi), frugivores and nectarivores (i.e., FSFNBi) (Bennet *et al.*, 2014; Bregman *et al.*, 2014; Sreekar *et al.*, 2015). Therefore, there are threats of disruption in provisioning of insect pest control, pollination and seed dispersal services in both agroecosystems and adjoining natural Forests (Şekercioğlu, 2012; Shahabuddin *et al.*, 2021). However, the IFS (mainly LCAS) and Forest provided resources for foraging e.g., foliage-dwelling arthropods larva (Ghosh-Harihar, 2013) to insectivores, flower-nectar to nectarivores and berries/seeds to frugivores. They also provided resources for breeding (e.g., for nesting) and microclimatic refugia (Şekercioğlu *et al.*, 2007).

Although, total species richness of PEBi species harboured in FAS was comparable to adjoining natural Forest (but declined in MOAS and LCAS), however alpha diversity and abundance per point were best sustained in Forest. The nine PEBi carnivore/piscivore bird species (dominated by forest generalist and openland species), were better retained in FAS, MOAS and Forest but declined in LCAS (6 species each vs. 3 species). In fact, two species of Owls protected under CITES Appendix II, were either very abundant (*Glaucidium cuculoides*) or only seen (*Glaucidium radiatum*) in FAS. The five terrestrial Raptors protected both globally (under CITES Appendix II), and nationally (under WPA 1972 Schedule I), were less represented in LCAS (only 2 species) than in other two IFS (3-4 species) or Forest (5 species). However, one scavenger viz., *Gyps himalayensis* and a piscivore i.e., *Haliaetus humilis*, protected under IUCN Red List Near threatened category, CITES Appendix II, and WPA 1972 Schedule I, were only observed in MOAS. On the other hand, protected and endemic insectivores and frugivores/granivores (dominated by forest specialists) were well sustained in all ecosystems except MOAS (6-8 species vs. 3 species). The four insectivorous PEBi species were very abundant or only observed in LCAS and Forest including *Actinodura nipalensis* and *Yuhina bakeri* (both Eastern Himalaya Endemic/ range-restricted species), as well as *Leiothrix lutea* and *Leiothrix argentauris* (both protected under CITES Appendix II). However, *Phylloscopus cantator* (an Eastern Himalaya Endemic species) was more abundant in MOAS and FAS, whereas, *Sitta formosa* (IUCN Red list Vulnerable species) was observed in FAS (and Forest). The three protected frugivores/granivores species included *Buceros bicornis* (protected under IUCN Red list Vulnerable category;

CITES Appendix I; WPA Schedule I), as well as *Lophura leucomelanos* and *Pavo cristatus* (both protected by WPA Schedule I), with former only observed in FAS, while other two in the LCAS and/or Forest as well. Therefore, the extirpation of these PEBi species will disrupt delivery of key ES (especially vertebrate pest control and scavenging, and nutrient deposition as well as insect pest control and seed dispersal ES) in the agroecosystems and adjoining Forests (Şekercioğlu, 2006, 2012; Bregman *et al.*, 2014; Shahabuddin *et al.*, 2021).

4.4.3. Alpha diversity of avian-mediated ecosystem service providers

In the present study, the alpha diversity and abundance per point of invertebrate pest control and pollination provider birds were significantly highest in LCAS and/or Forest and showed increasing trend (more clearly in case of pollination) along the agroecosystem-forest gradient. A large number of insectivores such as *Actinodura strigula, Leoithrix lutea, Grammatoptila striata, Phylloscopus reguloides, Phylloscopus poliogenys, Niltava grandis, Muscicapa ferruginea*, and *Pteruthius aeralatus* were observed very abundantly in LCAS and Forest. Furthermore, many other insectivores like *Machlolophus xanthogenys* (Photo plate 4.1D), *Turdus boulboul*, *Sitta himalayensis,* and *Suthora nipalensis* were altogether absent from FAS and MOAS. On the other hand, winter visitor insectivores such as *Lanius cristatus, Ficedula albicella, Pterorhinus ruficollis,* and *Monticola solitarius* were only encountered in FAS and MOAS, whereas, others in comparable abundance, e.g. *Minla ignotincta, Chrosophlegma flavinucha, Sitta frontalis, Phylloscopus xanthoschistos, Eumyias thalassimus,* which explains greater importance of these two IFS during winter season. Similarly, most of forest specialist pollinator birds were observed only (e.g., *Aethopyga ignicauda, A. nipalensis*) or more abundantly (e.g., *Aethopyga* *gouldie, A. saturata, Dicaeum ignipictus*) in Forest and LCAS, whereas, some forest generalist species such as *Aethopyga siparaja* (Photo plate 4.1A) were highly abundant in other two IFS. The observed dominance of insectivores and pollinators in Forest and LCAS is driven by the environmental variables, since elevation exerted positive effect on these two vulnerable sub-groups, whereas, MAT, MAP, and tree species richness had negative effects. The high persistence of these two ES providers is also linked to seasonal tracking for resources viz., insect larvae/adults by insectivores (Katuwal *et al.*, 2016; Şekercioğlu, 2012), and flower-nectar by nectarivores (Cotton, 2006; Mulwa *et al.*, 2012), which were better provisioned in LCAS and Forest. These avian sub-groups require undisturbed forests or retention of forest remnants in the agroecosystems for survival and breeding, declining along land use gradient (Sreekar *et al.*, 2015; Neuschulz *et al.*, 2016; Shahabuddin *et al.*, 2021). The high alpha diversity of insectivores and pollinators (especially forest specialists), in the traditional agroecosystem namely LCAS concurs with past studies from similar human-modified ecosystems (Bhagwat *et al.*, 2005a, b). Their vulnerability matches with those of FSFNBi and FSCIBi which were similarly found to be highly susceptible to anthropogenic land use and climate change. Woodpeckers helps to control infestation of wood-borer beetles, and promote natural regeneration in natural Sal forest stands, that dominates the tropical belt of Sikkim and elsewhere in the Himalaya) (Singh, 2010), which were more abundant and diverse during winter.

On the contrary, the alpha diversity (and abundance per point) of seed dispersal, vertebrate pest control & scavenging, nutrient deposition, and ecosystem engineering ES providing birds were highest in MOAS and FAS, declining along the agroecosystem-forest gradients. Conversely, along the land use intensity gradient (i.e., from Forest to MOAS), birds providing these four ES increased both in terms of species richness and abundance in the present study, which is driven by the significant negative effect of elevation (but positive by MAT and MAP) on these bird sub-groups. Although, birds providing seed dispersal services shows high vulnerability to forest disturbance (Sreekar *et al.*, 2015; Neuschulz *et al.*, 2016), however, the agroecosystems of Sikkim (specially FAS and MOAS) sustained higher species richness and abundance of these birds than adjoining natural Forests. The present finding could be due to dominance of forest generalist and openland species in frugivores/granivores communities of MOAS and FAS, against forest specialist in LCAS and forest. These birds show pronounced seasonal tracking for fruits/seeds resources (Rey, 1995; Şekercioğlu, 2012; Gleditsch *et al.*, 2017). The higher diversity of fruit- and seed-producing plants including *Ficus* spp., *Citrus reticulata*, shrubs, *Oryza sativa*, grasses in the FAS and MOAS helped sustain these functional groups. The large *Ficus* trees (particularly *Ficus religiosa* and *Ficus benghalensis*) (Cottee-Jones *et al.* (2015) maintained by the local communities in the Devithans/Deoralis and within their farms for religious and socio-cultural importance also helped maintain rich diversity of frugivores in the MOAS and FAS. The hunting of large forest specialist frugivores/granivores (e.g., different species of Hornbills, Pheasants, *Psilipogon* spp.) for domestic use or illegal wildlife trade, very prevalent in other parts of the Eastern Himalaya and northeast India (Grimmett *et al.*, 2011, 2019) are almost absent in the region, which further contributed to their retention in the mosaic landscape (specially in LCAS and Forest). Most of the seed dispersing birds such as *Psilopogon asiaticus, Pavo cristatus, Pycnonotus leucogenis* and *Lonchura striata* were very abundant in FAS and MOAS, while others e.g., *Buceros bicornis, Chloris spinoides* were only observed in these IFS. However, many forest specialist seed dispersing birds were very abundant (e.g., *Treron sphenurus, Hemixos flavala)* or restricted (e.g., *Psilipogon sirens, Carpodacus sipahi, Carpodacus subhimachalus, Procarduelis nipalensis, Pyrrhula erythrocephala*) in Forest and LCAS. Hence, LCAS and Forest are critical for sustaining forest specialist frugivores/granivores (Bregman *et al.*, 2014; Sreekar *et al.*, 2015), and safeguard them from land use and climate change (Peters *et al.*, 2019) in mosaic landscapes of Eastern Himalaya.

The nutrient depositing birds were well sustained (richness and abundance) in MOAS and FAS and declined along the agroecosystem-forest gradients due to their location near the banks of river and mountainous streams (at relatively lower elevations with higher MAT/MAP), that provided foraging and breeding habitats to these birds. These birds contributes to cycling of nitrogen and phosphorus in diverse ecosystems in riparian corridors (Fujida and Koike, 2007; Fujida and Kamada, 2016). Most of these birds were highly abundant (e.g., *Delichon nipalense*) in FAS and MOAS, whereas other species (e.g., *Ardeola bacchus, Enicurus scouleri, Enicurus immaculatus, Haliaetus humilis, Halcyon smyrnensis,* and *Motacilla citreola*) were not detected from LCAS and Forest. However, Forest and LCAS had comparable abundance of some species like *Motacilla cinerea* and *Enicurus schistaceus*. The land use change particularly resulting from hydro-power dams (4 dams/ 1000 km^2) and pharmaceutical companies (56 build till date, 48 currently operational), has greatly affected the forests, agroecosystems, and riparian ecosystems (and biodiversity harboured within them) in Sikkim (Pandit, 2017). The resulting disruptions of hydroperiod and water pollution in riparian corridors of the Eastern Himalaya will adversely affect avian-mediated nutrient-cycling across aquatic-terrestrial habitats (Schriever *et al.*, 2014; Pandit, 2017). Furthermore, it will disrupt food-web dynamics (Nakano and Murakami, 2001; Rundio and Lindley, 2012), and plant community composition and structure (Ellis, 2005) in the heterogeneous landscapes of riparian corridors. Therefore, there is a need for restoration of these riparian ecosystems, and prevent further diversion of forest and agroecosystems to other land use types.

The ecosystem engineer birds (richness and abundance) were better retained in FAS and MOAS (with comparatively higher tree species richness but lower tree density) that declined towards Forest for pooled data as well as during dispersal and winter seasons. On the other hand, Forest had highest values during breeding season, which concurs with a recent study (that surveyed birds during breeding season) in the western Himalaya (Shahabuddin *et al.*, 2021), which reported decline in richness and abundance of woodpeckers (the dominant ecosystem engineers in the present study) along land use gradient [i.e., from Oak forest (undisturbed, disturbed, lopped) and pine forest to the agroecosystems, and absent in urban sites]. The landscape heterogeneity, provided critical breeding sites in the form of large dead trees, cavities, cliffs and foraging habitats through mature trees for diverse ecosystem engineer birds (Cockle *et al.*, 2011; Sodhi *et al.*, 2011; Şekercioğlu *et al.*, 2016). The breeding food resources e.g., caterpillars/larvae in foliage, branches, bark, were well provisioned in the midstorey level in LCAS and Forest. In fact, during breeding season, I encountered that branches and bark on the stems of shade trees in LCAS were covered with thick-layer of Lepidoptera caterpillars, on which insectivores feed upon. The other two IFS provided refugia habitat for altitudinal migrants like *Tichodroma muraria, Sitta formosa,* and *Dendrocopos darjellensis* during dispersal and winter seasons. Thus, cavities and burrows build by ecosystem engineer birds are likely to be occupied by other birds (and faunal taxa) contributing various ES in the ecosystems including seed dispersal and insect pest control services in the next breeding seasons (Nummi and Holopainen, 2014). Ecosystem engineers such as *Chrysocolaptes* *guttacristatus* (Photo plate 4.1H) and *Picus chlorolophus* were highly abundant in FAS and MOAS, whereas others like *Yungipictus canicapillus* in LCAS and Forest.

The birds providing vertebrate pest control and scavenging services were more diverse and abundant in the MOAS and FAS (but declined in LCAS) at different spatio-temporal scale. About half of these birds (9 out of 19 species) are legally protected globally or within India, which were comparably species-rich and abundant in all ecosystems except LCAS (6 species each vs. 3 species). The other 8 non-protected scavenging/omnivorous birds were all observed in Forest, whereas, two piscivores viz., *Ardeola bacchus* and *Halcyon smyrnensis* in MOAS and FAS. However, 5 out of 6 scavenging/omnivorous birds (e.g., *Acridotheres tristis*, *Corvus* spp., and *Cissa chinensis*) were very abundant in the IFS. Therefore, integrating the diurnal birds of prey e.g., Raptors (Kross *et al.*, 2012), and their nocturnal counterparts i.e., various species of Owls (Lee, 1997; Meyrom *et al.*, 2009; Pande and Dahanukar, 2011; Vanitha *et al.*, 2014; Kross *et al.*, 2016), into the farms can contribute to control population outbreak of rodents (and other vertebrate pests) in diverse agroecosystems. This cost-effective and eco-friendly bio-control method can help reduce crop damage as well as safeguard public health (Kan *et al.*, 2014; Kross *et al.*, 2012, 2016). Likewise, the scavengers (and omnivores) can play crucial role in carcass and wastes disposal, energy cycling as well as disease control (Peterson *et al.*, 2001; Şekercioğlu, 2006; Markandya *et al.*, 2008; DeVault *et al.*, 2016) in agroecosystems and forest ecosystems. These vertebrate pest control and scavenging ES provider birds (like other avian ES provider and land-use sensitive and) were well retained in the mosaic landscape of Sikkim due to various reasons. First, it was due to high coverage of PAs and forests in Sikkim (which provided breeding habitat), and strict protection measures already in place for nine of these species under global

(IUCN Red List, CITES Appendix II) and/or national (WPA 1972 Schedule I) laws/conservation frameworks. Second, the reverence and informal protection bestowed upon these carnivores (e.g., Raptors and Owls) and scavengers/omnivores (e.g., Vultures, Crows) driven by their unique socio-cultural and religious significance (in Hinduism and Buddhism) are still kept intact by the ethnic communities in Sikkim (Grimmett *et al.*, 2011, 2019). Third, poisoning of vultures and other scavenging birds (both intentional or through use of banned veterinary drugs like Diclofenac) as well as hunting and illegal wildlife trades (of all birds of prey and other sub-groups), are almost absent in the region (Grimmett *et al.*, 2011, 2019). Finally, introduction of organic farming (i.e., lack of chemical pesticides/insecticides/herbicides use) has greatly benefitted these birds (occupying top level of the food chain), by reducing chances of breeding failure linked to biomagnifications. However, breeding grounds/habitats and nests of these birds (also of all other avian sub-groups) needs to be protected from any disturbance from the ongoing and future developmental activities (e.g., road construction/widening, hydro-power dams, industrialization and urbanization) and tourism sector across ecosystems in the Eastern Himalaya (Laiolo, 2004; Pandit, 2017; Banerjee *et al.*, 2019).

4.4.4. Seasonal dynamics in bird communities

The total as well as different land use sensitive guilds also showed strong seasonal dynamics. The bird communities clearly tracked season showing an increasing trend in both species richness and abundance per point from the start of breeding till dispersal seasons (pre monsoon<monsoon<post monsoon), followed by slight decline in winter relative to post monsoon in all ecosystems for total bird community or in LCAS and Forest for FSFNBi, FSUIBi, or in MOAS and FAS for FSCIBi. However,

IFS particularly MOAS and FAS continued to retain comparable or higher species richness and abundance of birds than Forest in winter seasons for TBi, FSFNBi, FSUIBi (also invertebrate pest control and pollination providers), as well as in pre-monsoon than LCAS for FSUIBi, FSMSIBi. The bird diversity and population trends in different habitats are also known to be influenced by seasons (Acharya *et al.*, 2010; Katuwal *et al.*, 2016; Elsen *et al.*, 2018; Goded *et al.*, 2018). The observed dominance of bird community in monsoon and post monsoon season across the four ecosystems followed the general trend in the Himalaya (Acharya *et al.*, 2010; Chettri, 2001; Elsen *et al.*, 2018; Katuwal *et al.*, 2016a) and elsewhere (Kunte *et al.*, 1999; Goded *et al.*, 2018). This clear seasonal pattern in the bird communities are associated with monsoon mediated reproductive cycles of the Himalayan birds (Ali and Ripley, 2002) when summer visitors (19 species in present study) also arrive for breeding.

The comparatively enhanced bird species richness in the agroecosystems than natural Forest during the winter is due to the influx of different migrant species. The altitudinal migrants (107 species in the present study) were slightly richer in LCAS and MOAS than Forest. These birds migrates down to the low- to mid-elevation area (<1800 m) from their breeding ground located in higher elevation to evade the harsh cold climate and food scarcity (Ali, 1962; Grimmett *et al.*, 2019). The incoming winter visitors (12 species in the present study) also showed higher preference for the IFS (particularly FAS) than Forest ecosystems. The agroecosystems provides important refugia to the bird communities particularly during the winter as well as during breeding and dispersal seasons in the Himalaya (Elsen *et al.*, 2016, 2018) and elsewhere (Goded *et al.*, 2018; Yabuhara *et al.*, 2019). Due to its dominance in the low- and mid-hills with low PAs coverage but high biodiversity including birds (Acharya *et al.*, 2011a), the IFS of Sikkim acted as sink habitat (Rahbek, 1997) during the dispersal and winter seasons (Elsen *et al.*, 2016, 2018). The spatio-temporal variation of montane bird diversity and assemblage, and their response to land use change in the tropical region is also strongly linked with fluctuations in temperature and rainfall (Acharya *et al.*, 2011a; Echeverri *et al.*, 2019; Srinivasan *et al.*, 2019).

The Himalayan agroecosystems supports high bird diversity and community composition of total birds during winter (Laiolo, 2004; Elsen *et al.*, 2016), as well as breeding seasons (Elsen *et al.*, 2018). However, this study clearly demonstrates the importance of the agroecosystems of Eastern Himalaya to support bird communities providing different ES providers (also including the different land use sensitive guilds) at different spatial scale and across seasons. Nevertheless, when compared to the three IFS types, Forests had higher or comparable alpha diversity and/or abundance per point for birds providing pollination and insect pest control (during breeding and dispersal season), ecosystem engineering (in breeding season) and seed dispersal services (in dispersal season). Hence, the three IFS can complement the adjoining natural Forest (outside the PAs) in sustaining the diversity of different land-use sensitive and vulnerable avian-mediated ES provider sub-groups at different spatial scale and across seasons in the mosaic landscape of Sikkim.

4.4.5. Relationship of bird alpha diversity with environmental variables

In the present study, tree basal area and density exerted negative effect only on abundance of total birds. The habitat variables likewise showed significant relationships (in terms of alpha diversity/abundance per point, and/or Shannon-Wiener diversity/ species richness) with at least one of the bird sub-groups. There was significant negative effect of tree species richness for insect pest control providers (i.e., insectivores), especially on FSMSIBi. The tree density exerted significant negative effect on ecosystem engineering service providers and FSUIBi (only abundance per point). The percentage canopy cover had significant negative effect for PEBi, whereas, it showed significant positive relationship for FSMSIBi. The tree basal area had significant positive effect for FSFNBi, and FSMSIBi. Past studies on total birds have also found significant effects of habitat variables, for example, tree species richness (Acharya *et al.*, 2011a), tree density and basal area (Clough *et al.*, 2009; Chettri, 2010; Acharya *et al.*, 2011a, Buechley *et al.*, 2015; Karanth *et al.*, 2016; Lee and Carrol, 2018), canopy cover (Van Bael *et al.*, 2007; Anand *et al.*, 2008), primary forest remnant (Anand *et al.*, 2010), patch size (Ehlers Smith *et al.*, 2018). The higher tree basal area, density and percentage canopy cover could have contributed to greater NDVI in LCAS and Forest. The net primary productivity (NPP) (Pellissier *et al.*, 2018) also positively determines the birds' richness and community composition in different ecosystems, as observed in the present study.

In the present study, H′ (but not alpha diversity/ abundance per point) for total birds increased significantly with elevation (but decline in MAT). Among the different land-use sensitive and avian-mediated ES provider sub-groups, alpha diversity, abundance per point and/or H′, richness of birds providing pollination and insect pest control services, and the related land use sensitive guilds viz., FSFNBi, FSCIBi, and FSUIBi increased with elevation (but declined with increasing MAT and/or MAP). On the contrary, diversity pattern for vertebrate pest control and scavenging, nutrient deposition and/or ecosystem engineering providers, and land-use sensitive guilds namely PEBi (dominated by former two ES providers), and FSMSIBi (dominated by woodpeckers and other cavity and burrow-excavating birds) significantly declined with elevation (but increased with MAT and/or MAP). Past studies has reported decline in diversity of total birds along the elevation gradient from the Eastern Himalaya (Chettri *et al.*, 2001; Acharya *et al.*, 2011a) and elsewhere (Jankowski *et al.*, 2013), similar to present observations on three ES providers i.e., vertebrate pest control and scavenging, nutrient deposition and ecosystem engineering, and two related land-use sensitive guilds viz., PEBi and FSMSIBi. In the present study, MAT (r^2 = -0.89) and MAP (r^2 = -0.84) strongly declined along the elevation gradient, similar to the prior report from the same study region (Acharya *et al.*, 2011a). The different climatic variables particularly, temperature and precipitation/AET are among the most important determinant of bird alpha diversity in the Eastern Himalaya (Acharya *et al.*, 2011a; Srinivasan *et al.*, 2019), and elsewhere in the tropical regions (Echeverri *et al.*, 2019).

The bird species were assembled according to vegetation, elevation, and climatic conditions, which differed between complex (i.e., LCAS, Forest) and simplified (MOAS, FAS) ecosystems. LCAS has more similarity with natural forests and PAs in terms of vegetation cover (Chettri *et al.*, 2005). In the present study, the LCAS was more similar to Forest in terms of habitat factors, i.e., higher pcc, tree basal area and density, but lower tree species richness, and environmental variables, viz., higher average elevation (but lower MAT, MAP), than it did with other two IFS types (which in turn showed opposite values) (elaborated in Fig. 4.1; Table 4.2; Appendix A, also shown by NMDS ordinations: Fig.4.11-4.12). The resulting landscape heterogeneity in present wildlife-friendly agricultural landscape could have enhanced diversity, abundance and species richness of birds including land use sensitive guilds by ensuring availability of nesting and sheltering places in non-crop habitats (Fischer *et al.*, 2011; Buechley *et al.*, 2015; Hiley *et al.*, 2016; Goded *et al.*, 2018; Katayama *et al.*, 2019).

4.4.6. Bird beta diversity

Along the agroecosystem-forest gradient, the pair-wise beta diversity largely followed a significant increasing trend for the total birds and different sub-groups (FSUIBi, PEBi, Invertebrate pest control), whereas, FSMSIBi as well as for Seed dispersal; Pollination; and Vertebrate pest control & Scavenging service providers increased non significantly along the gradient. Furthermore, FAS had greater (for FSCIBi and FSFNBi, Nutrient deposition, Ecosystem engineering), or comparable (in case of FSMSIBi and PEBi) values of pair-wise beta diversity than Forest, thus indicating gentle (i.e., less steep) declining trend along the land use gradient. The present finding differs from the steep decline in beta diversity usually reported along agriculture intensification or land use intensity gradient for birds and other taxa in India (Kunte *et al.*, 1999; Sreekar *et al.*, 2020) and elsewhere (Karp *et al.*, 2012; Baiser *et al.*, 2012; Gámez-Virués *et al.*, 2015; Almeida *et al.*, 2016; Hiley *et al.*, 2016) due to their scale dependent effects.

Comparison of trends observed for beta diversity (Fig. 4.9-4.10) with that of alpha diversity (Fig. 4.3-4.5) for total birds and different sub-groups (showing significant differences for both these indices) revealed interesting insights on likely biotic (taxonomic) homogenization or heterogenization in the bird communities. There was significant decline of beta diversity (relative to alpha diversity) in LCAS (for TBi, FSUIBi, FSCIBi, FSFNBi, PEBi, Insect pest control) as well in MOAS (for TBi, FSUIBi, Nutrient deposition and Ecosystem engineering providers), which implied likely taxonomic homogenization in them. However, there were corresponding signs of taxonomic heterogenization in FAS for most land use sensitive guilds, viz., FSUIBi, FSCIBi, FSFNBi, PEBi and ES providers, i.e., Nutrient deposition, Ecosystem engineering, Insect pest control as well as in Forest (for FSCIBi, PEBi, Insect pest control, Nutrient deposition, Ecosystem engineering), and LCAS (for Nutrient deposition). In fact, beta diversity were significantly highest in FAS followed by Forest for two land use sensitive guilds (viz., FSCIBi and FSFNBi), ES providers (Nutrient deposition, Ecosystem engineering), as well as comparable for FSMSIBi and PEBi communities. The exception to this trend were found in other sub-groups viz., for FSMSIBi, Pollination, Seed dispersal, as well as Vertebrate pest control & scavenging, which largely had similar trend of alpha diversity or lacked significant difference for beta diversity. This suggests likely taxonomic homogenization in MOAS and LCAS, but heterogenization in FAS and Forest in the bird communities (Baiser *et al.*, 2012; Karp *et al.* 2012; Socolar *et al.*, 2016).

The beta diversity decline in bird communities of LCAS could be linked to its location within sub-tropical to temperate zones, and thus less variability in terms of MAT, MAP, unlike that of FAS and Forests, which extended from tropical to temperate belts. Therefore, declined beta diversity in LCAS may not necessarily imply high degree of homogenization since its bird community was dominated by forest specialist insectivores, frugivores and nectarivores species (in higher abundance) (Socolar *et al.*, 2016). Furthermore, community composition of birds in LCAS was more closely related to Forest, with whom it formed a distinct cluster separated from other two IFS (MOAS and FAS) for total birds as well as most of the sub-groups. However, in case of birds providing Seed dispersal and Ecosystem engineering service, the species compositions in three IFS types formed a cluster distinct from the Forest. This might be due to the high sensitivity of the forest birds to disturbance, and need for undisturbed forest habitat to maintain beta diversity.

This study also supports the significance of maintaining a mosaic landscape (formed by different types of IFS and Forest), for sustaining/enhancing bird communities at different spatio-temporal scales. When compared to Forest, IFS (particularly LCAS and FAS), respectively retained higher/or comparable alpha diversity and/or beta diversity of different land use sensitive guilds, and avian-mediated ES providers, thereby highlighting their conservation potential. This maintenance of bird beta diversity in wildlife-friendly mosaic landscape of Sikkim (especially in FAS and LCAS), helped mitigate the taxonomic homogenization in the bird communities (Devictor *et al.*, 2008; Şekercioğlu, 2012; Gámez-Virués *et al.*, 2015; Almeida *et al.*, 2016). The present finding concurs with the prior reports of retention of comparable/ higher spatial and temporal beta diversity of birds and other faunal taxa in low-intensive (Tscharntke *et al.*, 2008; Doxa *et al.*, 2010), high nature value (Doxa *et al.*, 2012), organic and/or traditionally managed agroecosystems (Philpot *et al.*, 2007) and heterogeneous landscapes (Tscharntke *et al.*, 2008; García-Navas *et al.*, 2020), than the adjacent Forest ecosystems. The varieties of crops grown, differences in management and retention of heterogeneous vegetation in the three IFS of Sikkim (along with variation in habitat and environmental variables) also contributed to higher habitat and landscape heterogeneity in the study area.

The bird community composition (based on NMDS), was structured significantly by the different habitat (tree species richness, tree density, tree basal area, percentage canopy cover) and environmental (elevation, MAT and MAP) variables for total birds and/ or at least one of the land use sensitive guilds and ES providers. The ecosystems also significantly determined the community assemblage for total bird as well as three land use sensitive (FSUIBi, FSMSIBi, and FSCIBi) and all ES provider guilds. For total birds in the present study, I found significant positive relationship of overall pair-wise beta diversity (βbray) and its substitution components (βbray.bal) with geographical distance, ΔElevation, ΔMAT and ΔMAP, Δtree density, and Δtree basal area, and additionally of βbray with Δpercentage canopy cover**.** However, the nestedness component (βbray.gra) showed significant positive relationship with only Δtree species richness and Δtree basal area. Past studies, which mostly reported Sorensen dissimilarity index, has found significant and strong effect of various habitat and environmental factors for total pair-wise beta diversity, and its two components. The overall pair-wise beta diversity, along with its substitution (turnover) and/or nestedness components are strongly structured by elevation (Clough *et al.*, 2009; Acharya *et al.*, 2011a, Jankowski *et al.*, 2013; Si *et al.*, 2015; Li *et al.*, 2019; García-Navas *et al.*, 2020), and geographical distance between transects (Hu *et al.*, 2018, Sreekar *et al.*, 2020). They also depend on climatic factors e.g., MAP/rainfall, MAT/temperature, and NDVI/primary productivity (Li *et al.*, 2019; Zellweger *et al.*, 2017; Sreekar *et al.*, 2020).

Beta diversity and community composition of birds in agroecosystems is primarily structured by vegetation structure and composition, through enhanced canopy height/ canopy height variation (Zellweger *et al.*, 2017), habitat complexity and heterogeneity (Jankowski *et al.*, 2013; Hu *et al.*, 2018; García-Navas *et al.*, 2020), landscape heterogeneity/ landscape composition at broad scale (Tscharntke *et al.*, 2008; Morante-Filho *et al.*, 2016). It strongly depends on local factors, e.g. tree species richness and composition (Clough *et al.*, 2009; Jankowski *et al.*, 2013), tree density (Clough *et al.*, 2009) and basal area (Barlow *et al.*, 2007a). The high beta diversity (both overall and turnover component) is congruent with high taxonomic and phylogenetic turnover of plant communities (Acharya *et al.*, 2011b; Manish *et al.*, 2017; Manish and Pandit, 2018; Shooner *et al.*, 2018), and insects (Tscharntke *et al.*, 2008). Finally, beta diversity declines along land-use gradient (Sreekar *et al.*, 2018, 2020) as well as with distance to forest/PAs (Clough *et al.*, 2009), and differs across spatial (Karp *et al.*, 2012; Hu *et al.*, 2018; Sreekar *et al.*, 2020) and temporal (García-Navas *et al.*, 2020) scales.

The multiple site beta diversity was highly dominated by the substitution component (in case of both original and simulated values) for the total birds as well as different land use sensitive sub-groups and four avian-mediated ES providers. The present observation confirms the high landscape heterogeneity in Sikkim. The high habitat and landscape heterogeneity in Sikkim are attributed to its variability in the habitat variables (tree species richness, tree density tree basal area, and percentage canopy cover), and environmental variables (MAT, MAP, and elevation) which in turn showed varied response to the bird communities providing different ES in the present study (Fig. 4.1; Table 4.2; Appendix A). It is also linked to wildlife-friendly management of agroecosystems which are interspersed by the farm-forests and natural forests outside the PAs forming a mosaic landscape. The high coverage of forest cover (>31%) and PAs (>47%) (ISFR, 2019) in Sikkim, and proximity of the agroecosystems (particularly LCAS and FAS) to the PAs, also played positive role. Similar pattern of dominance of substitution components in beta diversity has been previously reported for birds and other faunal taxa (Si *et al.*, 2015, Baselga, 2017; Soininen *et al.*, 2018; Sreekar *et al.*, 2020) in heterogeneous landscapes. Past studies along the elevation gradient (300-4500 m) in the forest ecosystems from same region has also reported high beta diversity (turnover components) in birds (Acharya *et al.*, 2011a), and trees (Acharya *et al.*, 2011b).

I find that traditional and organic agroecosystems of Sikkim can play crucial role in enhancing beta diversity of birds including land use sensitive guilds, and consequently maintain different avian-mediated ES. The present observation is pursuant to past studies which reported that traditional, organic and high nature value agroecosystems can preserve high beta diversity by negating the land-use change and agricultural intensification driven biotic homogenization in birds (Tscharntke *et al.*, 2008; Doxa *et al.*, 2012), and butterflies (Sharma *et al.*, 2020). Therefore, conservation measures for total birds, different land use sensitive and four ES providers viz., insect pest control, pollination, seed dispersal, and ecosystem engineers (with high to very high dominance of substitution components in overall multiple-site beta diversity) must focus on Forest as well as all IFS of Sikkim (Dobrovolski *et al.*, 2012; Baselga, 2017; Sharma *et al.*, 2020). Conversely, comparatively higher contribution of nestedness component to overall multiple-site beta diversity (than other sub-groups) for vertebrate pest control and scavenging, and nutrient deposition ES providers implies that conservation strategies targeted on the Forest and FAS in the present study might largely suffice for conserving these two important yet understudied ES providers (Dobrovolski *et al.*, 2012). Hence, the land-sharing approach should be applied in the low- and mid-hills, as it can complement the PAs and consequently, land-sparing approach (above 1500 m) for conservation of biodiversity and ES in the Eastern Himalaya, especially Sikkim. A conservation measures incorporating a mix of land-sparing and land-sharing framework (Grass *et al.*, 2019), will be most suitable in the Eastern Himalaya, instead of only former that focuses exclusively on PAs (Phalan *et al.*, 2011, 2016; Ghosh-Harihar *et al.*, 2019; Manish and Pandit, 2019).

4.5. Conclusion

The IFS and adjoining forests of Sikkim Himalaya have great potential in conservation of birds as they harbour ~38.1% of Sikkim's birds (221 out of 580 species) in only about one-tenth of its TGA. More than half of bird community was composed of land use sensitive guilds including 18 conservation concern species. For the different avian-mediated ES, invertebrate pest control represented ~3/4thof the total bird species, whereas, contribution of other ES providers ranged from 7.24% (for pollination) to 15.84% (for ecosystem engineering). The bird alpha and beta diversity along with community composition were strongly influenced by different habitat and environmental variables that varied among the total birds, land use sensitive guilds, and avian-mediated ES providers. The dominance of substitution components in multiple site beta diversity implies that the conservation measures must focus on all the ecosystems. The IFS (specifically LCAS and FAS) and adjacent Forest ecosystems retained high diversity and community composition of birds including specialists and ES providers at different spatio-temporal scales. These agricultural landscapes were found to mitigate biotic homogenization across land use sensitive and ES provider guilds. The dominance of substitution components in multiple site beta diversity for both total birds as well as most of the sub-groups signifies that the conservation measures must focus on both agroecosystems and forest ecosystems. The study highlights the complementary role of the agricultural landscape in Sikkim Himalaya for biodiversity conservation, especially below 1500 m where coverage of PAs still remains low. Therefore, conservation strategies in the Eastern Himalaya can involve combination of both land-sparing and land-sharing frameworks, instead of only the former approach.

Photo plate 4.1. Representative bird species providing different types of avian ecosystem services observed in IFS and Forest ecosystems of Sikkim Himalaya. Pollinator: Crimson Sunbird *Aethopyga saturata* (A); Seed disperser: Blue-throated Barbet *Psilopogon asiaticus* (B); Grazer: Indian Peafowl *Pavo cristatus* (C); Insect pest control provider: Black-lored Tit *Machlolophus xanthogenys* (D); Vertebrate pest control provider: Crested Serpent Eagle *Spilornis cheela* (E); Scavenger: Common Green Magpie *Cissa chinensis* (F); Nutrient depositor: Little Forktail *Enicurus scouleri* (G); Ecosystem engineer: Greater Goldenback *Chrysocolaptes guttacristatus* (H).

Chapter 5

BUTTERFLY ALPHA AND BETA DIVERSITY ALONG AGROECOSYSTEM-FOREST GRADIENT

Water Snow Flat *Tagiades litigiosa litigiosa* Möschler, 1878

5.1. Introduction

Climate change, agricultural intensification, global environmental changes such as the habitat loss and fragmentation, results in negative consequences for biodiversity and natural capital (Chapin *et al.*, 2000; Foley *et al.*, 2011). Conservation of biodiversity in agroecosystems is a global concern (Vandermeer and Perfecto, 2007; Flohre *et al.*, 2011) as conversion of forest to agroecosystems has been one of the major causes for biodiversity loss through habitat transformation in terms of land-cover changes(Gibbs *et al.*, 2010; Pandit, 2017), landscape simplification and fragmentation (Ekroos *et al.*, 2010; Börschig *et al.*, 2013; Loos *et al.*, 2014; Dainese *et al.*, 2017) associated with rapid land-use intensification (Matson *et al.*, 1997, Tscharntke *et al.*, 2005; Billeter *et al.*, 2008; Karp *et al.*, 2012) and change (Foley *et al.*, 2005, Newbold *et al.*, 2015). The shift in research interest from assessment of biodiversity towards the concept of ES in the context of social-ecological systems (Chapin *et al.*, 2010) requires predictions about the ecological effects of landscape change on biodiversity and ecosystem functioning which ultimately should be translated into management decisions. Along a gradient of landscape contexts interspersed by different levels of human disturbance (from forests to agroecosystems), each landscape can have varied role in the maintenance of ES such as pollination. Pollination by insects is an important ES (Klein *et al.*, 2007; Ollerton *et al.*, 2011) and is associated with landscape factors that also benefit pollinators (Kremen *et al.*, 2007; Potts *et al.*, 2010; Batary *et al.*, 2011; Bommarco *et al.*, 2012; Hadley and Betts, 2012; Kennedy *et al.*, 2013). Various studies have recognized the relationship of pollinator community structure with landscape context (Steffan-Dewenter *et al.*, 2002; Carré *et al.*, 2009; Martins *et al.*, 2015) and pollination service (Bommarco *et al.*, 2012; Garibaldi *et al.*, 2013; Zou *et al.*, 2017).

The crucial role of pollinators in supporting global food supply has gained considerable attention in recent years as they influence several other ES (Losey and Vaughan, 2006; Lindström *et al.*, 2018). Butterflies are the most vulnerable wildlife groups reacting sensitively and rapidly to climate and habitat changes (Thomas, 2005), and butterfly communities respond to both land use type and management intensity, at local (field) and landscape scales (Rundlof *et al.*, 2008; Zingg *et al.*, 2018). Unlike most other insect groups, butterflies are well-documented, their taxonomy better understood, most species are easy to identify and much information is available on their ecology and life-history traits (Thomas, 2005). Additionally, butterflies are characterized by a short life-cycle, feeding and habitat specificity during several stages of their life-cycle, and high diversity and endemism (Dennis, 2010; ILTEO, 2015). These features make butterflies potentially prized as biodiversity indicators, accepted politically from, first of all, EU Commission, and in general by governments (including India) approving the status of butterfly populations as a surrogate of biodiversity and environmental health (Pollard and Yates, 1993; Brereton *et al.*, 2011; ILTEO, 2015).

In addition, butterflies offer a wide range of ES including pollination and pest control (Losey and Vaughan, 2006; Cardinale *et al.*, 2012) and their conservation benefits occur particularly at local as well as the landscape scale (Davis *et al.*, 2007). Landscape variables such as land cover and habitat structure are drivers of butterfly density (Ekroos *et al.*, 2010), because they regulate species' colonization and extinction (Sweaney *et al.*, 2014; Fourcade *et al.*, 2017; Bartomeus *et al.*, 2018). The common local variables reported to influence butterflies include floral abundance, plant community composition, vegetation height, and litter ground cover (Davis *et al.*, 2007; Poyry *et al.*, 2009). Thus, the effect is scale dependent, at local level, alpha diversity generally increases, but at regional level, beta diversity declines significantly (Ekroos *et al.*, 2010; Karp *et al.*, 2012). Studies have found direct relationship between butterfly richness and plant richness at different geographical scales (Hawkins and Porter, 2002; Menéndez *et al.*, 2007; Kitahara *et al.*, 2008). Climatic conditions also strongly determine the metabolism and behaviour of butterflies; therefore, they are suitable for examining species richness-environment relationships (White and Kerr, 2007).

To understand the effects of different management practices on butterflies along an agroecosystem-forest gradient, assessment of both alpha and beta diversity trend is crucial (Karp *et al.*, 2012; Baselga, 2013a). Butterflies represent an indicator taxa which responds rapidly to environmental and land use changes (Kremen, 1992; Singh and Pandey, 2004; Nelson, 2007; Rákosy and Schmitt, 2011; Ekroos *et al.*, 2013) and have been recognized as ecological indicators in diverse ecosystems around the world (Thomas, 2005; Nelson, 2007; Vu, 2007; Barlow *et al.*, 2007b; ILTEO, 2015; Herrando *et al.*, 2016; Basset *et al.*, 2017; Stuhldrehera and Fartmann, 2018).

I hypothesized that traditionally managed organic agroecosystems of Sikkim Himalaya, India can support, at different level, the pollination service provided by high alpha and beta butterfly diversity. This chapter has been designed along agroecosystem-forest gradient: (1) to understand the patterns of butterfly alpha diversity taking into account the variation across seasons, elevation, forest specialization and larval host specificity, (2) to understand the patterns of butterfly beta diversity (community composition), and (3) to find out plausible environmental determinants of butterfly diversity, both at alpha and beta level.

5.2. Materials and methods

5.2.1. Study area and site selection

As described in **chapter 3**, the present study covers three representative agroecosystems viz., (MOAS, FAS, and LCAS) and nearby natural Forest (as control) ecosystem identified along a gradient of shade tree diversity within the elevation of 600-2000 m AMSL in two districts (East and South) of Sikkim, Eastern Himalaya, India. These four ecosystems represent the agroecosystem-forest gradient in the study area. Sikkim is characterized by a very high diversity of butterflies (690 species; Haribal, 1992; Kunte, 2010) comprising almost 50% of the species found in Indian Sub-continent (Kehimkar, 2016). Twenty four transects (six per ecosystem of 1km length each, spaced ≥1 km apart) and 240 permanent sampling points (10 in each transect, spaced ≥100 m apart) across these four ecosystem were laid for butterfly sampling during the study.

5.2.2. Butterfly sampling

Fixed width point count method (Pollard, 1977; Acharya and Vijayan, 2015), a modified form of transect count, was used to sample butterflies along each transect. Sampling was undertaken by halting at the pre-established permanent point and recording the identity and abundances of butterflies for five minutes within the 5m radius plot. Each point was sampled 6-8 times covering four seasons viz., pre monsoon (March-May), monsoon (June-August), post monsoon (September-November) and winter (December-February) along the temporal range from December 2012 to August 2017. The total sampling effort was 1760 samples during the study period with 460 point counts in MOAS, 480 in FAS, 450 in LCAS, and 370 in Forests.

Butterflies were identified on the basis of the wings and identifying characters provided in the standard field guides (Wynter-Blyth, 1957; Haribal, 1992; Kehimkar, 2008). Individuals of doubtful species were photographed for their subsequent identification through a careful examination of upperwing and underwing characteristics. The sampled individuals were identified to the species level, however, Darts *Oriens* spp. and Dartlet *Potanthus* spp. could be identified only to genus level, because their exact identification requires the inspection of the male genitalia (Kunte *et al.*, 2018).

5.2.3. Biological variables of butterflies

I quantified two autecological traits of butterflies: a) larval host specificity (monophagous: within single genus, polyphagous: >1 genus) (Dainese *et al.*, 2017), and b) habitat specialization (forest specialist and generalist) following standard literatures (Haribal, 1992; Kehimkar, 2008, 2016; Sengupta *et al.*, 2014; Kunte *et al.*, 2018) supported by field observations.

Species under conservation concern were obtained from India Red Data Book of butterflies (Gupta and Mondal, 2005), WPA 1972 (Anonymous, 2010), and the CITES (UNEP-WCMC, 2018). This information was used to assess the relative contribution of protected species to the total butterfly community and pattern across the four ecosystems.

5.2.4. Environmental variables

In this study, to understand the environmental determinants of butterfly diversity, different variables representing habitat such as tree density (stems ha^{-1}), tree species richness, and tree basal area (m^2 ha^{-1}), percentage canopy cover (pcc: %), climatic viz., MAT (temp: °C) and MAP (rain: mm), and topographical e.g., elevation (m) factors were quantified following appropriate scientific methods (by field sampling as well as at GIS laboratory, Department of Zoology, Sikkim University) for each of the 24 transects across four ecosystems(described at detail in **chapter 4** of this thesis).

5.2.5. Statistical analysis

All analyses were done using R version 3.1.4 (R Core Team, 2017) using suitable packages.

5.2.5.1. Assessment of sampling completeness

To assess the completion of butterfly sampling, value of Chao1 (a nonparametric estimator of species richness that shows high precision) was estimated on the basis of a matrix of abundance data. The calculated value of Chao1 was plotted against sampling effort to generate species accumulation curve using “iNEXT” package in R.

5.2.5.2. Alpha diversity and determinants

Alpha diversity was measured as species richness (species per point) observed in each point during sampling (Fleishman *et al.*, 2003). I also assessed other community parameters such as abundance per point, total species richness, total abundance, Shannon-Wiener diversity index (H′) for the 24 transects sampled, four systems and overall. To account for slightly unequal sampling effort among study systems, I focused further analysis on alpha diversity, abundance per point and H′. Variation in butterfly alpha diversity, abundance per point and H′ among the systems, for total species (also across seasons and family), forest specialist; monophagous and protected species was tested using one-way ANOVA. For correct identification of the particular context actually making differences in the diversity parameters between the four ecosystems, post-hoc, pair-wise comparisons with Bonferroni correction was also carried out using the package “multcomp” in R.

Correlation test was used to explore the relationship of butterfly community parameters with different habitat and environmental variables, and correlation matrix plot was prepared to depict significant ($p < 0.05$) correlation using “corrplot” package in R. To understand the determinants of butterfly alpha diversity, I ran generalized linear mixed-effects models (GLMMs) using Negative Binomial distribution taking abundance per point and species per point as response variables. MAT and MAP showed strong correlation ($r^2 > 0.8$; $p < 0.05$) among themselves and with elevation, hence, elevation and MAP were retained in the final model (with either elevation or MAP in the same model). Therefore, the following variables: ecosystem (graded as MOAS, FAS, LCAS, Forest), elevation, MAP, season (graded as winter, pre monsoon, monsoon, post monsoon), percentage canopy cover, tree species richness, tree density and tree basal area as fixed effects factors and transect ID as random-effects factor were included in the final GLMMs. I standardized all the continuous covariates to mean 0 and SD 1. I have generated full model, null model and models with all valid combinations of the explanatory variables, compared and ranked models using the second-order Akaike Information Criterion (AICc) (Burnham and Anderson, 2002) with “lme4”, and “MuMIn” packages in R. I also estimated Akaike weights (AICw) that provide relative weights for any particular model in relation to the entire model set, which varies from 0 (no support) to 1 (complete support) (Hobbs and Hilborn, 2006). I summed up the AICw of all the models containing a particular covariate (covariate weight) within the subset (top models; $\Delta AICc < 4$) to identify the covariates that had the strongest influence. The model averaged estimates and their unconditional standard errors of the most parsimonious model (ΔAICc < 4) are presented here.

5.2.5.3. Beta diversity and determinants

Beta diversity was partitioned into components of spatial turnover and nestedness-resultant and into balanced variation in abundance and abundance gradients (Baselga, 2010, 2017). Using incidence-based (Sorensen dissimilarity) and abundance-based (Bray-Curtis dissimilarity) indices, the partitioning of beta diversity were quantified for both multiple site dissimilarity and pair-wise dissimilarity (Baselga, 2013a, 2017) with "betapart" package in R. In both the cases, original abundance based community data matrix for Bray-Curtis dissimilarity indices, and transformed presence/absence matrix (as 0 for absence and 1 for values ≥1 as presence from abundance data) were used to estimate the incidence-based Sorensen dissimilarity indices. In order to investigate how beta diversity could change in the different ecosystems, the pair-wise dissimilarity index for each transects pair was calculated. In particular, Bray-Curtis dissimilarity (βbray) and its partition due to balanced variation in abundance (βbray.bal) and due to abundance gradients (βbray.gra) as well as pair-wise Sorensen dissimilarity (βsor) and its components of turnover (βsim) and nestedness-resultant (βsne) were estimated. The homogeneity (i.e., distance of beta diversity values in relation to their respective centroids) and their significant difference among the ecosystems were assessed using one-way ANOVA. In order to identify the particular context driving the actual differences in the pair-wise beta diversity between the four ecosystems, Tukey's HSD post-hoc, pair-wise comparisons was also carried out.

Since our study involved set of>2 sites and with large heterogeneity, I also assessed multiple site beta diversity (Baselga, 2013b) to quantify overall compositional heterogeneity of butterfly communities in 24 transects. Therefore, in order to test if average multiple site dissimilarity indices (β_{BRAY}, β_{SOR} and their components) differ significantly from random expectation, I computed dissimilarity values for total butterfly communities (and also based on their habitat specialization and larval host specificity) using a resampling procedure, taking 1000 random samples of 10 sites computing the average dissimilarity values for both abundance-based dissimilarity and incidence based dissimilarity (Baselga, 2010, 2017).

Finally, the butterfly community composition for total species, forest specialist, monophagous and protected species of different transects across four ecosystems were compared through NMDS using Bray-Curtis distance, on which environmental variables were fitted subsequently. I used Monte-Carlo randomization test with 999 permutations to assess the significance of environmental variables. Performance of NMDS was tested using Kruskal's stress formula multiplied by 100 (McCune and Grace, 2002).

5.3. Results

5.3.1. Butterfly species richness and diversity

In total, I recorded 8019 individuals, representing 268 species of butterflies belonging to six families, which constitute around 39% of total butterflies reported from Sikkim Himalaya (Haribal, 1992; Kunte, 2010). Out of these, more than two-third are forests specialists, one-third monophagous, and one-fifth protected species of conservation concern, signifying well representation of the three land use sensitive guilds (Table 5.1; Appendix G).

Table 5.1. Patterns of alpha diversity and other community parameters of butterflies along agroecosystem-forest gradient of Sikkim, Eastern Himalaya. The values were pooled for each ecosystem and also overall. The value of Chao 1 is mean ± standard deviation.

Community parameters	**Landscape gradient**				**Total**
	MOAS	**FAS**	**LCAS**	**Forest**	
Total species richness (Sobs)	187	180	157	170	268
Chao1	253.36 ±24.9	216.39 ±13.8	198.93 ± 16.40	233.38 ±21.99	299.97 ±12.19
Sobs: Chao1	73.8%	83.18%	78.92%	72.84%	89.34
Forest specialization					
Forest specialist	138	121	111	111	200
Forest generalist	49	55	46	51	68
Larval host specificity					
Monophagous	65	56	50	58	89
Polyphagous	92	102	80	83	124
Data deficient	30	22	27	29	55
Forest specialist and Monophagous	59	48	41	50	77
Conservation concern species	33	22	25	28	53
CITES Appendix II	2	1		2	2
WPA 1972 (Schedule I, II, IV)	31	21	25	26	51
India Red Data Book	6	1	4	4	9

Out of the total butterfly species observed, 92 species were common to all the four ecosystems, whereas 81 species exclusive to single ecosystem type (with 21 species in Forests against 60 species in different IFS). For the pooled data, total species richness and abundance was highest in MOAS followed by FAS, Forest and LCAS. Data on larval host specificity was available for only 213 butterfly species, of which polyphagous was~1.4 times richer than monophagous species (124 vs.89 species). The species richness of forest specialist was ~2.8 times higher than forest generalist butterflies (197 vs.71 species). The conservation concern butterfly species constitutes around one-fifth (53 species) of the total butterflies with two species protected under CITES Appendix II (*Troides aeacus aeacus*, *Troides helena cerberus*), and 51 species under WPA 1972 (Schedule I: 9 species, II: 33 species, and IV: 9 species) of which all 9 WPA 1972 Schedule I species are included in India Red Data Book of butterflies (Table 5.1).

The completion of butterfly sampling in the present study was adequate because individual-based rarefaction curve (including extrapolation) almost reached asymptote in all ecosystems except Forest which showed characteristic deceleration associated with an approach to asymptote (Fig. 5.1).

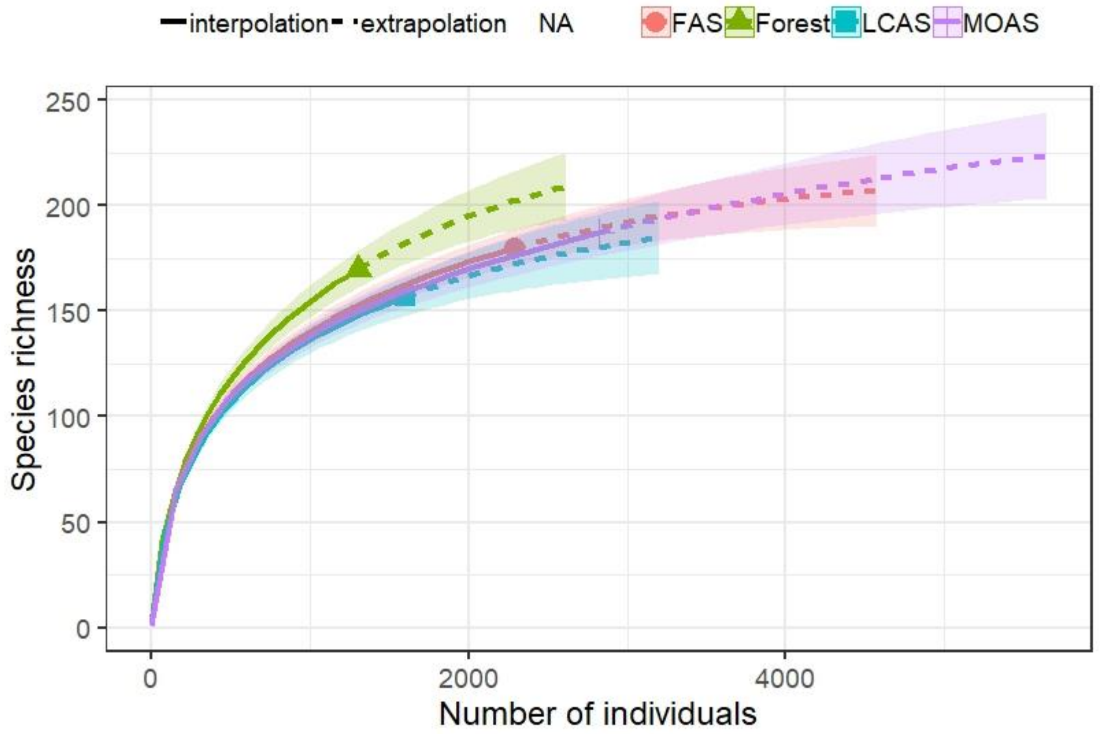


Fig. 5.1. Species accumulation curve for butterfly diversity (Species richness; q=0) along agroecosystem-forest gradient of Sikkim, Eastern Himalaya. Mandarin orange-based agroforestry systems (MOAS), farm-based agroforestry systems (FAS), large cardamom-based agroforestry systems (LCAS), and Natural forests (Forest).

5.3.2. Patterns and determinants of alpha diversity

Along the agroecosystem-forest-gradient, butterfly alpha diversity significantly declined from MOAS towards Forests (MOAS > LCAS > FAS > Forests) for total butterflies ($F_{3,\ 1756}$ = 38.71, p<0.001; Fig. 5.2a), forest specialist ($F_{3,\ 1756}$ = 29.12,

$p<0.001$; Fig.5.3b), monophagous ($F_{3,\,1756} = 26.56$, $p<0.001$; Fig. 5.2c) and protected species ($F_{3,\,1756} = 8.49$, $p<0.001$; Fig. 5.2d). Similar patterns were also observed for abundance per point for total butterflies ($F_{3,\,1756} = 35.22$, $p<0.001$; Fig. 5.2e), forest specialist ($F_{3,\,1756} = 25.25$, $p<0.001$; Fig. 5.2f), monophagous ($F_{3,\,1756} = 26.53$, $p<0.001$; Fig. 5.2g) and protected species ($F_{3,\,1756} = 8.08$, $p<0.001$; Fig. 5.2h). Shannon-Wiener diversity also showed declining patterns for the total butterflies (Fig. 5.2i) as well the three land use sensitive butterfly groups (Fig. 5.2j–l) but the relationship was statistically significant only for forest specialist ($F_{3,\,20} = 3.101$, $p<0.05$; Fig. 5.2j) and monophagous species ($F_{3,\,20} = 3.101$, $p<0.05$; Fig. 5.2k).

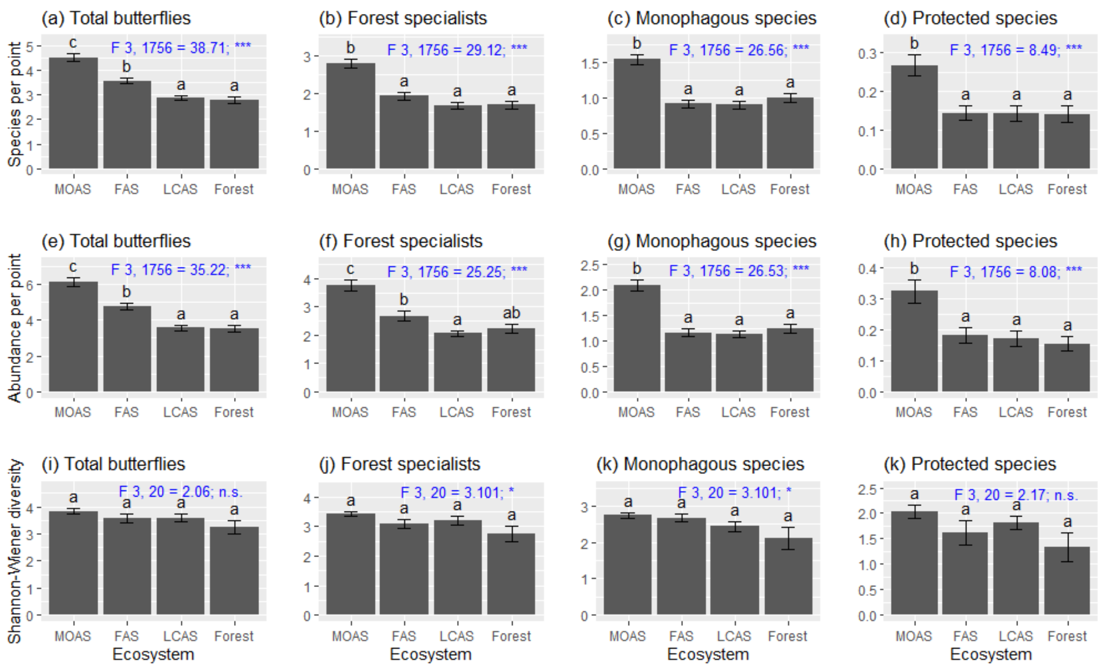


Fig. 5.2. Species per point, abundance per point and Shannon-Wiener diversity for total butterflies, forest specialist, monophagous and protected species along agroecosystem-forest gradient of Sikkim, Eastern Himalaya. Results of ANOVA test including significance (***: $p<0.001$; (*: $p<0.05$; n.s.: non-significant), and degree of freedom (between groups: 3 resulting from ecosystem types; within groups: 1756 or 20 resulting from point counts or transects) are also shown. For Bonferroni corrected post-hoc tests, bars not sharing letters are significantly different at the alpha level of $P = 0.05$. Mandarin orange-based agroforestry systems (MOAS), farm-based agroforestry systems (FAS), large cardamom-based agroforestry systems (LCAS), and Natural forests (Forest).

Bonferroni corrected post-hoc, pair-wise comparisons also revealed significant difference in alpha diversity and abundance per point for total butterflies and the three land-use sensitive guilds, also showed inwith higher values in MOAS compared to FAS, LCAS and Forest (all $p<0.01$). FAS also had significantly higher alpha diversity of total butterflies compared to LCAS and Forest (Bonferroni corrected post-hoc test: $p<0.05$ each), and also for abundance per point of forest specialist butterflies than LCAS ($p<0.05$: Bonferroni corrected post-hoc test). However, for the three land use sensitive butterfly groups, species per point, abundance per point and Shannon-Wiener diversity in Forests was comparable to FAS and LCAS (all $p>0.05$: Bonferroni corrected post-hoc test).

For the different butterfly families, the patterns of species per point (Fig. 5.3a-f) and abundance per point (Fig. 5.3g-l) showed significant differences but with marked variation along the agroecosystem-forest gradient. For Hesperiidae (Photo plate 5.1A,), species per point ($F_{3,\ 1756} = 6.757$, $p<0.001$; Fig. 5.3a) and abundance per point ($F_{3,\ 1756} = 7.18$, $p<0.001$; Fig. 5.3g) were both significantly higher in MOAS compared to LCAS and Forest (all $p<0.05$: Bonferroni corrected post-hoc test), but it did not differ between MOAS and FAS. For Papilionidae (Photo plate 5.1D), both species per point ($F_{3,\ 1756} = 22.74$, $p<0.001$; Fig. 5.3b) and abundance per point ($F_{3,\ 1756} = 18.7$, $p<0.001$; Fig. 5.3h) were significantly higher in MOAS compared to all other ecosystems including Forest (all $0.001<p<0.05$: Bonferroni corrected post-hoc test), but was significantly similar among Forest, FAS and LCAS. For Pieridae (Photo plate 5.1C), species per point ($F_{3,\ 1756} = 37.6$, $p<0.001$; Fig. 5.3c) and abundance per point ($F_{3,\ 1756} = 30.6$, $p<0.001$; Fig. 5.3i) were significantly higher in MOAS and FAS compared with both LCAS and Forest (all $0.001<p<0.05$: Bonferroni corrected), declining along the agroecosystem-forest gradient.

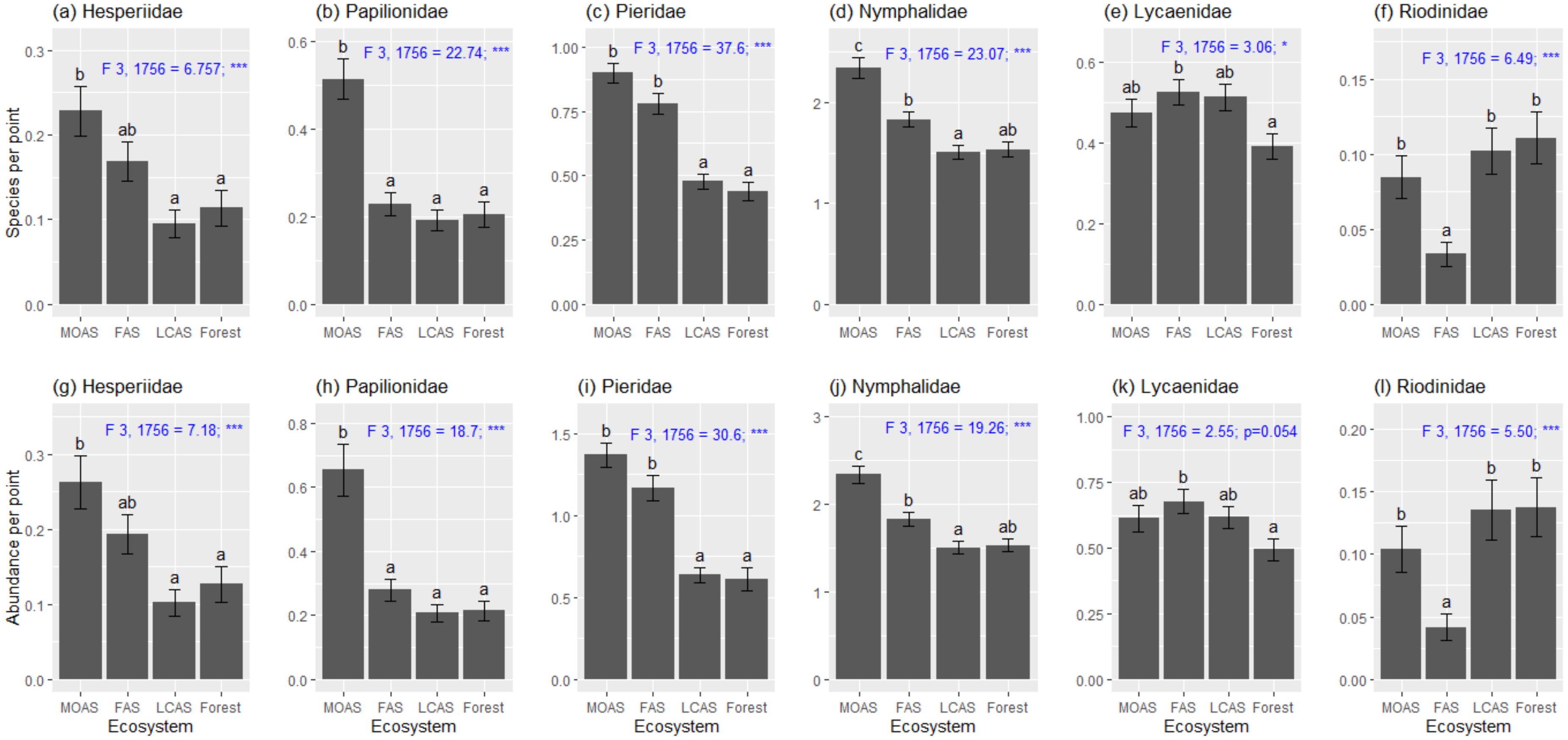


Fig. 5.3. Family wise patterns of species per point and abundance per point for the six butterfly families (Hesperiidae, Lycaenidae, Riodinidae, Nymphalidae, Pieridae and Papilionidae) along agroecosystem-forest gradient of Sikkim, Eastern Himalaya. Mandarin orange-based agroforestry systems (MOAS), farm-based agroforestry systems (FAS), large cardamom-based agroforestry systems (LCAS), and Natural forests (Forest). Results of ANOVA test including significance (***: p<0.001; n.s.: non-significant) degree of freedom are also shown. For Bonferroni corrected post-hoc tests, bars not sharing letters are significantly different at the alpha level of P = 0.05.

For Nymphalidae (Photo plate 5.1E), both species per point ($F_{3,\ 1756} = 23.07$, $p<0.001$; Fig. 5.3d) and abundance per point ($F_{3,\ 1756} = 19.26$, $p<0.001$; Fig. 5.3i) was significantly higher ($0.001<p<0.05$: Bonferroni corrected post-hoc test), in MOAS than all other study ecosystems, and also in FAS compared to LCAS. For Lycaenidae (Photo plate 5.1B), species per point ($F_{3,\ 1756} = 3.06$, $p<0.05$; Fig. 5.3e) and abundance per point ($F_{3,\ 1756} = 2.55$, $p=0.054$; Fig. 5.3k) differed significantly and near-significantly, respectively, with higher values in FAS than Forest only ($p<0.05$: Bonferroni corrected). For Riodinidae (Photo plate 5.1F), species per point ($F_{3,\ 1756} = 6.49$, $p<0.001$; Fig. 5.3f) and abundance per point ($F_{3,\ 1756} = 5.50$, $p<0.001$; Fig. 5.3l) was significantly higher (all $p<0.05$: Bonferroni corrected post-hoc test) in Forest, followed by LCAS and MOAS compared to FAS. When compared to Forest, the IFS supported significantly higher species and abundance per point for five butterfly families: Nympahlidae and Pieridae (in MOAS, FAS), Hesperiidae and Papilionidae (in MOAS), Lycaenidae (in FAS), broadly showing declining pattern along the agroecosystem-forest gradient. One the other hand, Riodinidae was significantly similar between Forest, LCAS and MOAS but lowest in FAS, and broadly increased towards Forest and LCAS.

Two-way ANOVA test showed significant effect of ecosystems and seasons, and their interactions on butterflies' species per point (Ecosystem: $F_{3,\ 1744}=41.42$, $p<0.001$; Season: $F_{3,\ 1744}=21.77$, $p<0.001$; Ecosystem:Season: $F_{3,\ 1744}=7.77$, $p<0.001$) and abundance per point (Ecosystem: $F_{3,\ 1744}=37.76$, $P<0.001$; Season: $F_{3,\ 1744}=22.45$, $p<0.001$; Ecosystem:Season: $F_{3,\ 1744}=7.92$, $p<0.001$). Along the temporal scale (winter to post monsoon), there was a clear increasing trend in richness and abundance of butterflies in Forests, but no clear trend in other ecosystems with highest values in MOAS and FAS during post monsoon but in LCAS during pre monsoon.

Along the agroecosystem-forest gradient, butterflies were significantly more speciose (Fig. 5.4A) and abundant (Fig. 5.4B) in MOAS compared to other two IFS and Forest during winter and monsoon seasons (Bonferroni corrected Post-hoc test: $0.001<p<0.05$). During pre-monsoon, significantly higher species per point was observed in FAS compared to Forest, whereas, higher abundance per point in FAS and LCAS compared to Forest ($0.001<p<0.05$) (Fig. 5.4A, B). However, during post-monsoon, Forests along with MOAS and FAS had significantly highest species per point and abundance per point compared to LCAS (Fig. 5.4A, B).

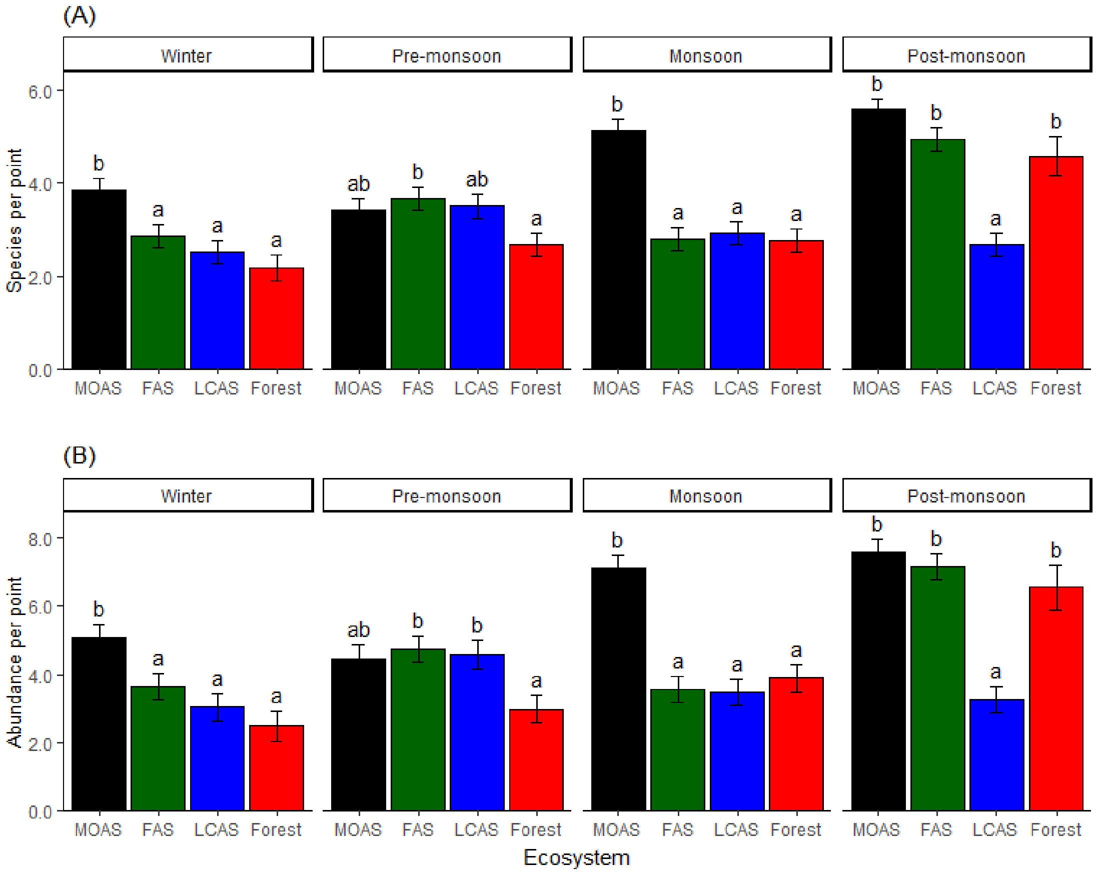


Fig. 5.4. Error bar showing the resilience of butterfly community, species per point (A) and abundance per point (B), across different seasons (winter, pre monsoon, monsoon and post monsoon) along agroecosystem-forest gradient of Sikkim, Eastern Himalaya. Mandarin orange-based agroforestry systems (MOAS), farm-based agroforestry systems (FAS), large cardamom-based agroforestry systems (LCAS), and Natural forests (Forest). For Bonferroni corrected post-hoc, pair-wise comparisons, bars not sharing letters are significantly different at the alpha level of $p = 0.05$.

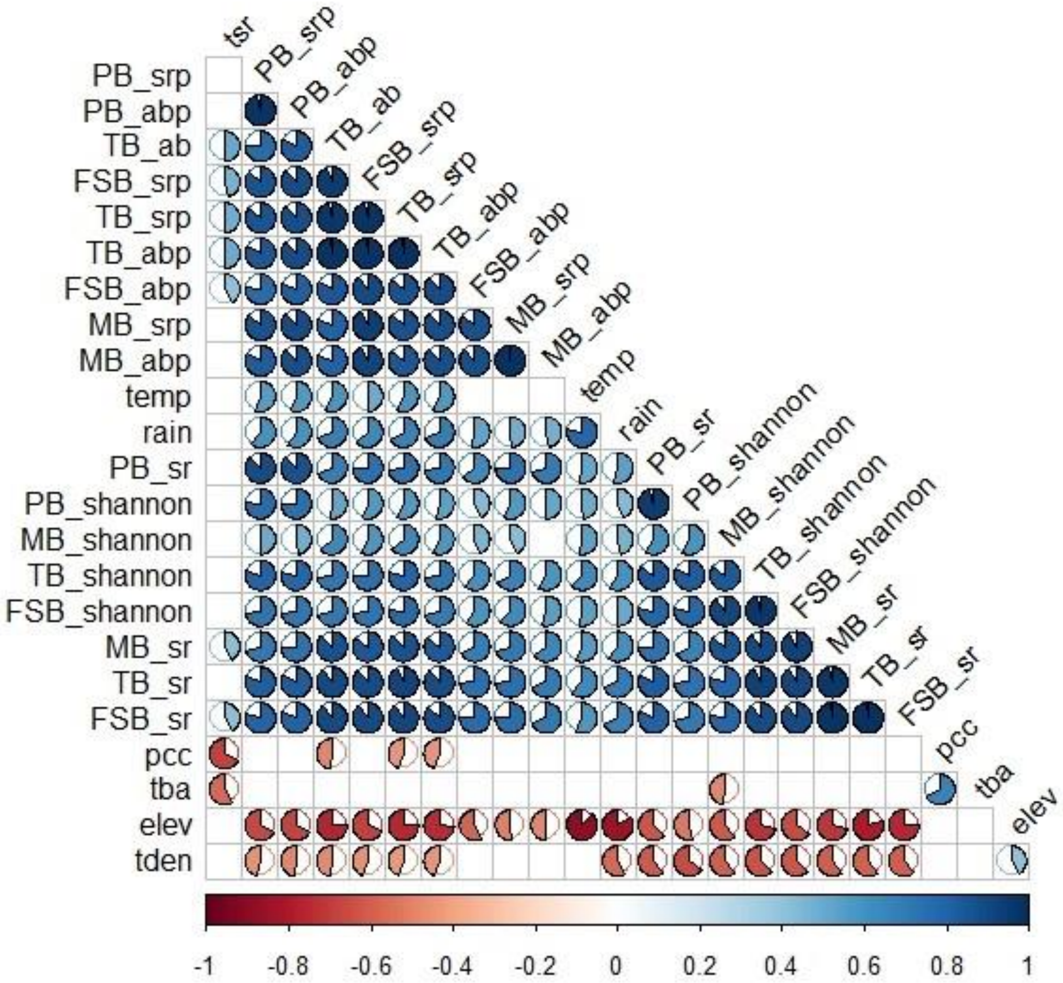


Fig. 5.5. Correlation matrix showing significant ($p<0.05$; $p<0.01$) and strong ($r>0.45$) correlation coefficients between the butterfly diversity measures and predictor variables along the agroecosystem-forest gradient of Sikkim, Eastern Himalaya. The size and colour (-1: dark red to +1: dark blue) of the pie in the matrix code for correlation strength and direction. The habitat and environmental variables are: elevation (elev), mean annual precipitation(rain), mean annual temperature (temp), percentage canopy cover (pcc), tree species richness (tsr), tree density (tden), and tree basal area (tba); the butterfly diversity parameters has two parts separated by underscore, first part in upper-case letters represents the butterfly guilds: total butterflies (TB), forest specialist butterflies (FSB), monophagous butterflies (MB), protected butterflies (PB); and second part represents their diversity parameters, species richness per point (srp), abundance per point (abp), species richness (sr), abundance (ab), Shannon-Wiener diversity (shannon).

Table 5.2. Model averaged estimates and their unconditional standard errors for covariates (best models withΔAICc<4) based on species per point (6 models) and abundance per point (4 models) of total butterfly communities along the agroecosystem- forest gradient of Sikkim, Eastern Himalaya. MAP: Mean Annual Precipitation, pcc: percentage canopy cover, Mandarin orange-based agroforestry systems (MOAS), large cardamom-based agroforestry systems (LCAS), and Natural forests (Forest).

	Estimate	Standard error	z value	Pr(>|z|)	Covariate weight
Species per point					
(Intercept)	1.188	0.050	23.967	< 2e-16 ***	-
SeasonPost Monsoon	0.190	0.042	4.546	5.50e-06 ***	1.00
SeasonPre Monsoon	-0.023	0.043	0.531	0.59509	1.00
SeasonWinter	-0.184	0.044	4.229	2.35e-05 ***	1.00
EcosystemForest	0.076	0.109	0.695	0.48717	0.07
EcosystemLCAS	0.111	0.102	1.084	0.27821	0.07
EcosystemMOAS	0.142	0.094	1.514	0.13001	0.07
Elevation	-0.206	0.041	4.984	6.00e-07 ***	1.00
pcc	0.010	0.048	0.207	0.83627	0.22
Tree basal area	0.010	0.042	0.228	0.81988	0.22
Tree density	-0.082	0.039	2.133	0.03292 *	0.87
Tree species richness	0.122	0.042	2.885	0.00391 **	1.00
Abundance per point					
(Intercept)	1.452	0.060	24.223	< 2e-16 ***	-
SeasonPost Monsoon	0.211	0.047	4.476	7.6e-06 ***	1.00
SeasonPre Monsoon	-0.068	0.047	1.430	0.15271	1.00
SeasonWinter	-0.242	0.048	5.043	5.0e-07 ***	1.00
EcosystemForest	-0.002	0.129	0.012	0.99027	0.12
EcosystemLCAS	0.175	0.121	1.440	0.14983	0.12
EcosystemMOAS	0.192	0.099	1.940	0.05238	0.12
Elevation	-0.217	0.043	5.033	5.0e-07 ***	0.60
MAP	0.256	0.048	5.376	1.0e-07 ***	0.40
pcc	-0.032	0.069	0.466	0.64092	0.32
Tree basal area	0.021	0.058	0.368	0.71316	0.24
Tree density	-0.092	0.043	2.117	0.03430*	0.66
Tree species richness	0.153	0.048	3.186	0.00144**	1.00

Results from correlation tests (Fig. 5.5) and GLMMs (Table 5.2, Appendices H-I) identified similar determinants of butterfly alpha diversity and abundance per point. Butterfly alpha diversity and abundance per point showed significant positive relationship with tree species richness but significant negative relationship with tree density and percentage canopy cover, however, effect was non-significant negative for basal area. Among the different environmental variables, butterfly alpha diversity and abundance per point showed highly significant strong relationship with effect being negative for elevation but positive for MAT and MAP. For butterfly alpha diversity, six models with ΔAICc < 4 were identified as best models by GLMMs and based on model-averaging (best models with ΔAICc < 4) four covariates (season, elevation, tree species richness and tree density) had significant and strong effect. For butterfly abundance per point, GLMMs identified 12 models with ΔAICc < 4 as best models and model-averaging revealed five covariates (season, elevation, MAP, tree species richness and tree density) showing significant and strong effect.

5.3.3. Patterns and determinants of beta diversity

Nighty-two butterfly species (34.33%, 6820 individuals) were common to four ecosystems, whereas, 81 species (30.22%, 145 individuals) were exclusively observed in single ecosystem type with 60 species in the IFS (MOAS: 24, FAS: 21 and LCAS: 15 species) vs. 21 species in Forest. I observed significantly highest pair-wise beta diversity in Forest (βbray=0.4642; βsor=0.4295) for both abundance-based dissimilarity ($F_{3, 20} = 3.64$; $p<0.05$) and incidence-based dissimilarity ($F_{3, 20} = 5.16$; $p<0.01$), which declined towards MOAS (βbray=0.2992; βsor=0.2505) through FAS and LCAS (Fig. 5.6a, b). Post-hoc, pair-wise comparisons with Tukey's HSD revealed significant difference for pair-wise beta diversity of butterflies based on both the dissimilarity indices ($p<0.05$ for βbray; $p<0.01$ for βsor) between MOAS and Forest (Fig. 5.6a, b) and marginally significant difference between LCAS and Forest ($0.05<p<0.1$).

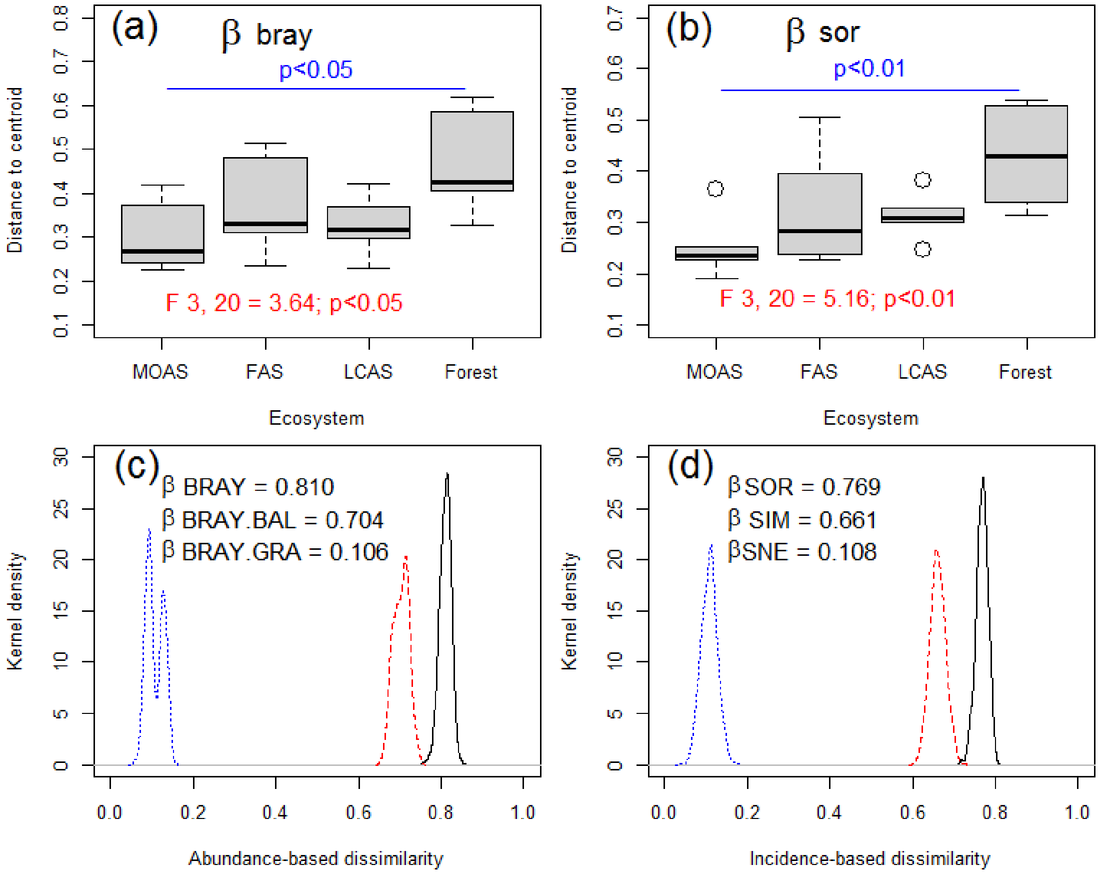


Fig. 5.6. Beta diversity of butterflies along agroecosystem-forest gradient of Sikkim, Eastern Himalaya. The pair-wise beta diversity, (a) βbray, (b) βsor are shown by distance to centroid; and multiple-site beta diversity (computed by resampling across 1000 samples of 10 transects in different ecosystems) shown by kernel density for (c) Bray-Curtis index, solid line: β_{BRAY}, dashed line: $\beta_{BRAY.BAL}$, and dotted line: $\beta_{BRAY.GRA}$; and (d) Sorensen index, solid line: β_{SOR}; dashed line: β_{SIM}; and dotted line: β_{SNE}. Mandarin orange-based agroforestry systems (MOAS), farm-based agroforestry systems (FAS), large cardamom-based agroforestry systems (LCAS), and Natural forests (Forest). Results of ANOVA test including degree of freedom (3: between group, 20: within group resulting from Ecosystems and Transects, respectively), significance level, and Tukey's HSD post-hoc, pair-wise comparisons are also given.

I found very high overall multiple-site beta diversity (β_{BRAY}=0.898; β_{SOR}=0.876) dominantly contributed by the substitution components ($\beta_{BRAY.BAL}$=0.830; β_{SIM}=0.806) than nestedness components ($\beta_{BRAY.GRA}$ =0.068; β_{SIM}=0.069) for both dissimilarity indices. Simulations of multiple-site beta diversity and their components based on the two multiple-site dissimilarity indices resulted in slight decline in overall (β_{BRAY}=0.810; β_{SOR}=0.768), substitution components ($\beta_{BRAY.BAL}$=0.704; β_{SIM}=0.660) but increase in nestedness components ($\beta_{BRAY.GRA}$=0.106; β_{SNE}=0.107) when compared to original values. However, simulated multiple-site beta diversity was still highly dominated by substitution component in abundance-based (Fig. 5.6c) and incidence-based (Fig. 5.6d) multiple-site dissimilarity indices.

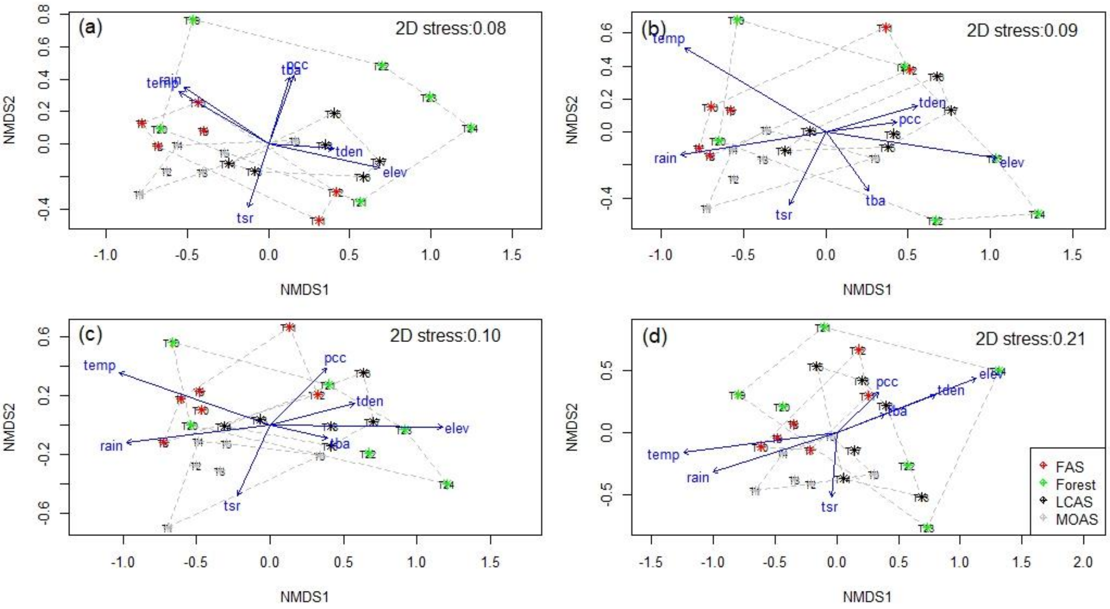


Fig. 5.7. NMDS of butterfly species composition along agroecosystem-forestgradient of Sikkim, Eastern Himalaya: sites with convex hull volume having overlaid environmental parameters (vectors as arrows) for total butterflies (a), forest specialist (b), monophagous (c) and protected species (d). Further information for ordination statistics is given in Appendix J. Mandarin orange-based agroforestry systems (MOAS), farm-based agroforestry systems (FAS), large cardamom-based agroforestry systems (LCAS), and Natural forests (Forest).

Based on NMDS, butterfly community assemblages were significantly determined by ecosystems for total species (Fig. 5.7a), forest specialist (Fig. 5.7b) and monophagous (Fig. 5.7c) but not for protected species (Fig. 5.7d). Forest and LCAS transects were clearly characterized by higher tree density, elevation, tree basal area, percentage canopy cover but lower tree species richness, MAT, and MAP. In contrast, MOAS and FAS transects were characterized by higher MAT, MAP and tree species richness, but lower elevation, percentage canopy cover, tree density and tree basal area (Fig. 5.7a, c; also see Appendix J).

5.4. Discussion

5.4.1. Alpha diversity

In the present study, species richness, Shannon-Wiener diversity, alpha diversity, abundance per point for both the total butterflies as well as the three land-use change sensitive guilds were higher (in MOAS) or comparable (in FAS and LCAS) to Forests. The present observation of high species richness, Shannon-Wiener diversity and proportion of total species richness of butterflies is comparable to or greater than the previous studies in agroecosystems or forest from the biodiversity hotspots of the Himalaya, Indo-Burma, Western Ghats and Sri Lanka, and elsewhere (Kunte *et al.*, 1999; Bobo *et al.*, 2006; Barlow *et al.*, 2007b; Dolia *et al.*, 2008; Kitahara *et al.*, 2008; Bhardwaj *et al.*, 2012; Ekroos *et al.*, 2013; Acharya and Vijayan, 2015; Chettri, 2015; Myers *et al.*, 2015; Chettri *et al.*, 2018b; Šálek *et al.*, 2018; Dewan *et al.*, 2019) (Fig. 5.8; Table 5.3; Appendix K).

For butterflies' total species richness (from studies reviewed in the present study), study region had marginally significant effect, whereas, ecosystem and the interaction (region*ecosystem) term showed non-significant effects (See Table 5.3 for Two-way ANOVA results). However, studies from Himalaya and Indo-Burma reported upto ~1.5 to ~2 times higher average values compared to other two regions. The observed total species richness of butterflies in different IFS and Forest of Sikkim was greater than the median (Fig. 5.8a) and mean (Table 5.3) values from the Himalaya and other regions reviewed in the present study.

For butterflies' Shannon-Wiener diversity, the effect was significant by region, marginally significant by ecosystem but non-significant by interactions term (See Table 5.3 for Two-way ANOVA results). Across regions, Shannon-Wiener diversity showed significant difference only in case of the Forest Ecosystems (but not for Agroecosystems), with mean values significantly higher in Himalaya and Indo-Burma (both $p<0.05$: Bonferroni corrected post-hoc test) or marginal significantly higher in Western Ghats & Sri Lanka ($p=0.063$) compared to elsewhere. Between ecosystems, Shannon-Wiener diversity differed significantly only for elsewhere where Agroecosystems had on average >2 times higher values than Forests. Butterflies' Shannon-Wiener diversity in different IFS and Forest of Sikkim was greater than or comparable to the median (Fig. 5.8b) and mean (Table 5.3) values from Himalaya and other regions reviewed in the present study.

For proportion of total species richness of butterflies, ecosystem had significant effect, region had non-significant and interactions term showed marginally significant effect (results of two-way ANOVA and Bonferroni corrected post-hoc test are provided in Table 5.3). However, the mean values were significantly higher in Agroecosystems compared to Forests for elsewhere ($p<0.05$), but were significantly similar across ecosystems in the three biodiversity hotspots. On the other hand, the mean values were not significantly different between study regions in both Agroecosystems and Forests. The IFS (specially MOAS and FAS) and Forest of Sikkim showed greater or comparable proportion of total species richness of butterflies than median (Fig. 5.8c) and mean (Table 5.3) values in the Himalaya (for both ecosystems), Western Ghats & Sri Lanka (in both ecosystems), Indo-Burma (in Agroecosystems), and elsewhere (for Forests).

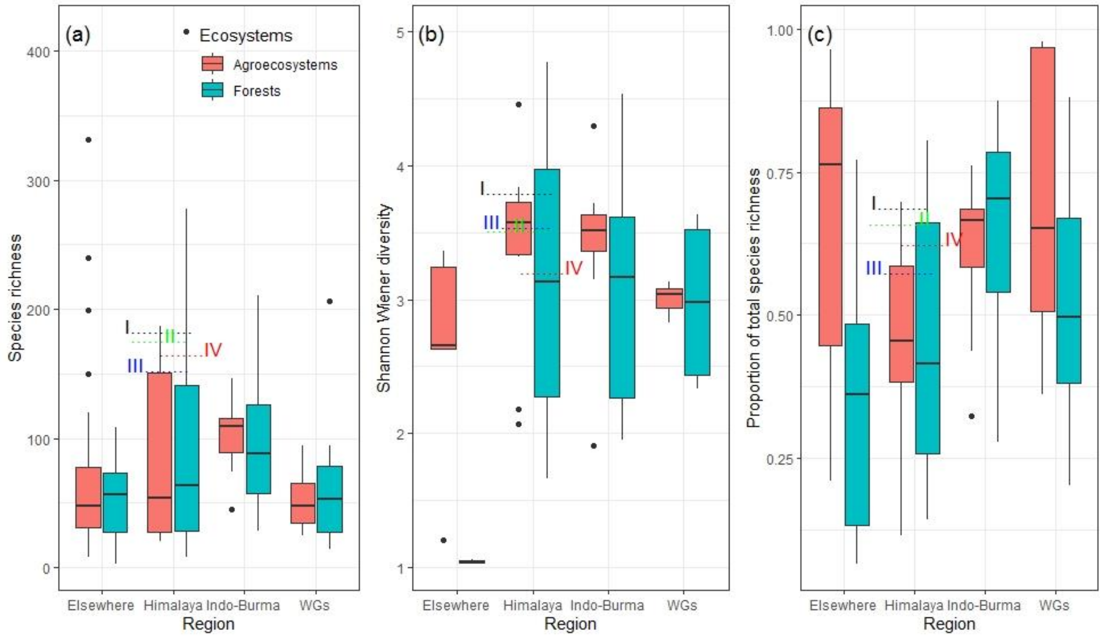


Fig. 5.8. Comparison of total species richness (a), Shannon-Wiener diversity (b), proportion of total species richness (c) of butterflies in different ecosystems (I: MOAS; II: FAS; III: LCAS; IV: Forest) of Sikkim, Eastern Himalaya, India with those reported from the biodiversity hotspots of Himalaya, Indo-Burma, Western Ghats and Sri Lanka (WGs) and Elsewhere. Mandarin orange-based agroforestry systems (MOAS), farm-based agroforestry systems (FAS), large cardamom-based agroforestry systems (LCAS), and Natural forests (Forest).

The highest butterfly alpha diversity and abundance per point in MOAS could be attributed to higher tree species richness, diversity and density in the traditionally managed agroecosystems of the Himalaya (Sharma and Vetaas, 2015; Sharma *et al.*, 2016b). Secondly, MOAS, among the three IFS, provide the most suitable habitat for butterflies since it has shade trees, involves intercropping of pulses, vegetables, and fruits, providing larval hosts and adult nectar even during winter when mandarin orange is in fruiting/ripening stage and harvested. The combination of all these positive land-use properties makes the butterflies and, consequently, the provision of pollination service more resilient in MOAS.

Table 5.3: Results of two-way ANOVA, summary statistics and Bonferroni corrected post-hoc, pair-wise comparisons of total species richness, Shannon-Wiener diversity and proportion of total species richness of butterflies in the biodiversity hotspots of the Himalaya (including from present study), Indo-Burma, Western Ghats and Sri Lanka (WGs) and Elsewhere. Mean ±SE values not sharing letters across regions (vertically: a, b) and ecosystems (horizontally: X, Y) are significantly different at the alpha level of P = 0.05; Bonferroni corrected. Df: Degree of freedom; sum sq.: Sum of squares.

Diversity parameter	**Two-way ANOVA test**						**Ecosystems**	
	Factor	**df**	**Sum sq.**	**Estimate**	**P-value**	**Regions**	**Agroecosystems**	**Forests**
Total species richness	Regions	3	31372	2.167	p=0.096	Himalaya	86.6±22.0 a; X	98.1±13.6 a; X
	Ecosystems	1	12	0.003	p=0.96	Indo Burma	**101.5±24.6** a; X	**101.8±20.1** a; X
	Regions: Ecosystems	3	5007	0.346	p=0.79	Western Ghats & Sri Lanka	53.1±24.6 a; X	68.1±24.6 a; X
	Residuals	108	521107			Elsewhere	71.4±12.1 a; X	51.8±20.9 a; X
						Present study: MOAS	187	170 (in Forest)
						. FAS	180	
						LCAS	157	
Shannon Wiener diversity	Regions	3	6.62	**3.110**	**p<0.05**	Himalaya	**3.38±0.27** a; X	**3.13±0.21** b; X
	Ecosystems	1	2.16	3.041	p=0.09	Indo Burma	**3.39±0.30** a; X	**3.04±0.25** b; X
	Regions: Ecosystems	3	2.37	1.112	p=0.35	Western Ghats & Sri Lanka	3.00±0.49 a; X	2.98±0.42 ab$^{\#}$; X
	Residuals	51	36.18			**Elsewhere**	2.62±0.38 a; Y	1.04±0.60 a; X
						Present study: MOAS	3.84	3.25 (in Forest)
						. FAS	3.56	
						LCAS	3.59	
Proportion of total species richness	Regions	3	0.308	1.908	p=0.14	Himalaya	0.45±0.08 a; X	0.46±0.07 a; X
	Ecosystems	1	0.234	**4.355**	**p<0.05**	Indo Burma	0.61±0.08 a; X	**0.64±0.09** a; X
	Regions: Ecosystems	3	0.367	2.276	p=0.09	Western Ghats & Sri Lanka	**0.69±0.10** a; X	0.53±0.10 a; X
	Residuals	65	3.496			**Elsewhere**	0.65±0.06 a; Y	0.35±0.08 a; X
						Present study: MOAS	0.70	0.63 (in Forest)
						. FAS	0.67	
						LCAS	0.59	

The observed dominance of butterfly community in monsoon and post monsoon season across the four systems is in accordance to the general trend in the Himalayan region (Sengupta *et al.*, 2014; Acharya and Vijayan, 2015; Chettri, 2015; Singh *et al.*, 2015) and elsewhere (Kunte, 1997; Kunte *et al.*, 1999). The flight period of almost all species of butterflies (259–262 species) in the present study fell during pre monsoon to post monsoon (through monsoon) seasons against 144 species in winter (Appendix G). These butterflies might have synergized their life cycle and flight period pattern with abundant resource availability in the habitat as larval host and adult food plants during pre monsoon to post monsoon seasons compared to their decline in winter when most of them enters diapause or migrates out (Wynter-Blyth, 1957; Sengupta *et al.,* 2014).

Butterfly alpha diversity and abundance per point had significant and strong positive relationship with tree species richness, MAT, MAP and strong negative relationship with elevation, tree density, and percentage canopy cover, whereas tree basal area had weak and non-significant effect. Similarly, based on GLMMs, I found significant and strong effect of season, elevation, tree species richness and tree density on both butterfly alpha diversity and abundance per point, and MAP on abundance per point but weak and non-significant response to tree basal area and percentage canopy cover. Influence of various environmental variables on butterfly communities have commonly been reported, for example, plant diversity and richness (Bhardwaj *et al.*, 2012; Ekroos *et al.*, 2013; Loos *et al.*, 2014), tree species richness and density (Chettri, 2010; Bobo *et al.*, 2006; Acharya and Vijayan, 2015), tree basal area (Barlow *et al.*, 2007b), canopy cover and canopy openness (Barlow *et al.*, 2007b; Dolia *et al.*, 2008; Bhardwaj *et al.*, 2012; Loos *et al.*, 2014; Chettri, 2015), natural forest remnant and % primary forest (Barlow *et al.*, 2007b). The highest alpha diversity and abundance per point in MOAS can thus be explained due to the occurrence of highest tree species richness, but lowest tree density, tree basal area and percentage canopy cover (detailed in Fig. 4.1 of Chapter 4).

I found elevation to be the most crucial determinants showing negative effect on butterfly alpha diversity and abundance per point. The decline in butterfly diversity along the elevational gradient was reported from the Himalayan region (Uniyal, 2007; Acharya and Vijayan, 2015; Chettri, 2015; Dewan *et al.*, 2021). Other topographical and landscape variables, e.g., landscape heterogeneity (Kumar *et al.*, 2009; Ekroos *et al.*, 2013; Myers *et al.*, 2015; Perović *et al.*, 2015), primary forests in the surrounding landscape (Barlow *et al.*, 2007b) also determines the butterflies' richness and community composition in different ecosystems, as observed in the present study. Different climatic variables, e.g., temperature and precipitation (Acharya and Vijayan, 2015) and season (Kunte, 1997; Barlow *et al.*, 2007b; Bhardwaj *et al.*, 2012) are also known to influence the diversity trend and population fluctuation in butterflies.

5.4.1.1. Forest specialist, monophagous and protected species

I observed high diversity of specialist (forest specialist, monophagous) and protected species in both Forests and different IFS, which could be due to organic and traditional management practices of agroecosystems in Sikkim. The traditional and organically managed agroecosystems retain high butterfly diversity including forest specialist, monophagous and conservation concern species comparable to or even greater than forest ecosystems (Mas and Dietsch, 2003; Bobo *et al.*, 2006; Bubova *et al.*, 2015). Additionally, it might be because of lesser impacts of non-commercial small farms on biodiversity than commercial large scale farms (Konvicka *et al.*, 2016;

Šálek *et al.*, 2018) due to small per capita landholding in Sikkim (Bhutia, 2015), and mosaic landscapes of agroecosystems with forests (Villemey *et al.*, 2015).

Landscape simplification filters species trait and drive biotic homogenization of butterfly community composition, gradually replacing the specialists by generalist species (Börschig *et al.*, 2013; Gámez-Virués *et al.*, 2015). The plant-abundance relationship is mediated by traits of butterflies and is strongest for monophagous, less mobile and habitat specialist species (Curtis *et al.*, 2015; Dainese *et al.*, 2017). The monophagous butterflies are highly specialized for host plants and are mostly forest specialist (e.g., 86.21% monophagous species observed in Forests and 82–90.77% in the IFS were forest specialist in the present study). Along the land use gradient, plant richness generally declines (Foody and Cutler, 2003) but Himalayan traditional agroecosystems reportedly retain high tree diversity and density (Sharma and Vetaas, 2015; Sharma *et al.*, 2016b). In both forests and agroecosystems, persistence of endemic, monophagous, forest specialist, resident and threatened butterflies depend on native vegetation with vertical structural complexity, and responds positively when plant community is dominated by native and endemic species (Barlow *et al.*, 2007b) but negatively with exotic plants (Dolia *et al.*, 2008). The seasonal dynamics in butterfly communities, as observed in the present study, can be closely linked with the plant phenological behaviour (Barlow *et al.*, 2007b) and availability of larval host (Sengupta *et al.*, 2014; Curtis *et al.*, 2015; Dainese *et al.*, 2017) and adult nectar plants (Kitahara *et al.*, 2008; Jain *et al.*, 2016).

5.4.2. Beta diversity

The pair-wise beta diversity increased significantly from MOAS to Forests through FAS and LCAS. However, the increase was not a steep one as would generally occur in intensive agriculture (Ekroos *et al.*, 2010). In butterfly community, landscape simplification associated with land use change and agricultural intensification has been reported to diminish beta diversity (Börschig *et al.*, 2013; Gámez-Virués *et al.*, 2015; Dainese *et al.*, 2017). Such patterns of increasing beta diversity along agroecosystems-forest gradient has been commonly reported from India (Kunte *et al.*, 1999) and elsewhere (Bobo *et al.*, 2006; Ekroos *et al.*, 2010; Schulze *et al.*, 2010; Francesconi *et al.*, 2013). Higher spatial and temporal beta diversity has been reported for butterflies from agroecosystems that are managed traditionally (Uchida and Ushimaru, 2015) and in forest canopy than understorey (Fordyce and DeVries, 2016). However, multiple site beta diversity was highly dominated by the substitution component for both the total as well as specialist butterflies. Dominance of substitution components implies that the conservation measures for butterflies must focus on all the ecosystems types rather than only on biodiverse system (Dobrovolski *et al.*, 2012). The dominance of substitution component in beta diversity indicates the presence of heterogeneous landscape driven by organic and traditionally managed agricultural practices in Sikkim Himalaya. Such patterns has also been reported in heterogeneous landscapes for butterflies (Uchida and Ushimaru, 2015; Baselga, 2017; Dainese *et al.*, 2017; Pereira *et al.*, 2017) and other faunal taxa (Soininen *et al.*, 2018). The organic and traditionally managed agroecosystem can mitigate the biotic homogenization associated with land use change and agricultural intensification. The relatively higher importance of substitution components of beta diversity for specialist and nestedness components for generalist butterflies can be explained by low dispersal ability of specialists (that requires undisturbed habitat and depends on specialized host plant) than generalists (which can survive in disturbed habitats and have wide range of host plants)

(Dobrovolski *et al.*, 2012). Beta diversity of butterflies is primarily structured by resource availability in the form of larval host and adult nectar plants (Mac Nally *et al.*, 2004) and is coupled with high species and phylogenetic turnover of plant communities (Acharya *et al.*, 2011b; Kemp *et al.*, 2017; Manish *et al.*, 2017; Manish and Pandit, 2018; Shooner *et al.*, 2018).

The butterfly community assemblage based on NMDS were significantly determined by ecosystems for total species, forest specialist and monophagous (but not for protected species), with species compositions in two IFS (MOAS, FAS) clustered together different from Forest and LCAS. The butterfly community composition was significantly structured by all seven habitat and environmental vectors (tree species richness, tree density, percentage canopy cover, tree basal area, elevation, MAT and MAP) for total species but only elevation, MAT, MAP and tree density for the land use sensitive sub-groups of butterflies. Beta diversity in butterflies depend on plant species richness (Kemp *et al.*, 2017), understorey coverage (Pereira *et al.*, 2017), canopy openness (Pereira *et al.*, 2017), basal area (Barlow *et al.*, 2007b). It also varies along spatial scale (Fordyce and DeVries, 2016) and season or temporal scale (Uchida and Ushimaru, 2015; Fordyce and DeVries, 2016), thus explaining the spatio-temporal variation in butterfly beta diversity and species composition. In this study, the variability in the habitat and environmental variables (Fig. 4.1, Table 4.2 detailed in chapter 4; Appendix A) among the ecosystems led to high landscape and habitat heterogeneity (Myers *et al.*, 2017) which also enhanced butterfly beta diversity and species composition. Therefore, mosaic landscape of Sikkim Himalaya (outside the PAs network) sustained high taxonomic alpha and beta diversity and community composition in butterflies.

5.5. Conclusion

The agricultural environments are contrasting challenges in using: the imperative for intensifying food production for the growing human population, on one hand, and the need for protecting biodiversity and ES within them, on the other (Foley *et al.*, 2011). There are efforts to improve the current agricultural policies in many countries, such as the European Common Agricultural Policy (CAP) (Zingg *et al.*, 2018) and organic farming in Sikkim in India (Bhutia, 2015). This study underlines the necessity to promote new greening measures for protection of biodiversity rich areas in agricultural landscapes and subsequent management requirements (Pe'er *et al.*, 2016).

The study has highlighted the importance of organic and traditionally managed agricultural landscapes of the eastern Himalayan region for sustaining high taxonomic alpha and beta diversity, and species composition of butterflies, including specialists and conservation concern species, as well as the provision of pollination service. They can broadly complement biodiversity conservation of PAs in tropical and subtropical zones (<1500m) experiencing high anthropogenic pressure but low PA and forest coverage. However, primary forests are irreplaceable for conservation and persistence of biodiversity (Gibson *et al.*, 2011). Study on functional and phylogenetic diversity of butterfly and other taxa will provide much deeper understanding on importance of these ecosystems in terms of conservation. Himalayan biodiversity hotspot is threatened by continuing habitat loss, forest fragmentation, land use change and agricultural intensification, expanding urbanization and developmental activities. Hence, there is urgent need to fully implement sustainable development model and move into traditional and organic management of agriculture, as already being practiced in Sikkim, to preserve its rich biodiversity and the provision of ES.

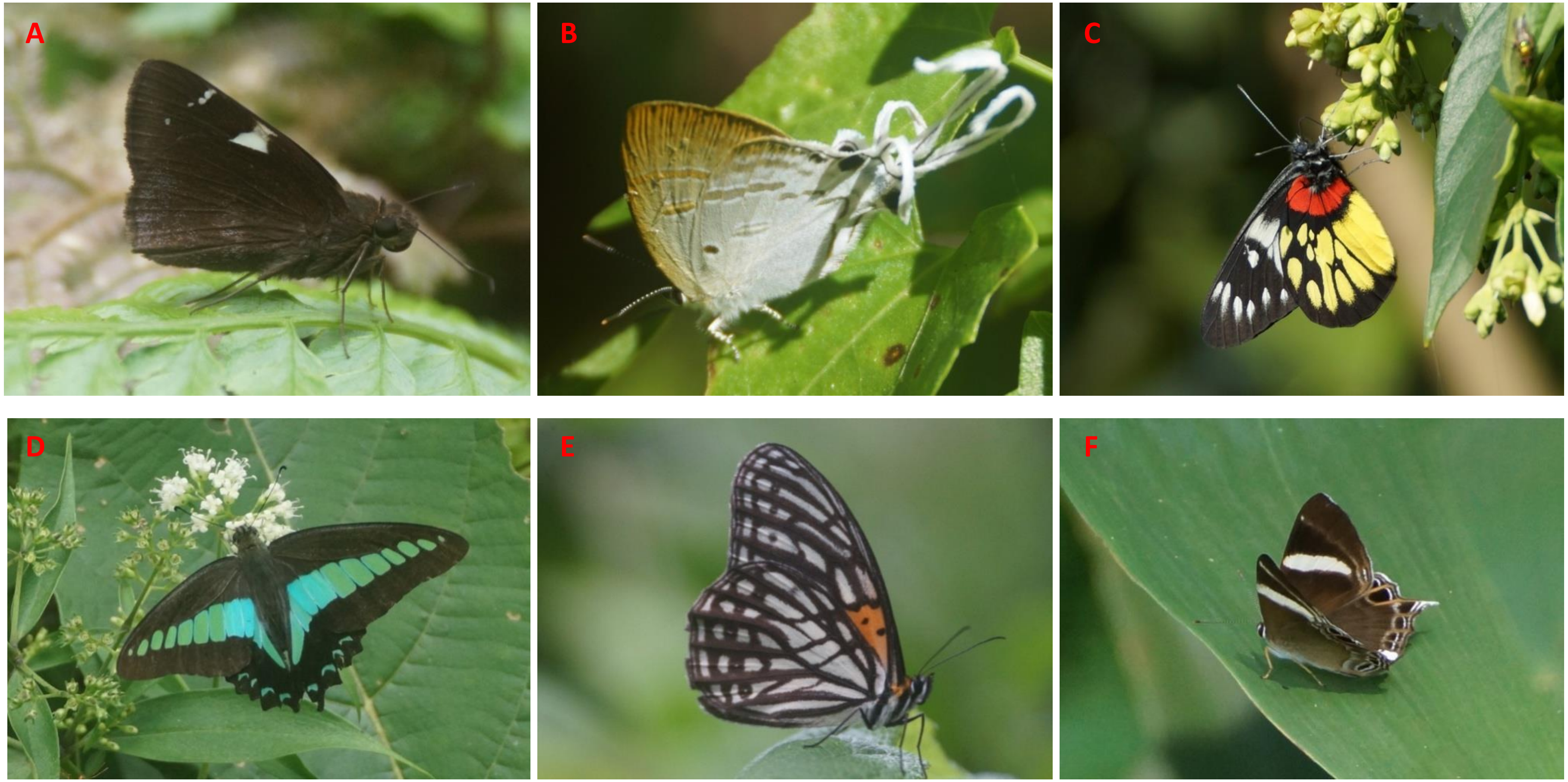


Photo plate 5.1. Representative butterfly species from the different families observed in IFS and adjoining Forest ecosystems of Sikkim, Eastern Himalaya. Hesperiidae: Spotted Demon *Notocrypta fiesthamelii alysos* Moore, 1865 (A); Lycaenidae: Fluffy Tit *Zeltus amasa amasa* Hewitson, 1865 (B); Pieridae: Red-base Jezebel *Delias pasithoe pasithoe* Linnaeus, 1767(C); Papilionidae: Common Bluebottle *Graphium sarpedon sarpedon* Linnaeus, 1758 (D); Nymphalidae: Tigerbrown *Orinoma damaris* Gray, 1846 (E); Riodinidae: Tailed Judy *Abisara neophron neophron* Hewitson, 1866 (F).

Chapter 6

FUNCTIONAL DIVERSITY AND TRAIT COMPOSITION OF BIRDS AND BUTTERFLIES ALONG AGROECOSYSTEM-FOREST GRADIENT

Red-tailed Minla *Minla ignotincta*

6.1. Introduction

The dichotomy between biodiversity conservation and, unabated habitat destruction and unsustainable agricultural intensification has attracted much attention and research focus in the recent decades (Gibbs *et al.*, 2010; Tscharntke *et al.*, 2012; Laurance *et al.*, 2014). Researchers have highlighted the inadequacy of PAs, and consequently failure of land-sparing conservation approach to fully secure the global biodiversity and associated ES against land use and climate change (Singh, 1999; Watson *et al.*, 2014; Coad *et al.*, 2019; Velazco *et al.*, 2019). Such findings has led to the recognition of complementary role of land-sharing approach in biodiversity conservation in different parts of the world (Fischer *et al.*, 2008; Chazdon *et al.*, 2009; Michael *et al.*, 2016; Grass *et al.*, 2019). Recently, there is a shift in studies on the effect of land-use and climate change on biodiversity from TD and community composition to FD and functional composition (Tilman, 2001; Mouchet *et al.*, 2010; Cadotte *et al.*, 2011; Hanspach *et al.*, 2015; Lindenmayer *et al.*, 2015; Correia *et al.*, 2020). The FD and functional composition can most appropriately represent the biodiversity-ecosystem-function (BEF) relationships, and are better suited to understand community assembly rules (environmental filtering, limiting similarity/biotic filtering, neutral theory) and consequences of biodiversity loss due to their high sensitivity to land-use change and intensification (Hausner *et al.*, 2003; Flynn *et al.*, 2009, Devictor *et al.*, 2010; Tinoco *et al.*, 2018; Goded *et al.*, 2019; Matuoka *et al.*, 2020). The limiting similarity in various traits results in higher FD values, whereas environmental filtering leads to lower FD values than random assembly (Mouchet *et al.*, 2010; Ding *et al.*, 2013). Numerous studies across the world has reported spatial mismatch or varied congruence in the patterns and determinants of FD with TD in diverse taxa such as birds, butterflies and plants

(Mayfield *et al.*, 2005; Flynn *et al.*, 2009; Monnet *et al.*, 2014; Aguirre-Gutiérrez *et al.*, 2017; Jonason *et al.*, 2017; Zhang *et al.*, 2020a; Rurangwa *et al.*, 2021).

Habitat destruction for creating agricultural lands (Gibbs *et al.*, 2010) and landscape simplification driven by rapid and unsustainable agricultural intensification (Matson *et al.*, 1997; Tscharntke *et al.*, 2005; Newbold *et al.*, 2015) has led to widespread decline in biodiversity and the ES in both natural forest ecosystems and adjacent agroecosystems. Ecosystem resilience depends on functional redundancy and response diversity (Laliberté *et al.*, 2010). Landscape simplification leads to biotic homogenization at different facets of taxonomic, functional and phylogenetic diversity of birds, butterflies, and other taxa (Börschig *et al.*, 2013; Gámez-Virués *et al.*, 2015; Almeida *et al.*, 2016; Bregman *et al.*, 2016; García-Navas *et al.*, 2020; Rurangwa *et al.*, 2021). It also drives loss of FD and homogenization in functional composition due to its adverse effect on functional redundancy and response diversity in birds, butterflies and plants (Flynn *et al.*, 2009; Loos *et al.*, 2014, 2015; Luck *et al.*, 2013, 2015; Hanspach *et al.*, 2015; Matuoka *et al.*, 2020).

Landscape heterogeneity not only promotes retention of taxonomic and functional diversity in birds (Fischer *et al.*, 2011; Morante-Filho *et al.*, 2016), but also its interactions with FD drives the potential for different avian-mediated ES including natural pest control, seed dispersal, pollination, scavenging, nutrient deposition and ecosystem engineering (Şekercioğlu, 2006, 2012; Barbaro *et al.*, 2014, 2017, 2019; Bregman *et al.*, 2016; Şekercioğlu, *et al.*, 2016; Ibarra *et al.*, 2017; Peters *et al.*, 2019). Similarly, landscape heterogeneity promotes both taxonomic diversity and FD in butterflies (Kumar *et al.*, 2009; Perović *et al.*, 2015; Hanspach *et al.*, 2015) and also determines the potential for butterfly-mediated ES such as pollination and pest control (Cardinale *et al.*, 2012; Cussera *et al.*, 2016; Jain *et al.*, 2016). It has been increasingly recognized that traditional and organic agroecosystems has high potential for conservation of biodiversity (both TD and FD) and retention of associated ES (Mcneely and Schroth, 2006; Tscharntke *et al.*, 2008; Martin *et al.*, 2012; Tuck *et al.*, 2014; Goded *et al.*, 2019; Katayama *et al.*, 2019). These wildlife-friendly agricultural landscapes also mitigate the homogenization of functional composition due to their high landscape heterogeneity (Fahrig *et al.*, 2011; Fischer *et al.*, 2011; Ekroos *et al.*, 2013; Perović *et al.*, 2015; Hiley *et al.*, 2016; Goded *et al.*, 2018).

In India, past studies on TD and community composition of birds were mostly undertaken in the Western Ghats, along with few studies in the Himalaya (details given in Chapter 2 and 4 of this thesis). Similarly, few such studies are conducted on butterflies in agroecosystems, mainly from the Western Ghats (Kunte, 1997; Kunte *et al.*, 1999; Shahabuddin and Ali, 2001; Dolia *et al.*, 2008). Previous studies on birds and butterflies in human modified landscapes including agroecosystems (and also natural forest ecosystems) of India has been limited to TD and community composition except a recent study on both TD and FD of frugivorous birds in human-modified landscape in Assam, North East India (Cottee-Jones *et al.*, 2015). The studies on the effects of land use on both FD and functional composition of birds and butterflies (like most other taxa) are highly under-represented from the tropical regions (but see Clough *et al.*, 2009; Flynn *et al.*, 2009; Tscharntke *et al.*, 2012; Bregman *et al.*, 2016; Rurangwa *et al.*, 2021), especially in the Himalaya. Very few studies from the tropical ecosystems has assessed the potentiality of traditional and organic agroecosystems in sustaining high FD and mitigating homogenization of functional composition of birds, butterflies, or other taxa (Tscharntke *et al.*, 2008; Martin *et al.*, 2012). The anthropogenic land use and climate change have threatened the hyperdiverse tropical ecosystems with negative consequences for biodiversity (at different facets: taxonomic, functional and phylogenetic diversity) and functional composition consequently resulting into disruption of ES (Börschig *et al.*, 2013; Pandit, 2017; Barlow *et al.*, 2018; Peters *et al.*, 2019; Rurangwa *et al.*, 2021). These threats are particularly high in the low to mid elevations which are dominated by agricultural landscape but harbour the highest biodiversity across taxa, a pattern reported from tropical mountains globally as well as within the Himalaya (Zomer *et al.*, 2001; Pandit, 2017;Acharya *et al.*, 2011a, 2011b; Peters *et al.*, 2019).

The birds and butterflies are two widely recognized bio-indicator taxa among the vertebrates and invertebrates, respectively to assess health of diverse ecosystems (Kremen, 1992; ILTEO, 2015; Herrando *et al.*, 2016). The FD is the diversity of traits that determine roles or function of species in an ecosystem, while, functional composition is the multivariate distribution of traits across co-occurring species (Tilman, 2001; Gravel *et al.*, 2016). In this chapter, I hypothesized that organic and traditionally managed agroecosystems of Eastern Himalaya sustain high FD and mitigate homogenization of functional composition of birds and butterflies, and retain associated ES. The main objectives of this chapter were: (1) to understand the pattern and plausible determinants of FD of birds and butterflies along agroecosystem-forest gradient of Sikkim, Eastern Himalaya, India, and (2) to understand pattern and drivers of functional composition of birds and butterflies.

6.2. Materials and Methods

6.2.1. Study area and sampling sites

As detailed in **Chapter 3**, this study was conducted in Sikkim, which forms an important part of globally significant biodiversity hotspot of Himalaya (Mittermeier *et* *al.*, 2011) and agrobiodiversity hotspot of Eastern Himalaya (Sharma *et al.*, 2016a). The study covered three representative agroecosystems namely, MOAS, FAS, LCAS, along with adjoining Forest ecosystems within East and South districts of Sikkim covering the elevation gradient of 600 m to 2000 m AMSL.

For each type of IFS and adjoining Forest ecosystems, three different study sites were selected for field sampling. I laid 24 transects (6 per ecosystem of 1km length each, spaced ≥1 km apart) and 240 permanent sampling points (10 in each transect, spaced ≥100 m apart) across these four ecosystem.

6.2.2. Sampling and life-history traits

6.2.2.1. Bird sampling and functional traits

Birds were sampled along the predetermined transect following open-width point count method (Bibby *et al.*, 2000). The sampling procedure and effort spent in field during the study for data collection is given in detail in Chapter 4 of this thesis.

The FD and functional composition of bird communities were computed covering a species-trait matrix of 17 life-history and autecological traits which includes 11 categorical (biogeographic range size, elevational range size, breeding onset/ average egg laying date, clutch size, nest substrate, ES, habitat specialization, foraging substrates, mixed flocks (individuals of at least 2 species usually insectivores that move together while foraging), migration status, global population trend (as per IUCN) and six continuous traits (body mass, generation length, habitat breadth, diet breadth, foraging substrate breadth, foraging method breadth; details given in Table 6.1). These traits were collated from standard literatures (Ali, 1962, 1977; Ali and Ripley, 2002, Beletsky, 2006; Dunning, 2008; Grimmett *et al.*, 2011, 2019; del

Hoyo *et al*., 2020; BirdLife International, 2020a), supplemented by observations in the field during bird sampling. These traits are key indicators of individual bird species' responses to environmental and land-use change (Barbaro and Van Halder, 2009; Flynn *et al.*, 2009; Bregman *et al.*, 2014; Ehlers Smith *et al.*, 2015; Hanspach *et al.*, 2015; Jonason *et al.*, 2017; Rurangwa *et al.*, 2021) and provision of different avian-mediated ES (Şekercioğlu, 2006; Bregman *et al*., 2016; Şekercioğlu *et al.*, 2016; Peters *et al.*, 2019).

Table 6.1. Life history and autecological traits of birds used to calculate functional diversity and functional composition.

Sl. No.	Traits	Categories/range	Trait code
1	Body mass	5 - 9797.95 g	BM
2	Generation length	2.4 - 18.4 years	GenLen
3	Elevational range size	1. <1000 m	ER_small
		2. 1000-2000 m	ER_medium
		3. >2000 m	ER_large
4	Biogeographic range	1. Himalaya	BR_narrow
		2. Himalaya & Indo-Burma	BR_ low
		3. South, SE Asia	BR_med
		4. Oriental region and beyond	BR_widesp
5	Breeding onset	1. March	L_March
		2. April	L_April
		3. May	L_May
		4. June -September	L_June
6	Clutch size	1. 1-3 eggs (small)	CS_small
		2. 3-4 eggs (medium)	CS_med
		3. >4 eggs (large)	CS_large
7	Nest substrate	1. Cavity (tree or others)	NS_cavity
		2. Ground	NS_ground
		3. Shrubs or small tree	NS_shrub
		4. Large trees	NS_tree
		5. Build structures	NS_build
		6. Brood parasitic	NS_BbroodParasitic
8	Ecosystem Services	1. Invertebrate pest control	ES_InvPestCont
		2. Pollination	ES_Pollination
		3. Seed dispersal	ES_SeedDispers
		4. Waste disposal	ES_WasteDispos
		5. Ecosystem engineering	ES_EcoEngineer
		6. Nutrient deposition	ES_NutrDep
9	Habitat specialization	1. Forest specialist	HS_FS
		2. Forest generalist	HS_FG

		3. Open-land species	HS_OA
10	Habitat breadth	1-7	hab_breadth
11	Diet breadth	1-6	diet_breadth
12	Foraging substrate breadth	1-5	ForSubs_breadth
13	Foraging substrates	1. Ground	FS_ground
		2. Understorey	FS_understorey
		3. Midstorey	FS_midstorey
		4. Canopy	FS_canopy
		5. Water around surface	FS_water
		6. Air	FS_air
14	Foraging method breadth	1-5	ForMeth_breadth
15	Mixed-species flocks	Species joining mixed flock	Mixed.flocks
16	Migration status	1. Resident	Mig_Res
		2. Altitudinal migrant	Mig_AlM
		3. Summer visitor	Mig_SV
		4. Winter visitor & passage migrant	Mig_WV.PM
17	Global population trend	1. Increasing	trend_incr
		2. Stable	trend_stable
		3. Decreasing	trend_decr
		4. Unknown	trend_unkn.

6.2.2.2. Butterfly sampling and functional traits

Fixed width point count method (a modified form of transect count) was followed for sampling butterflies along the transect (Details provided in Chapter 5). Butterflies were sampled for five minutes within the 5 m radius plot at the pre-established permanent point and identified at the wing with the help of photo plates and identifying characters provided in Wynter-Blyth (1957), Haribal (1992), and Kehimkar (2008, 2016).

The FD and functional composition of butterfly communities were computed covering a species–trait matrix of 14 life-history traits (Table 6.2) which includes 13 categorical (elevational range, biogeographic range, eggs laid, egg shape, larval host specificity, larval host plant type, adult feeding guild, habitat specialization, microhabitat specialization, daily activity, flight ability, defence mechanism, migration) and one continuous traits (Wingspan).

Table 6.2. Life history and autecological traits of butterflies used to calculate functional diversity and functional composition.

Sl. No.	Traits	Categories/range	Trait code
1	Wingspan (in cm)	20-155 mm	Wingspan
2	Elevational range	1. <1000m	ER_small
		2. 1000-2000m	ER_medium
		3. >2000m	ER_large
3	Biogeographic range	1. Himalaya	BR_Himalayas
		2. Himalaya & Indo-Burma	BR_ Himalayas & IB
		3. Oriental region and beyond	BR_oriental
4	Eggs laid	1. Singly	EL_singly
		2. in batches	EL_batches
5	Egg shape	1. Cylindrical	ES_cylindrical
		2. Dome	ES_dome
		3. Spherical	ES_spherical
		4. Turban	ES_turban
6	Larval host specificity	1. Monophagous	LHS_monoph
		2. Polyphagous	LHS_polyph
7	Larval host plant type	1. Grasses	LHP_grass
		2. Herbaceous dicots	LHP_herb
		3. Shrub	LHP_shrub
		4. Tree	LHP_tree
8	Adult feeding guild	1. Floral nectar	AFG_floral nectar
		2. Ripe fruits	AFG_ripe fruits
		3. Urine/carrion	AFG_ carrion
		4. Sap	AFG_sap
9	Habitat specialization	1. Forest specialist	HS_FS
		2. Forest generalist	HS_FG
10	Microhabitat specialization	1. Understorey	Habit_shade
		2. Forest canopy or gap	Habit_sun
11	Daily activity	1. Crepuscular	DA_crepuscular
		2. Diurnal	DA_diurnal
12	Flight ability	1. Fast and erratic	FA_fast
		2. Slow close to the ground	FA_slow
13	Defence mechanism/Colouration	1.Conspicuous colouration (chemically defended species and their mimics)	C_conspicuous
		2. Dark colouration (Cryptic species)	C_cryptic
14	Migration	Migratory species	Migratory

The functional traits for each species were retrieved from literatures (Wynter-Blyth, 1957; Haribal, 1992; Kehimkar, 2008, 2016; and Kunte *et al.*, 2018), and supplemented by field observations. Data for larval host specificity was available for only 213 butterfly species (124 polyphagous species, 89 monophagous species; 55

data deficient). In case of larval host plant type, the data deficient species were assigned the traits of co-genera or co-sub-family. These selected butterfly traits are considered as important indicators of individual species responses to land-use and environmental changes (Koh, 2007; Barbaro and Van Halder, 2009; Hanspach *et al.*, 2015; Jonason *et al.*, 2017; Van Halder *et al.*, 2017; Kaltsas *et al.*, 2018), and provision of different butterfly-mediated ES (Cardinale *et al.*, 2012; Kehimkar, 2008, 2016; Jain *et al.*, 2016).

6.2.3. Environmental variables

To identify plausible determinants of FD and functional composition of birds and butterflies in the present study, I quantified different habitat and environmental variables for each of the 24 transect. I followed the quadrat method (size 20m*20m at each permanent point; 10 per transect) for sampling trees (by measuring stems>20cm gbh). From these vegetation data, I calculated tree species richness (tsr), tree density (tden.: stems ha^{-1}), and tree basal area (tba: m^2 ha^{-1}) for each transects. I also quantified MAT (temp: °C) and MAP (rain: mm) from WorldClim database (<www.worldclim.org>; Fick and Hijmans, 2017) using ArcGIS 10.4. The mean values of elevation (elev: m), latitude (°N) and longitude (°E) were obtained by using hand-held GPS. Using a spherical convex densitometer, I obtained percentage canopy cover (pcc: %) by averaging 40 readings (four in each direction of 10 permanent points) for each transects (details provided in Chapter 4 of this thesis; Appendix A).

6.2.4. Data analysis

All analyses were done in R version 3.1.4 (R Core Team, 2017) using various packages.

6.2.4.1. Functional diversity

To compute FD, it is necessary to: (a) account for species abundance, (b) consider multiple traits (continuous and categorical) simultaneously and (c) calculate all facets of FD (Mason *et al.*, 2005). Since no single indices fulfil all these criteria, therefore different complementary indices are quantified, including functional richness, evenness, divergence, and dispersion (Mouillot *et al.*, 2013). Using the functional traits for birds (Table 6.1) and butterflies (Table 6.2), four multidimensional FD indices were calculated separately for the two taxa: (1) functional richness (FRic; the amount or "volume" of functional space filled by an assemblage); (2) functional evenness (FEve; the evenness of abundance distribution in a functional trait space); (3) functional divergence (FDiv; the spread of abundance along a functional trait axis) (Mason *et al.*, 2005; Villéger *et al.*, 2008), and(4) functional dispersion (FDis; the multivariate measure of trait dispersion) (Laliberté and Legendre, 2010). Additionally, single-trait metrics with community-weighted mean (CWM) trait values (birds: CWM_bodymass; butterflies: CWM_Wingspan) were also calculated. Among all traits, the body mass for birds (Ding *et al.*, 2013, Luck *et al.*, 2013) and wingspan for butterflies (Öckinger*et al.*, 2010; Sekar, 2012; Kuussaari*et al.*, 2014) are the most important trait determining their response to land-use change. For comparison, two indices of the TD viz., Shannon-Wiener diversity (described in Chapter 4 for birds and Chapter 5 for butterflies) and species richness were also quantified for each transects. The TD and the observed FD (FRic, FEve, FDiv, and FDis) of bird and butterfly communities were compared among the four ecosystems using one-way ANOVA. The comparison and one-way ANOVA test was also carried out for CWM_bodymass of birds and CWM_Wingspan of butterflies. To correctly identify the particular context making the real differences in the functional diversity parameters, Bonferroni corrected post-hoc, pair-wise comparisons was carried out using the “multcomp” package in R. The Pearson correlation tests were carried out separately for birds and butterflies to understand the relationships of different FD indices with their corresponding TD values, and different environmental variables.

Among FD indices, FRic was strongly correlated with total species richness in butterflies (r = 0.94) and birds (r = 0.6). Hence, the null model approach was used to test whether changes in observed FRic, FEve, FDiv, and FDis were independent of changes in species richness (and Shannon-Wiener diversity). I generated 999 communities by randomly choosing species from the species-pool separately, for birds (221 species detected across 24 transects) and butterflies (268 species detected across 24 transects) without replacement and by randomly assigning the species to each transect but ensuring constant species richness within a transect.

6.2.4.2. Functional composition and trait-environment relationships

I used two complementary approaches of RLQ analysis and fourth-corner analysis to test the relationships between functional composition and environment variables. The RLQ analysis allowed to directly link functional traits with environmental variables through the ordination of abundance matrix of species, which allows in generalizing coinertia analysis for examining the joint structure of three-table data sets of R (environmental variables matrix), L (species abundance matrix), and Q (functional traits matrix) (Doledec *et al.*, 1996; Ribera *et al.*, 2001; Dray *et al.*, 2003). The RLQ analysis approach maximizes the covariance between sites and species on the basis of environmental variables of the sites and the species traits (Doledec *et al.*, 1996). It is a doubly constrained ordination method since the covariance between R-table and Q-table is constrained by the abundance of species, present in L-table

(Hausner *et al.*, 2003). The RLQ analysis was conducted separately for birds and butterflies based on separate ordination on environmental variables (R; Hill-Smith principal component analysis), species abundance matrix (L; correspondence analysis) and species life traits (Q; Hill-Smith principal component analysis). I used fourth-corner analysis to further test the associations (strength and direction) of species abundances (Table L) with environmental variable matrix (Table R) and functional trait matrix (Table Q) (Dray *et al.*, 2014) for both the taxa separately. The fourth-corner model allows relating individual species traits with different environmental variables by fitting a predictive model of L as a function of R and Q and their interactions (Dray *et al.*, 2014). I tested significance of fourth-corner analysis using model 6, which is a combination of two null models: models 2 (permutations of sites) and 4 (permutations of species) with 999 permutations separately for bird and butterfly communities (Dray and Dufour, 2007).

All the statistical analyses were conducted in R3.1.4 (R Core Team, 2017) using the packages "vegan" for TD, "FD" for FD metrics, "picante" to perform randomization, "corrplot" for correlation tests, "ade4" for RLQ analysis and fourth-corner analysis (Barbaro and Van Halder, 2009; Ding *et al.*, 2013; Dray *et al.*, 2014; Hanspach *et al.*, 2015; Barbaro *et al.*, 2017; Chen *et al.*, 2019).

6.3. Results

6.3.1. Functional diversity of birds

The FD for total bird communities showed variation among different ecosystems. FRic was highest in MOAS and showed clear declining trend (near-significantly: $F_{3, 20} = 2.76$, p= 0.07; Fig. 6.1a) along agroecosystem-forest gradient. The FDis was significantly highest in FAS ($F_{3, 20} = 6.44$, p<0.01; Fig. 6.1d), whereas, FEve (Fig.

6.1b) and FDiv (Fig. 6.1c) were highest (but non-significantly) in Forests, and LCAS, respectively. The CWM_bodymass was highest (non-significantly) in FAS, followed by Forest, MOAS and declined in LCAS (Fig. 6.1e). The pattern of FRic contrasted with that of bird species richness (Fig. 6.1f) and Shannon-Wiener diversity (see Chapter 4: Fig. 4.3m). Bonferroni corrected post-hoc analysis further showed significant difference in FDis of birds only between FAS and Forest ($p<0.05$), whereas, for species richness only between Forest and LCAS ($p<0.05$).

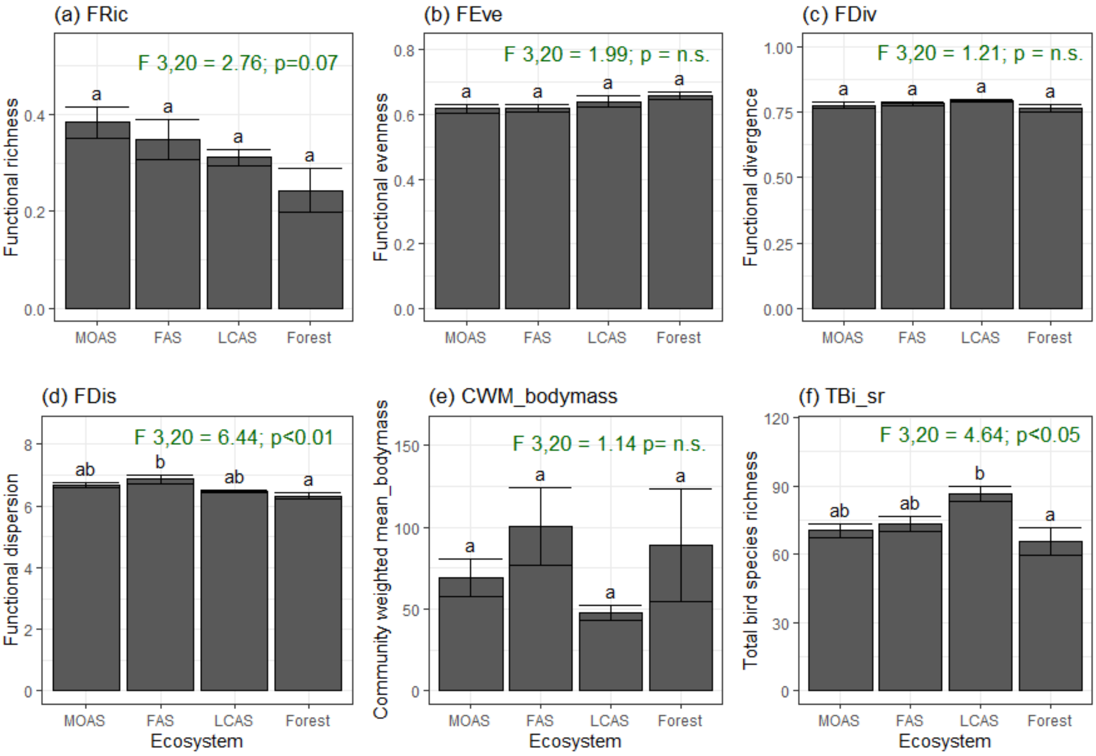


Fig. 6.1. Pattern of the different functional diversity indices (a-e) and species richness (f) for total birds along agroecosystem-forest gradient of Sikkim, Eastern Himalaya, India. Results of ANOVA test including F-statistics, degree of freedom (3: between group, 20: within group coming from Ecosystems and Transects, respectively), and significance level ($p<0.01$; $p<0.05$; n.s.: non-significant) are also depicted. For Bonferroni corrected post-hoc, pair-wise comparisons, bars not sharing letters are significantly different at the alpha level of $p = 0.05$.

For total birds, strong significant positive correlation was found for FRic with species richness, and for FDiv with FDis. With the different environmental variables, FRic showed weak (non-significant) negative correlation with elevation, pcc, tree density, tree basal area, and positive correlation with tree species richness, MAT and MAP (Fig. 6.2).

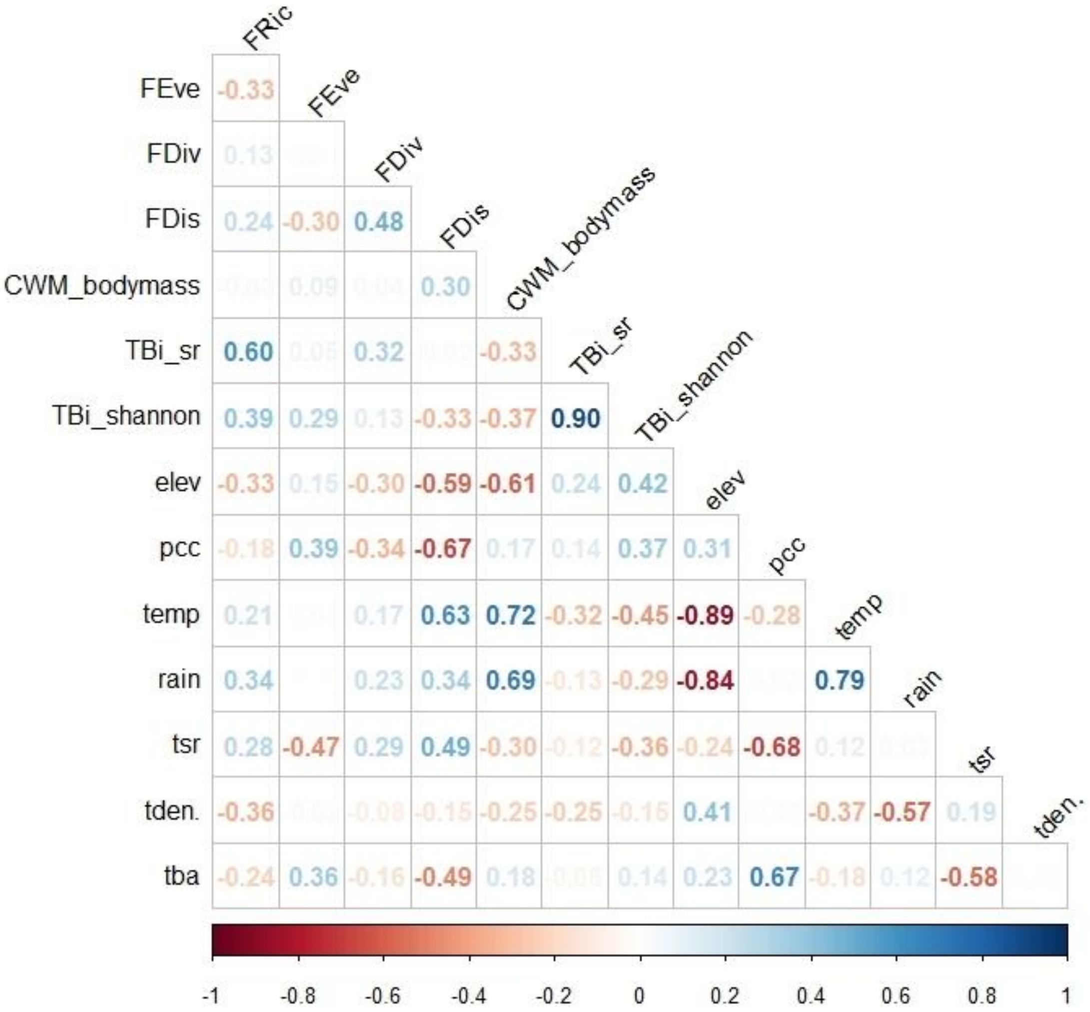


Fig. 6.2. Correlation matrix showing significant ($p<0.05$) and strong ($r>|0.40|$) correlation coefficients between the bird functional diversity measures and predictor variables along the agroecosystem- forest gradient of Sikkim, Eastern Himalaya. The habitat and environmental variables are: elevation (elev), percentage canopy cover (pcc), mean annual precipitation(rain), mean annual temperature (temp), tree species richness (tsr), tree density (tden.), and tree basal area (tba); the bird diversity parameters are: total birds species richness (TBi_sr), Shannon-Wiener diversity (TBi_shannon), functional richness (FRic), functional evenness (FEve), functional divergence (FDiv), functional dispersion (FDis), community-weighted mean of body mass (CWM_bodymass).

FEve had significant negative correlation with tsr but positive (non-significant) correlation with pcc and tree basal area. FDiv correlated non-significantly positively with tree species richness, but negatively with elevation and pcc. FDis showed strong significant correlation (relationship being positive with tree species richness and MAT, but negative with elevation, pcc and tree basal area). CWM_bodymass correlated significant positively with MAT and MAP but negatively with elevation (Fig. 6.2).

For birds, the expected FRic values (mean FRic of 999 randomization) increased significantly and linearly only with Shannon-Wiener diversity (r^2=0.33; p<0.001; Fig. 6.3a) and species richness (r^2=0.46; p<0.001; Fig. 6.3e). Such relationships of expected values of FEve, FDiv and FDis with Shannon-Wiener diversity (Fig. 6.3b-d) and species richness (Fig. 6.3f-h) were weak (r2≤0.09; p<0.001).

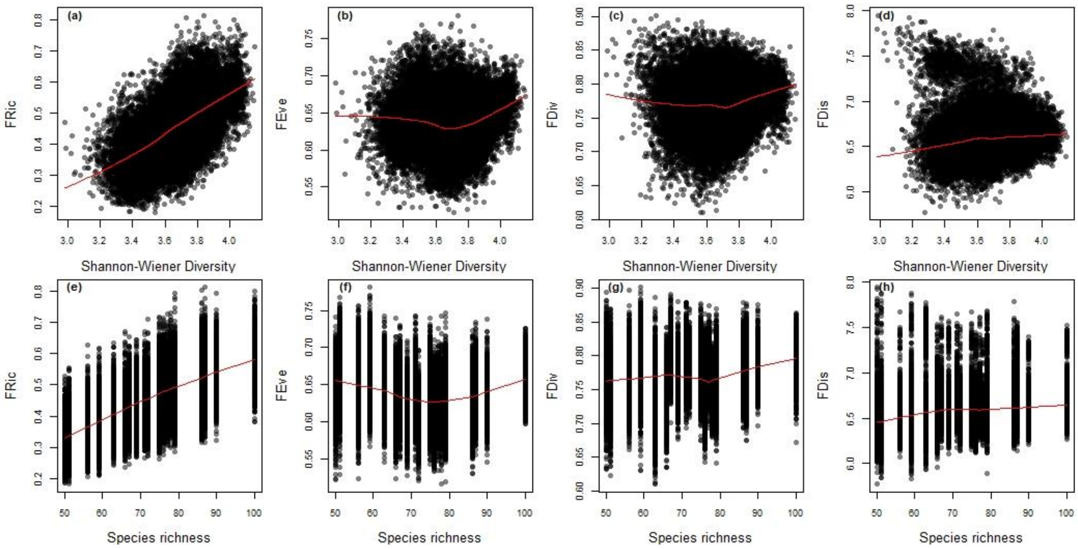


Fig. 6.3. Simulated functional diversity indices vs. Shannon-Wiener diversity and species richness for the four metrics of functional diversity of birds for 24 transects in the different IFS and Forest ecosystems of Sikkim, Eastern Himalaya. The values obtained in each of 999 randomization test are denoted by black dots, while red line depicts fitted regression.

6.3.2. Functional diversity of butterflies

Analysis of FD for total butterfly communities along the agroecosystem-forest gradient revealed highest (near significantly) FRic ($F_{3, 20}$ = 2.82, p=0.06) in MOAS which declined along the gradient (Fig. 6.4a), whereas, FDis was significantly highest ($F_{3, 20}$ = 3.43, p<0.05) in LCAS (Fig. 6.4d). However, highest (though non-significantly) FEve (Fig. 6.4b) and FDiv (Fig. 6.4c) was observed in Forests. The CWM_Wingspan was highest (non-significantly) in MOAS, which declined in LCAS (Fig. 6.4e). The patterns of FRic mirrored the declining pattern of butterfly species richness (Fig. 6.4f) as well as Shannon-Wiener diversity (see Chapter 5: Fig. 5.2i). Post-hoc, pair-wise comparisons with Bonferroni correction showed significant difference only for butterfly FDis between FAS and Forest ($p<0.05$).

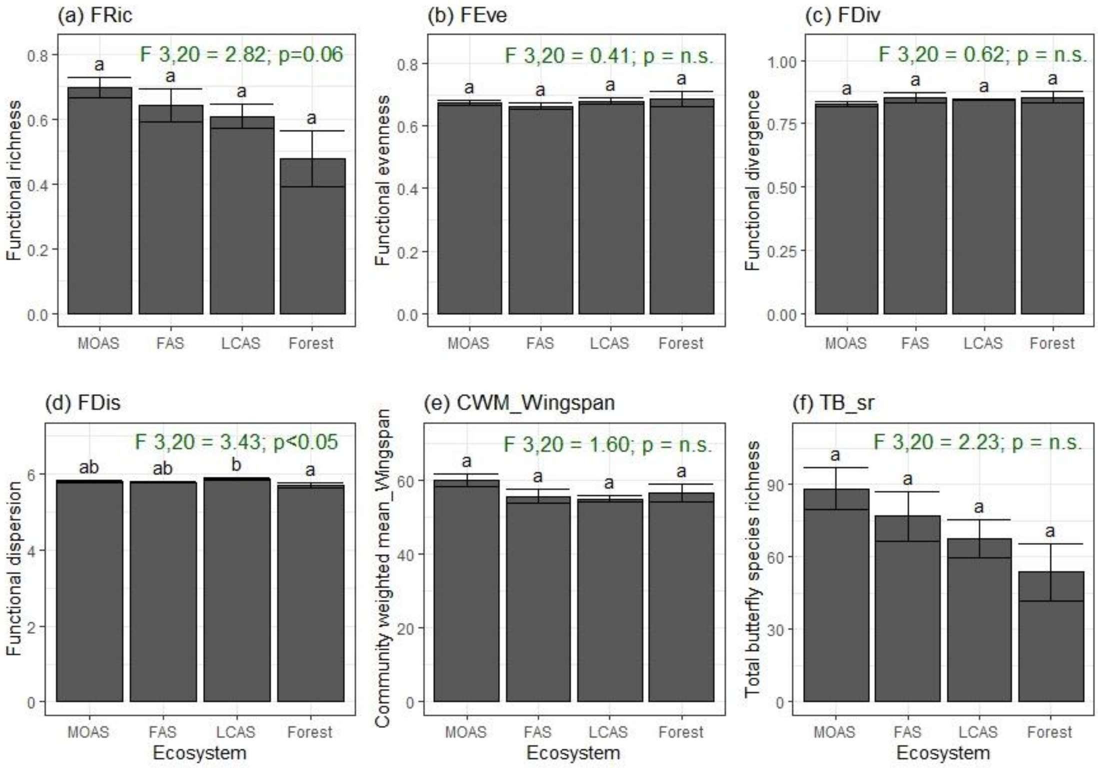


Fig. 6.4. Pattern of the different functional diversity indices (a-e), and species richness (f) for total butterflies along agroecosystem-forest gradient of Sikkim, Eastern Himalaya. Results of ANOVA test including F-statistics, degree of freedom (3: between group, 20: within group, coming from Ecosystems and Transects, respectively), and significance level ($p<0.05$; n.s.: non-significant) are also shown. For Bonferroni corrected post-hoc, pair-wise comparisons, bars not sharing letters are significantly different at the alpha level of $p = 0.05$.

For butterfly communities, there was highly significant correlation of TD (species richness, Shannon-Wiener diversity) with different FD indices with relationship being positive for FRic, FDis and CWM_Wingspan but negative for FDiv. Similarly, FDis showed significant positive correlation with FRic, whereas FDiv correlated significant negatively with FRic. There was strong significant relationship of butterfly FD indices with all environmental variables except pcc and tree species richness. The FRic and CWM_Wingspan had significant positive correlation with MAT and MAP, but significant negative correlation with elevation and tree density. There was also significant negative correlation between FRic and tree basal area.

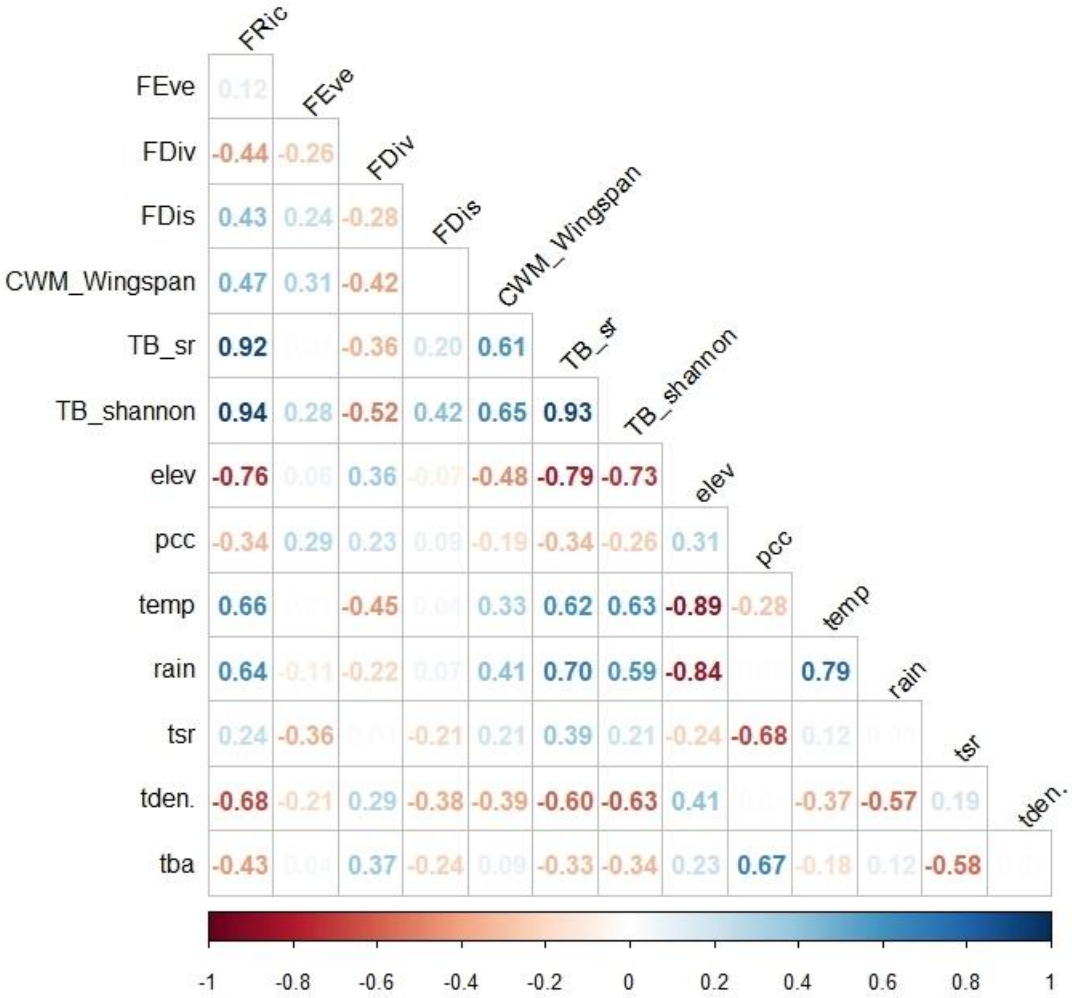


Fig. 6.5. Correlation matrix showing significant (p<0.05) and strong (r>|0.40|) correlation coefficients between the butterfly functional diversity measures and predictor variables along the agroecosystem-forest gradient of Sikkim, Eastern Himalaya. The habitat and environmental variables are: elevation (elev), percentage canopy cover (pcc), mean annual precipitation(rain), mean annual temperature (temp), tree species richness (tsr), tree density (tden.), and tree basal area (tba); the butterfly diversity parameters are: total butterfly species richness (TB_sr), total butterfly Shannon-Wiener diversity (TB_shannon), functional richness (FRic), functional evenness (FEve), functional divergence (FDiv), functional dispersion (FDis), community-weighted mean of wingspan (CWM_Wingspan).

The FDiv correlated significantly negatively with MAT, but positively (non-significant) with elevation, tree basal area, tree density, and pcc. The FEve showed weak negative and positive correlation with tree species richness and pcc, respectively. The FDis also had weak negative correlation with tree species richness, density and basal area (Fig. 6.5).

For butterflies, the expected FRic values (mean FRic of 999 randomization) increased significantly and linearly with Shannon-Wiener diversity (r^2=0.82; p<0.001; Fig. 6.6a) and species richness (r^2=0.85; p<0.001) (Fig. 6.6e). The expected FEve and FDiv declined (but FDis increased) weakly with Shannon-Wiener diversity (Fig. 6.6b-d) and species richness ($r^2 \leq 0.1$; p<0.001; Fig. 6.6f-h).

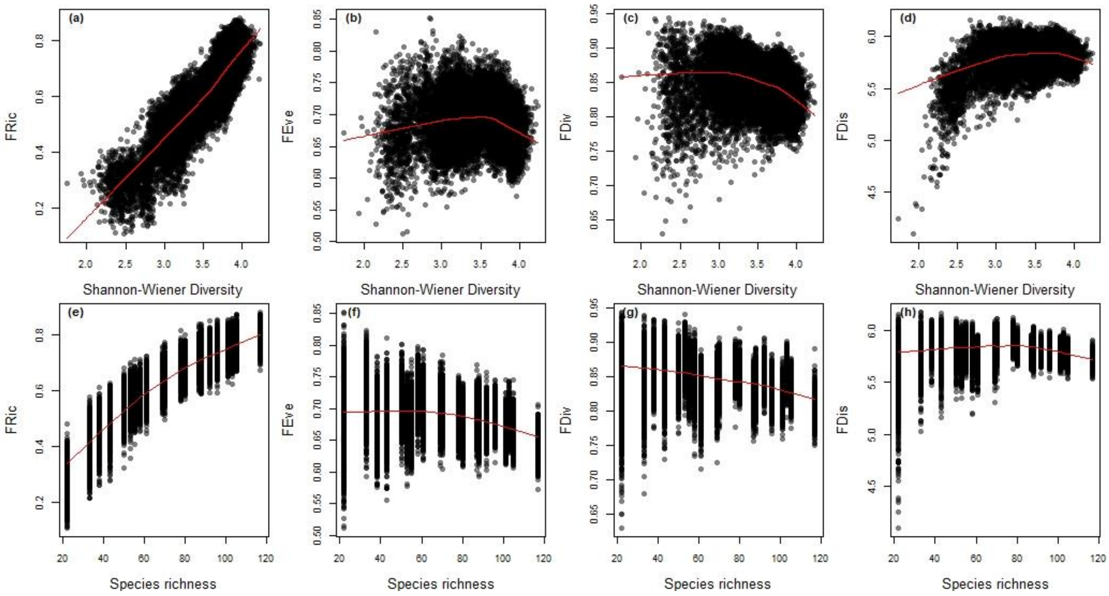


Fig. 6.6. Simulated functional diversity indices vs. species richness and Shannon-Wiener diversity for the four metrics of functional diversity of butterflies along the agroecosystem- forest gradient of Sikkim, Eastern Himalaya. The values obtained in each of 999 randomization tests are denoted by black dots, while red line depicts fitted regression.

6.3.3. Determinants of functional composition-in birds

There was a significant overall association between bird species traits and landscape variables (permutation test, p< 0.01) in the RLQ analysis with the first two axes accounting for 97.5% variance (Table 6.3). The first axis of RLQ analysis (explained variance: 91.4%) correlated strongly positively with MAT, MAP, tree species richness but negatively with elevation, pcc, tree density and tree basal area. It corresponded with the land-use change gradient from simplified agroecosystems (MOAS, FAS) with high tree species richness but low tree density and tree basal area having higher MAT, MAP towards more complex ecosystems (LCAS, Forests) with high tree density, tree basal area at higher elevation (Table 6.3, 6.5; Fig. 6.7a).

Table 6.3. Summary of RLQ analyses for bird functional composition.

Total inertia: 1.855			
Projected inertia (%):			
	Axis1	Axis2	
	91.423	6.064	
Eigenvalues decomposition:			
	eig.	covariance	correlation
1	1.696	1.302	0.333
2	0.113	0.335	0.156
Inertia &coinertia R			
	inertia	max	ratio
1	3.104	3.128	0.993
1+2	5.281	5.383	0.981
Inertia &coinertia Q			
	inertia	max	ratio
1	4.927	5.567	0.885
1+2	7.063	9.618	0.734
Correlation L			
	corr	max	ratio
1	0.333	0.581	0.573
2	0.156	0.413	0.377

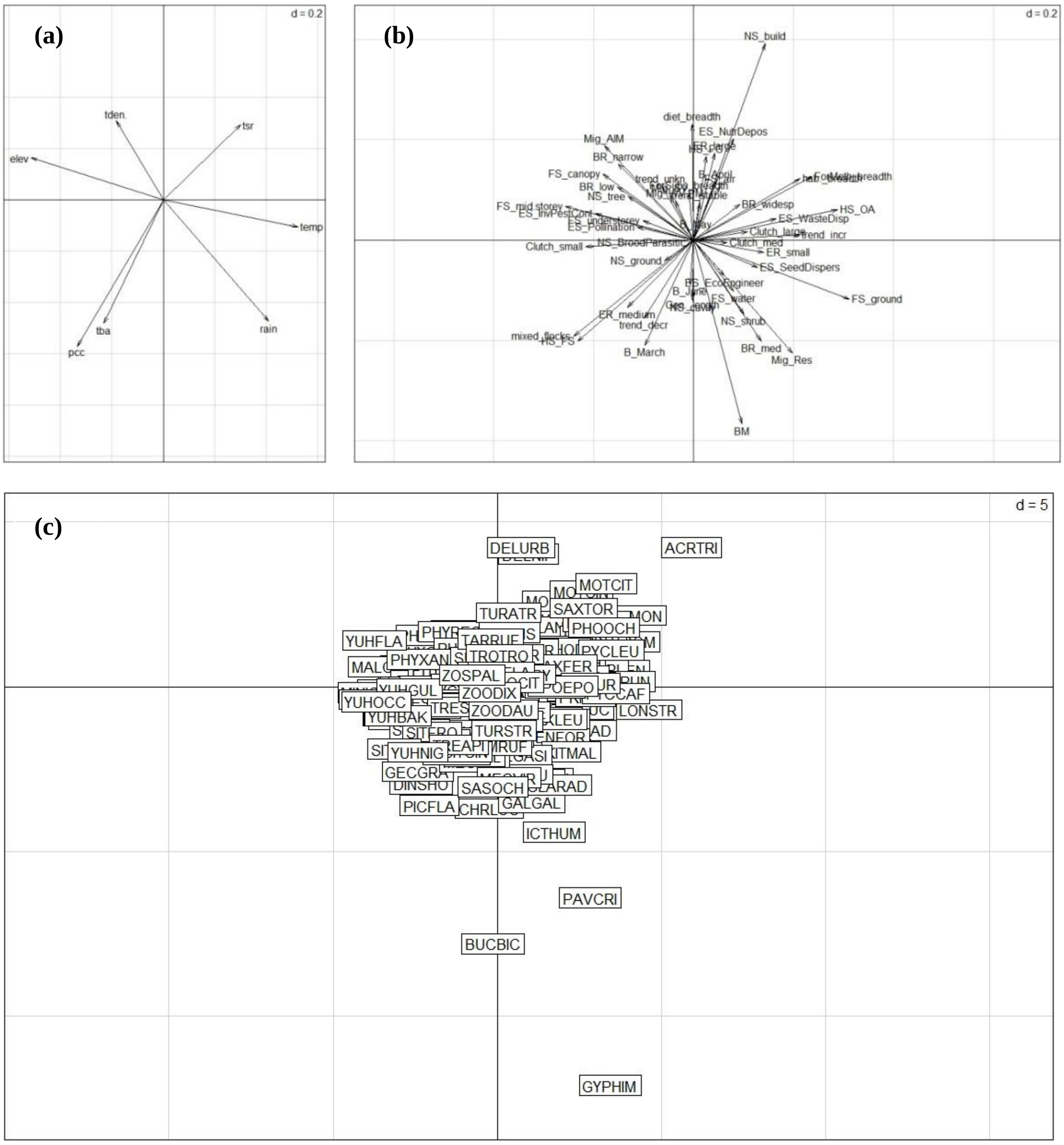


Fig. 6.7. Graphical display of RLQ scores of (a) landscape variables, (b) life trait categories and (c) bird species along agroecosystem-forest gradient of Sikkim, Eastern Himalaya. Abbreviation of bird species uses the first three letters each of genus and species from scientific names, e.g., PICFLA= *Picus flavinucha*. R- Landscape variables matrix, L- species abundances matrix, and Q- functional traits matrix. I have elaborated environmental variable codes of Fig. 6.7a in Table 6.5, Bird functional trait codes of Fig. 6.7b in Table 6.1, and six letters species code of Fig. 6.7c in Appendix B.

In the first axis of RLQ, widespread (BR_widesp), generalists (HS_OA), ground forager (FS_ground) birds and species with high foraging method (ForMeth_breadth) and habitat breadth (hab_breadth), medium to large reproductive potential (Cluth_med; Clutch_large), increasing global population trend (trend_incr), provisioning ES of seed dispersal (ES_SeedDespers) or waste disposal (ES_WasteDisp) but with low elevational range (ER_low) showed more affinity for the MOAS and FAS (Fig. 6.7b), e.g., *Lonchura striatus*, *Pycnonotus cafer* (Fig. 6.7c). Contrastingly, forest specialist (HS_FS), altitudinal migrant (Mig_AlM), mixed-species flocking birds (mixed_flocks) and species having narrow to low biogeographic range size (BR_narrow, BR_low), medium elevation range (ER_medium), foraging in understorey (FS_understorey), midstorey (FS_midstorey) or canopy (FS_canopy), nesting in tree (NS_tree), ground (NS_ground), or brood parasitic (NS_brood parasitic), low reproductivity (Cluth_small), and contributing ecosystem functioning of pest control (ES_InvPestCont), pollination (ES_Pollination) preferred the LCAS and Forest ecosystems which had greater landscape complexity (Fig. 6.7b), e.g., *Yuhina occipitalis*, *Zosterops palpebrosus* (Fig. 6.7c).

The second axis of RLQ analysis (explained variance: 6.1 %) correlated negatively with tree basal area and percentage canopy cover (Table 6.3, 6.5; Fig. 6.7a). The bird species with high diet breadth (diet_breadth), large elevation range (ER_large), breeding onset in April (B_April), large foraging strata breadth (ForStrata_bredth), habitat generalists (HS_FG), winter visitor and passage migrant (Mig_WV.PM), stable global population trend (trend_stable), foraging in air (FS_air), nesting in buildings (NS_build), contributing to nutrient deposition (Nutr_Depos) were associated with the simplified agroecosystems (MOAS and FAS) (Fig. 6.7b). The typical species were *Turdus atrogularis*, *Delichon urbicum*, etc. (Fig. 6.7c). On the contrary, large bodied (BM), resident (Mig_Res) bird species with early to late breeding onset (B_March, B_June), foraging in and around water (F_water), medium biogeographic range (BR_med), nesting in shrubs (NS_shrub) or cavity (NS_cavity), declining global population trend (trend_decr), contributing to ecosystem engineering

(ES_EcoEngineer) corresponded with LCAS and Forests, the two ecosystems with high habitat and landscape complexity (having higher tree basal area, pcc) (Fig. 6.7b), e.g., *Picus flavinucha* (Fig. 6.7c).

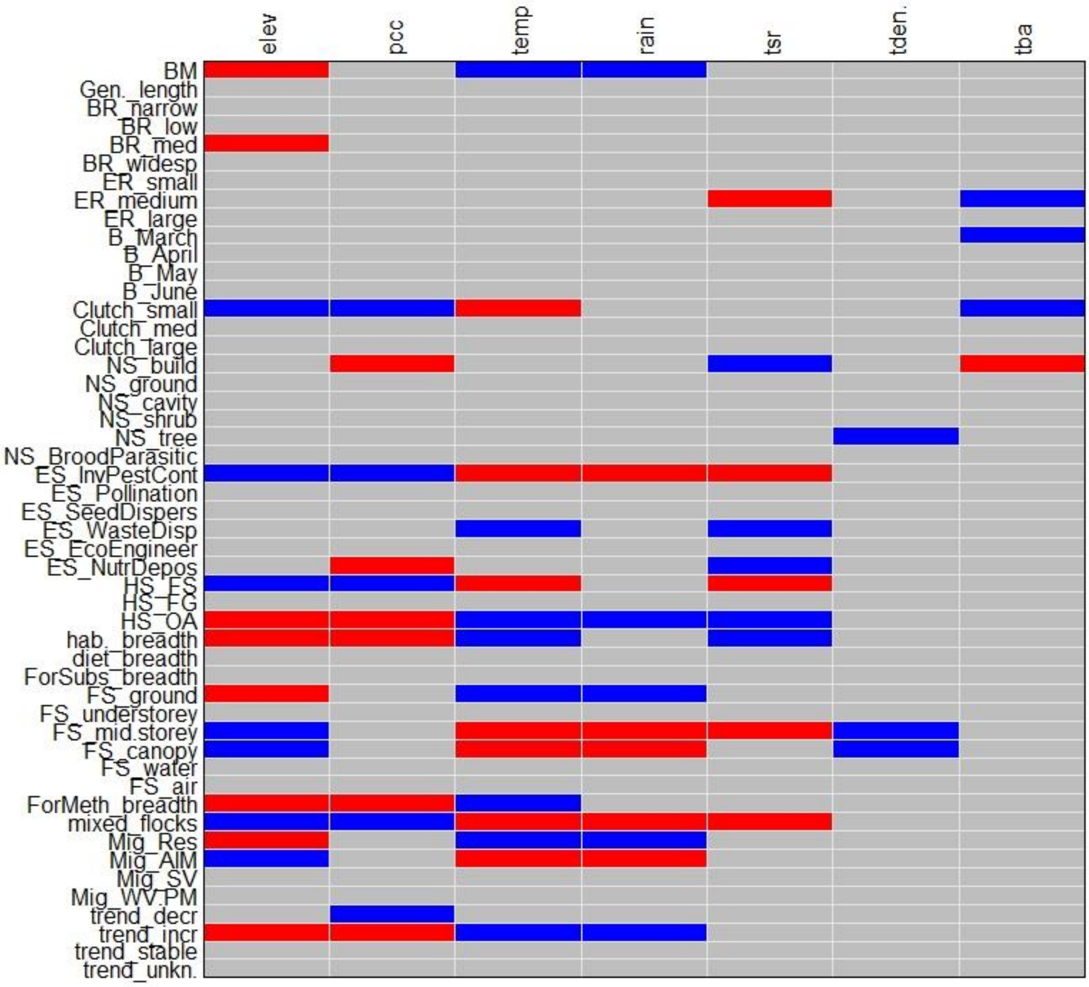


Fig. 6.8. Results of fourth corner analysis showing significant ($p<0.05$) positive (blue) and negative (red) correlation of different bird functional traits with the landscape variables along agroecosystem-forest gradient of Sikkim, Eastern Himalaya. The details of codes for functional traits are given in Table 6.1, while environmental variables are detailed in Table 6.5 and Appendix A.

The global test of fourth-corner analysis reflected highly significant relationship of bird functional composition with both site environmental variables (Model 2, $p<0.01$) and species' functional traits (Model 4, $p<0.01$). I observed significant (at $p<0.05$) relationship with at least one of the environmental variables (mostly with elevation, pcc, MAT, MAP, tree species richness, but also with tree density, tree basal area) for 14 out of 17 bird functional traits (Fig. 6.8) which resembled results from the RLQ analysis (Fig. 6.7). These traits were body mass, biogeographic range size, elevational range, breeding onset, clutch size, nest substrate, ES, habitat specialization, habitat breadth, foraging stratum, foraging method breadth, species joining mixed flocks, migratory status, and global population trend (Fig. 6.8).

6.3.4. Determinants of functional composition in butterflies

RLQ analysis revealed a significant overall association between butterfly species traits and landscape variables (permutation test, p<0.01), with the first two axis accounting for 97.6% variance (Table 6.4). The first axis of RLQ analysis (explained variance: 87.1 %) correlated strongly positively with MAP, MAT, tree species richness but strongly negatively with elevation, tree density and percentage canopy cover (Table 6.4, 6.5; Fig. 6.9.a). It corresponded with the land-use change gradient from agroecosystems with high tree species richness but low tree density and tree basal area as well as higher MAT and MAP (MOAS, FAS) to more complex ecosystems with high tree density, tree basal area at higher elevation (LCAS, Forests). Butterfly species with widespread biogeographical range, medium elevation range, large wingspan, conspicuously coloured, larval host plants from tree and shrub, cylindrical-shaped eggs, occurred preferably in simplified agroecosystems i.e., MOAS and FAS (Fig. 6.9.b), e.g., *Euploea mulciber* (EUPMUL), *Papilio krishna* (PAPKRI) (Fig. 6.9.c). On the other hand, species with narrow (BR_Himalaya) or low (BR_Himalaya & Indo-Burma) biogeographical range, narrow elevation range (ER_narrow), cryptic colouration (C_cryptic) were linked with the complex ecosystems of LCAS and Forests (Fig. 6.9.b), e.g., *Heliophorus moorei* (HELMOR), *Dodona adonira* (DODADO) (Fig. 6.9.c).

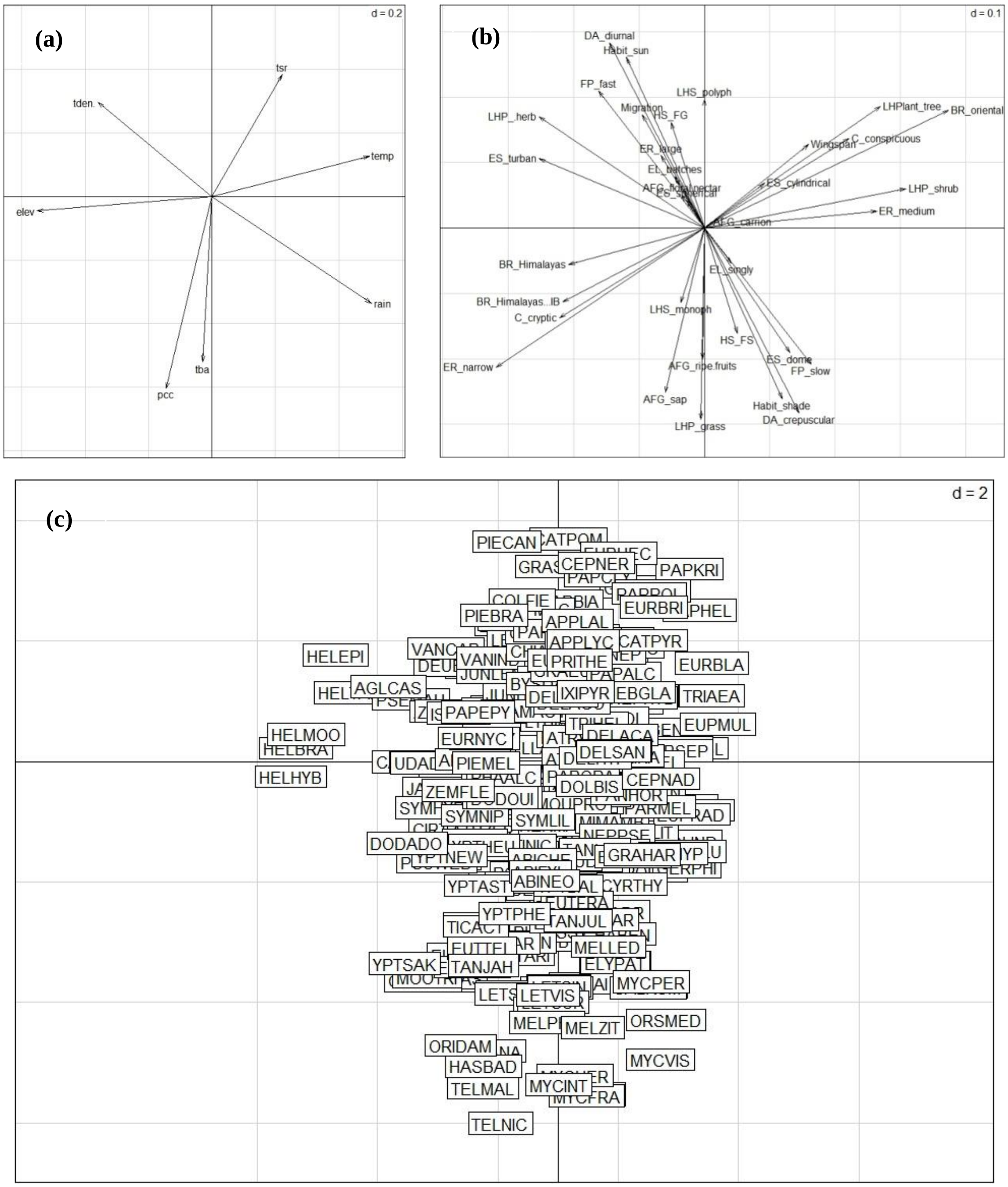


Fig. 6.9. Graphical display of RLQ scores of (a) landscape variables, (b) life trait categories and (c) butterfly species along agroecosystem-forest gradient of Sikkim, Eastern Himalaya. Abbreviation of butterfly species uses the first three letters of genus and species from scientific names, e.g., HELEPI: *Heliophorus epicles*. R- Landscape variables matrix; L- species abundances matrix; Q- functional trait matrix. The code for environmental variables of Fig. 6.9a are detailed in Table 6.5; functional trait codes used in Fig. 6.9b are explained in trait table for butterflies (Table 6.2); and six letters species code of Fig. 6.9c are elaborated in Appendix G.

Table 6.4. Summary of RLQ analyses for butterfly functional composition.

Total inertia: 1.492			
Projected inertia (%):			
	Axis1	Axis2	
	87.101	11.488	
Eigenvalues decomposition:			
	eig.	covariance	correlation
1	1.299	1.140	0.390
2	0.171	0.414	0.113
Inertia &coinertia R:			
	inertia	max	ratio
1	3.108	3.122	0.995
1+2	5.444	5.576	0.976
Inertia &coinertia Q:			
	inertia	max	ratio
1	2.754	7.762	0.355
1+2	8.514	11.330	0.751
Correlation L:			
	corr	max	ratio
1	0.390	0.640	0.609
2	0.113	0.455	0.248

The second axis (explained variance: 11.5 %) correlated negatively with pcc and tree basal area (Table 6.4, 6.5; Fig. 6.9.a). The butterfly species with polyphagous larval host specificity (LHS_polyph), generalist habitat (HS_FG), microhabitat specialized to forest canopy or gap (Habit_sun), large elevation range (ER_large), migratory (Migration), diurnal (DA_diurnal), fast or erratic flight (FP_fast), high reproductive potential (EL_batches), adult feeding on floral nectar (AFG_floral.nectar) tended to occur in the two comparatively simplified agroecosystems (MOAS and FAS) (Fig. 6.9.b), e.g., *Catopsilia pomona* (CATPOM), *Pieris canidia* (PIECAN) (Fig. 6.9.c). On the contrary, species with microhabitat specialized to forest understorey (Habit_shade), narrow larval host specificity (LHP_monoph), low reproductive potential (EL_singly), larval stage feeding on grass (LHP_grass), weak flight (FP_weak), dome-shaped egg (ES_dome), rotten fruits- or sap- feeding as adults (AFG_fruit, AFG_sap), crepuscular (DA_crepuscular) were linked to more complex ecosystems namely LCAS and Forests (Fig. 6.9.b), e.g., *Mycalesis francisca* (MYCFRA), *Hasora badra* (HASBAD) (Fig. 6.9.c).

Table 6.5. Correlations between environmental variables and RLQ axes for birds and butterflies.

Variables	Abbreviation	Birds		Butterflies	
		Axix 1	Axis 2	Axis 1	Axis 2
Elevation (m)	elev	**-0.672**	0.055	**-0.737**	-0.020
Percentage canopy cover (%)	pcc	**-0.435**	-0.192	-0.204	**-0.299**
Mean annual temperature (° C)	temp	**0.685**	-0.035	**0.665**	0.047
Mean annual precipitation (mm)	rain	**0.534**	-0.159	**0.660**	-0.177
Tree species richness	tsr	**0.393**	0.098	**0.292**	0.195
Tree density (stems ha^{-1})	tden.	**-0.239**	0.104	**-0.473**	0.137
Tree basal area (m^2 ha^{-1})	tba	**-0.301**	-0.161	-0.062	**-0.282**

High correlation values are marked in bold.

For the butterfly functional composition, global test of fourth-corner analysis revealed highly significant relationships with landscape environmental variables (Model 2, p<0.01), and species traits (Model 4, p<0.05). I observed significant (at p<0.05) relationship of the six butterfly traits, viz. wingspan, elevational range, biogeographic range size, egg shape, larval host plant type and Defence mechanism/Colouration with the different environmental variables (most interactions with elevation, pcc, tree species richness, tree density, MAT, and MAP but only one with tree basal area) (Fig. 6.10), resembling the results obtained in RLQ analysis (Fig. 6.9). However, the significant associations for butterflies were less (6 out of 14 traits) when compared to birds (14 out of 17 traits).

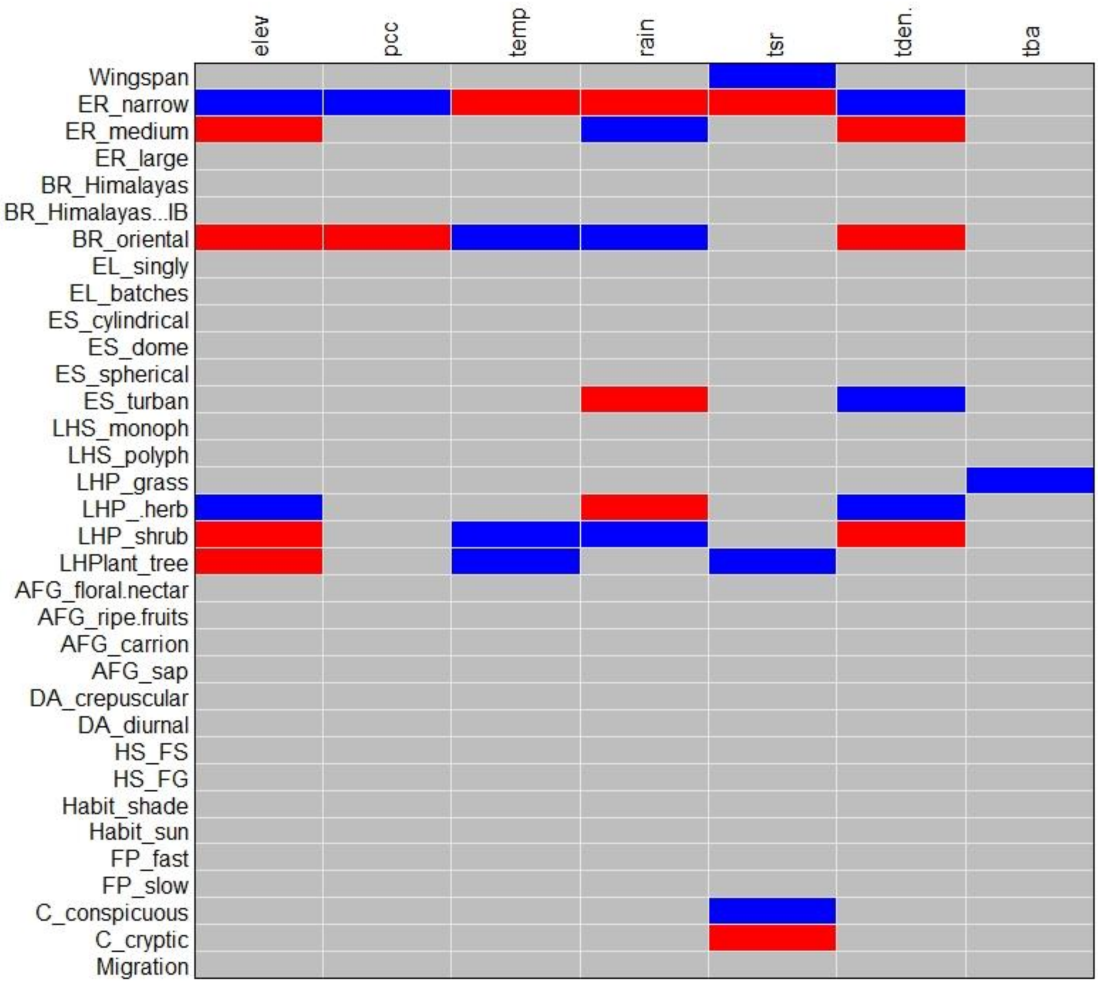


Fig. 6.10. Results of Fourth corner analysis showing the significant ($p<0.05$) positive (blue) and negative (red) correlation of different butterfly functional traits with the landscape variables along agroecosystem-forest gradient of Sikkim, Eastern Himalaya. The details of codes for functional traits are given in Table 6.2, while, environmental variables are detailed in Table 6.5 and Appendix A.

6.4. Discussion

6.4.1. Bird functional diversity

For bird communities in the present study, FRic correlated positively with species richness and Shannon-Wiener diversity. Past studies in bird communities have also found positive relationship of FRic with species richness (Flynn *et al.*, 2009; Mouchet *et al.*, 2010; Ding *et al.*, 2013; Barbaro *et al.*, 2014; Lee and Carrol, 2018) and FDis (Echeverri *et al.*, 2019), or of FDis with Shannon-Wiener diversity (Hanspach *et al.*, 2015). Such positive relationships between TD and FD results in positive biodiversity-ecosystem functioning relationship (Cardinale *et al.*, 2012), but magnitude will depend on FD values (Cadotte *et al.*, 2011).

The FRic declined along the agroecosystem-forest gradient with highest value in MOAS, whereas FDis and TD were significantly highest in LCAS. Past studies have reported similar spatial mismatch between TD and FD in birds, plants and other taxa (Flynn *et al.*, 2009; Monnet *et al.*, 2014; Rurangwa *et al.*, 2021). Bird FRic and species richness strongly depends on habitat quality, enhanced by increased patch size having moderate or low level of basal area and large or inter-connected habitat patches in the landscape (Bovo *et al.*, 2018; Lee and Carrol, 2018). The low to moderate level of basal area facilitates formation of heterogeneous vegetation structure with high understorey cover and enhances the range of traits to persist in the environment, thereby leading to increased functional dissimilarity among coexisting birds species (Lee and Carrol, 2018). The relationship of FRic was weak (non-significant) negative with elevation, pcc, tree density, tree basal area, but positive with tree species richness, MAT and MAP, which explains its declining pattern along the agroecosystems-forest gradient. Bird FRic (like species richness) often depends on various habitat features such as forest types and vertical vegetation diversity (Sitters *et al.*, 2016a), landscape heterogeneity and diversity (Barbaro *et al.*, 2014, 2017; Lee and Martin, 2017), and tree FD (Barbaro *et al.*, 2019). The bird FRic have also been found to depend positively with NPP (Pellissier *et al.*, 2018), plant FRic (Dehling *et al.*, 2014), proximity to forest and proportion of ‘wildlife-friendly’ habitat (Cannon *et al.*, 2019), forest cover (for forest birds: Matuoka *et al.*, 2020), habitat heterogeneity (García-Navas *et al.*, 2020), patch size (for Frugivorous birds: Bovo *et al.*, 2017).

However, bird FRic declines with elevation (Dehling *et al.*, 2014; García-Navas *et al.*, 2020), and land-use change (Ibarra and Martin, 2015; Bregman *et al.*, 2016).

Contrary to pattern for FRic, Forests showed highest value of FEve, increasing along the agroecosystem-forest gradient. The agroecosystem-forest gradient in the present study corresponded with significant increase in tree basal area and pcc, but decline (near-significant) in tree species richness. Bird FEve was structured significant positively by pcc and tree basal area, whereas, significant negatively by tree species richness, therefore explaining the observed pattern and underlying their role in shaping resource utilization. The FEve depicts the abundances of species in functional space, and help to understand resource utilization (whether under- or over-utilized), which in turn influences productivity and vulnerability to forest disturbances like land-use change, and invasion (Mason *et al.*, 2005; Mouchet *et al.*, 2010; Sayer *et al.*, 2017; Lee and Carrol, 2018). Such decline in FEve along land-use gradient of Sikkim is in accordance with some past studies elsewhere (Edwards *et al.*, 2013; Prescott *et al.*, 2016). Assuming that resources are evenly distributed, Bird communities used resources more efficiently within forests and LCAS than in relatively simplified agroecosystems (i.e., FAS and MOAS) in the present study (Mason *et al.*, 2005). Past studies have found bird FEve to increase with landscape diversity, structural complexity, forest and grass cover (Barbaro *et al.*, 2014, 2017; Sayer *et al.*, 2017), but decline with MAP (Barbaro *et al.*, 2019). It also depends on horizontal vegetation diversity, structural heterogeneity and forest types (Sitters *et al.*, 2016a, 2016b), and climatic conditions (García-Navas *et al.*, 2020).

The FDiv measures the spread of abundance along a functional trait axis, and degree of niche differentiation, with high FDiv in a community indicating that abundant species are far from the center of functional space, and consequently implying high niche differentiation, efficient resource use, and enhanced ecosystem functioning (Mason *et al.*, 2005; Villéger *et al.*, 2008; Mouchet *et al.*, 2010). In the present study, highest FDiv in LCAS, indicates likely high niche differentiation and low resource competition, whereas, reduced FDiv in the other two IFS and Forest ecosystems likely implies low niche differentiation and reduced resource utilization efficiency in the bird communities. This observed pattern is in agreement with past reports of reduced bird FDiv in overstocked forests than moderately stocked forests (Lee and Carrol, 2018), or in forest interior than forest edge (Barbaro *et al.*, 2014). The bird FDiv correlated weak positively only with tree species richness, but negatively with elevation and pcc. The FDiv in birds has been reported to respond positively to forest cover, plant diversity (in dry forest), vertical vegetation diversity, compositional landscape heterogeneity but negatively to amount of agricultural lands (Sitters *et al.*, 2016a, 2016b; Lee and Martin, 2017; Matuoka *et al.*, 2020). It also depends on elevation (García-Navas *et al.*, 2020), as well as landscape heterogeneity and grass cover (Barbaro *et al.*, 2017).

FDis ‘the multivariate measure of trait dispersion’ was significantly highest in FAS, followed by MOAS indicating that bird community is composed of functionally different from the mean trait composition, whereas, decline of FDis in LCAS and Forests indicates low bird species dispersion from the functional space centroid (Laliberté and Legendre, 2010). Similar pattern of increased bird FDis, coupled with declined ecosystem function has been found along land-use change and intensification gradient in forest ecosystems (Sayer *et al.*, 2017) and agroecosystem-dominated landscapes (Prescott *et al.*, 2016). The FDis correlated positively with tree species richness, but negatively with elevation, pcc and tree basal area. Bird FDis increases with landscape diversity (Barbaro *et al.*, 2014), local tree cover and landscape context (Echeverri *et al.*, 2019), but decline with forest cover (Matuoka *et al.*, 2020), and distance from nearest forest and land-use intensity (for Frugivores; Cottee-Jones *et al.*, 2015). It also depends on elevation (Montaño-Centellas *et al.*, 2020), vertical vegetation diversity and forest types (Sitters *et al.*, 2016a; Pedley *et al.*, 2019).

CWM_bodymass indicates the community-weighted mean of body mass for birds. The body mass, an important trait of birds is strongly related to its metabolic rate, lifespan, resource utilization trend, trophic level and extinction vulnerability, persistence against land-use change and their contribution to various avian-mediated ecosystem functions (Ding *et al.*, 2013, Luck *et al.*, 2013; Si *et al.*, 2017; Tinoco *et al.*, 2018). In the present study, CWM_bodymass for total bird communities correlated significant positively with MAT and MAP but negatively with elevation. CWM_bodymass of birds has been found to respond positively with grass cover, MAT, spider (as prey) abundance but negatively with MAP (Barbaro *et al.*, 2017, 2019). The CWM-bodymass in the present study was highest in FAS (100.5 g), followed by Forest (89.11 g), which declined in MOAS (69.35 g) and LCAS (47.66 g). CWM_bodymass declines with deforestation/patch size reduction for land use sensitive guilds such as cavity-nesting (Ibarra *et al.*, 2017) and frugivorous birds (Bovo *et al.*, 2017). However, there are also reports of decline in body mass of birds with increasing forest cover (Matuoka *et al.*, 2020), which together with predominance of Passerines and mixed flocking insectivorous species (Matuoka *et al.*, 2020; Zhang *et al.*, 2020b) explains the declined CWM_bodymass for birds in LCAS in this study.

LCAS had high FD (i.e., FEve and FDiv) of birds, even greater/comparable to adjacent Forest ecosystem and greater than other two IFS types (MOAS and FAS: having higher FRic or FDis). Thus, signifying the role of LCAS (along with adjacent Forest ecosystem) in enhancing BEF relationship and maintaining crucial ecosystem functioning of (i) invertebrate pest control, (ii) pollination, and (iii) ecosystem engineering (Şekercioğlu, 2006, 2012; Crowder *et al.*, 2010; Maas *et al.*, 2013; Bregman *et al.*, 2016; Cussera *et al.*, 2016; Barbaro *et al.*, 2017, 2019; Ibarra *et al.*, 2017; Maseko *et al.*, 2019) which are most vulnerable to land-use change. The present observation is linked to the positive habitat features (high tree density, tree basal area, pcc, elevation; proximity to PAs), optimum climatic features, and shade tree diversity resulting into greater landscape and habitat heterogeneity(Fischer *et al.*, 2007; Luck *et al.*, 2012, 2013; Maas *et al.*, 2013; Hanspach *et al.*, 2015; Lee and Martin, 2017; Goded *et al.*, 2019). Conversely, MOAS and FAS also contributed to other three avian-mediated ES. MOAS and FAS sustained seed disperser birds primarily due to high diversity and density of fruiting trees including *Ficus* spp., and *Citrus* spp. in these wildlife-friendly agroecosystems (Corlett, 1998; Cottee-Jones *et al.*, 2015; Bregman *et al.*, 2016; Mangan *et al*., 2017; Martínez and García, 2017; Bovo *et al.*, 2018). They also sustained nutrient depositing birds due to their location at relatively low elevation and proximity to the banks of rivers and mountain streams, thereby providing habitats to a large number of aquatic birds (Şekercioğlu, 2006, 2012; Fujita and Kameda, 2016). The birds providing scavenging and vertebrate pest control service were promoted in these relatively simplified agroecosystems, due to greater proximity to human-habitations, availability of carcasses/ wastes, vertebrate prey like rodents, and socio-cultural practices of local communities (Kross *et al.*, 2012, 2016;

Cuthbert *et al.*, 2016; De Vault *et al.*, 2016). Thus, FD interacted with landscape heterogeneity to drive the potential for the different avian-mediated ES.

Land-use change and intensification causes loss of FD, and consequently declined ecosystem resilience through selective extinction of certain body mass or land use sensitive groups (e.g., large frugivores and forest specialist insectivores) and declined community specialization in birds (Edwards *et al.*, 2013; Ibarra and Martin, 2015; Cottee-Jones *et al.*, 2015; Bregman *et al.*, 2016; Sayer *et al.*, 2017). Among other factors, FD in bird communities also depends on MAP/rainfall (Seymour *et al.*, 2015; Barbaro *et al.*, 2019), and vegetation structure (Seymour *et al.*, 2015).

Based on niche theory, the higher FRic but low FEve in the simplified agroecosystems can be viewed as evidence of limiting similarity affecting the bird communities of MOAS and FAS, whereas relatively reduced FRic but higher FEve in the complex ecosystems implies the evidence of environmental filtering structuring the bird communities of LCAS and Forests (Mouchet *et al.*, 2010; Pakeman, 2011).

6.4.2. Butterfly functional diversity

In the present study, butterfly FRic was highest in MOAS followed by FAS and declined along the agroecosystem-forest gradient of Sikkim. In contrast, FDis was significantly highest in LCAS, whereas FEve and FDiv were highest in Forests. The IFS (MOAS, LCAS) of Sikkim retained not only high TD (species richness and Shannon-Weiner diversity) but also FD (FRic, FDis) of butterflies. This firstly relates to agroecosystems under wildlife-friendly management in Sikkim, that sustained high butterfly TD (Mcneely and Schroth, 2006; Tscharntke *et al.*, 2008; Goded *et al.*, 2019), FD (Hanspach *et al.*, 2015; Goded *et al.*, 2019), and butterfly-mediated ES (Losey and Vaughan, 2006; Cussera *et al.*, 2016; Kehimkar, 2016). Small farm size, landscape heterogeneity, high tree species richness, agrodiversity (including non-crop elements) provisioning larval host plants, year round flower nectar and ripened/rotting fruits also positively affected the FD of butterflies in the study (Sharma *et al*., 2016a; Goded *et al*., 2019).

I found a strong significant positive relationship of FD (FRic, FDis) with TD (species richness and Shannon Wiener diversity) for butterflies in the present study. This can be attributed to presence of traditionally managed agroecosystems (Hanspach *et al*., 2015), and old-growth secondary forests (Sayer *et al*., 2017) in the study landscapes. The selection effect implies that wide range of functional trait values are likely to be captured due to enhanced local species pool size, and hence, could explain the strong relationship between FD and TD in butterflies (Huston, 1997; Mayfield *et al*., 2010; Cadotte *et al*., 2011). A positive linear relationship is found when species converge into relatively discrete functional strategies, with species richness exceeding FRic, whereas, a proportional relationship between species richness and FRic would result from random or uniform distribution of species in functional space (Diaz and Cabido, 2001).

The observed relationship of butterfly FRic with the different environmental variables is related to butterfly communities being most species rich and abundant in ecosystems MOAS and FAS with high MAT and MAP (Acharya and Vijayan, 2015) and plant/tree species richness (Chettri, 2015; Riva *et al*., 2020). Conversely, the declined FRic in LCAS and Forest ecosystems is due to their location at relatively higher elevation (Acharya and Vijayan, 2015; Chettri, 2015), with high pcc, tree density, and tree basal area (see Fig. 4.1 in Chapter 4 of this thesis).

Land-use intensification leads to decline in FD and ecosystem resilience in butterflies (Kühsel and Blüthgen, 2015), and plants (Laliberté *et al.*, 2010). However, wildlife-friendly agroecosystems, as in the present study helped stabilize FD in butterflies and plants (Hanspach *et al.*, 2015; Goded *et al.*, 2019). In the present study, the butterfly FDis showed only weak negative correlation with tree species richness, basal area, and density. The butterfly FDis responds positively with average vegetation height and understorey cover (Aguirre-Gutiérrez *et al.*, 2017), landscape compositional heterogeneity (i.e., floral diversity) and landscape configurational heterogeneity (i.e., vegetation structure diversity) (Paradis, 2020), but negatively with proportion of unsuitable habitats (Aguirre-Gutiérrez *et al.*, 2017), which explains its significantly highest value in LCAS.

CWM_Wingspan indicates the community-weighted mean of wingspan for butterflies. The average wingspan, a proxy for dispersal ability (Sekar, 2012), is one of the most important determinants of persistence of butterflies against land-use change (Öckinger *et al.*, 2010; Kuussaari *et al.*, 2014). The CWM_Wingspan tended to decline along the agroecosystem-forest gradient in the Eastern Himalaya, due to its significantly positive relationship with MAT and MAP, but negative with elevation and tree density. The study signifies the role of IFS (particularly MOAS) in harbouring butterflies with significantly large average wingspan, and can thus mitigate the adverse effects of land-use change on butterflies in the Eastern Himalaya (Barbaro and Van Halder, 2009; Perović *et al.*, 2015).

The butterfly communities are strongly linked with plants for host plants at larval stage and floral nectar at adult stages (Corlett, 2004). Butterflies are important pollinators in diverse ecosystems including agroecosystems since they can pollinate flowers using both olfactory and visual cues, and carry pollens to long distances (Balasubramanian, 1990; Andersson *et al.*, 2002; Borges *et al.*, 2003; Cussera *et al.*, 2016). Their role in pollination have been further recognized due to their ability to enhance pollination services and crop yield (Kremen *et al.*, 2007; Garibaldi *et al.*, 2013; Zou *et al.*, 2017) amid the global decline in bee pollinators (Potts *et al.*, 2010).

Therefore, high TD and FD of butterflies in IFS and adjoining Forests ecosystems will enhance BEF relationship. and consequently help maintain crucial pollination services in the heterogeneous landscape (Cardinale *et al.*, 2012; Hanspach *et al.*, 2015; Goded *et al.*, 2019) formed by mosaic of IFS and adjacent Forest ecosystems in Sikkim. Except for one insectivorous butterfly species, namely Forest Pierrot *Taraka hamada* whose larva feeds on scale insects, I only observed butterflies contributing pollination services (Haribal, 1992; Jain *et al.*, 2016; Kehimkar, 2016). Nonetheless, the butterflies also indirectly contribute to natural insect pest control services in the natural forest and agroecosystems. They constitutes an important component of the food chain since they provision caterpillar at larval stage(Hammond and Miller, 1998; Losey and Vaughan, 2006) and at adult stage they are preyed upon by the natural predators such as insectivorous birds, spiders, etc. (Kehimkar, 2016).

The higher FRic but low FEve in the simplified agroecosystems can be viewed as likely evidence of limiting similarity affecting the butterfly communities of MOAS and FAS, whereas, relatively reduced FRic but higher FEve in the complex ecosystems likely implies the evidence of habitat/environmental filtering in the butterfly communities of LCAS and Forests as per niche theory (Mouchet *et al.*, 2010; Pakeman, 2011; Dong *et al.*, 2019).

6.4.3. Drivers of bird functional composition

I observed significant overall association between bird species traits and landscape variables in the RLQ analysis, and traits showed clear segregation along the agroecosystem-forest gradient. This trait segregation was structured by the habitat and environmental variables since the first axis of RLQ analysis (which explained 91.4% variance) correlated negatively with elevation, pcc, tree density and basal area but positively with MAT, MAP and tree species richness. Similar to this study, past studies have found strong effects of tree basal area and tree density (Maseko *et al.*, 2019; Nava-Díaz *et al.*, 2020), woody vegetation cover/canopy cover (Barbaro and Van Halder, 2009; Sitters *et al.*, 2016a), and land-use/management intensity (Edwards *et al.*, 2013; Hanspach *et al.*, 2015) on bird functional composition. Bird functional composition is also by determined by vegetation structure and composition, understorey cover, landscape heterogeneity, and distance to forest (Hanspach *et al.*, 2015; Sitters *et al.*, 2016a; Chen *et al.*, 2019; Nava-Díaz *et al.*, 2020). This study also reports (previously unreported) strong negative effect of elevation, but positive effect of MAT, MAP and tree species richness on bird functional composition.

The traits specific to generalists functional groups of birds were associated more with simplified ecosystems of MOAS and FAS, and consequently provisioning avian-mediated ES of seed dispersal, waste disposal, nutrient deposition. Contrastingly, traits specific to specialists or land-use change vulnerable functional groups of birds preferred the complex ecosystems of LCAS and Forests which had greater habitat and landscape complexity, contributing ecosystem functioning of insect pest control, pollination and ecosystem engineering. The bird community with traits specific to the conservation concern species preferred the LCAS and Forests.

I found that there was environmental filtering of bird functional traits and consequently, the associated avian-mediated ES along the agroecosystems-forest gradient in the Eastern Himalaya. The results from the fourth-corner analysis further showed that bird functional composition was strongly shaped by the environmental variables and species' functional traits. I observed significant relationship for almost all bird functional traits (14 out of 17) with at least one of the environmental variables, further confirming results from the RLQ analysis. Thus, although birds can thrive in simplified agroecosystems (i.e., in MOAS and FAS), however, Forests are irreplaceable for sustaining the specialists/conservation concern birds and associated ES providers viz., insect pest control, pollination and ecosystem engineering in the Eastern Himalaya and in preventing trait filtering (Barbaro and Van Halder, 2009; Edwards *et al.*, 2013; Barnagaud *et al.*, 2014; Hanspach *et al.*, 2015; Tinoco *et al.*, 2018; Chen *et al.*, 2019; Matuoka *et al.*, 2020; Zhang *et al.*, 2020b).

6.4.4. Drivers of butterfly functional composition

The butterfly species traits showed significant overall association with environmental variables, with first two axes accounting for 97.6% variance in the RLQ analysis. The first axis of RLQ analysis matched the agroecosystem-forest gradient from simplified agroecosystems (MOAS, FAS) to more complex ecosystems (LCAS, Forests) of the Eastern Himalaya.

In the present study, the RLQ analysis for butterflies showed that the first axis had strongly positive relationship with MAT, MAP, tree species richness but negative relationship with elevation, tree density, whereas, second axis had negative relationship with pcc and tree basal area. Butterfly functional composition depends on larval host plant richness and elevation (Pavoine *et al.*, 2014), canopy cover/woody vegetation cover and shrub cover (Barbaro and Van Halder, 2009; Hanspach *et al.*, 2015). The functional composition of butterflies is also structured by land-use type/management intensity, proportion and connectivity of cropland/woodland in the landscape, landscape heterogeneity (Cleary *et al.*, 2009; Barbaro and Van Halder, 2009; Pavoine *et al.*, 2014; Hanspach *et al.*, 2015; Perović *et al.*, 2015; Archaux *et al.*, 2018). The present study additionally found strong positive effects of MAT, MAP, tree species richness, and negative effects of tree density, tree basal area on butterfly functional composition.

Butterfly species preferring the simplified agroecosystems (MOAS, FAS) had traits such as widespread biogeographical range, medium elevation range, large wingspan, conspicuously coloured, larval host plants from tree and shrub, or laying cylindrical-shaped eggs, and few typical species such as *Euploea mulciber, Papilio krishna,* etc. Similarly, the diurnal, polyphagous, habitat generalist, migratory, forest canopy or gap- tolerant species with large elevation range, fast or erratic flight, high reproductive potential, or feeding on floral nectar-at adult also preferred MOAS and FAS, e.g., *Catopsilia pomona*, *Pieris canidia*. These butterfly traits are specific to generalist or non-conservation concern species. On the contrary, species linked with the complex ecosystems (LCAS and Forests) had traits viz., narrow or small biogeographical and elevation range sizes, and cryptic colouration e.g., *Heliophorus moorei*, *Dodona adonira*. Similarly, crepuscular, monophagous, forest understorey butterfly species with weak flight, low reproductive potential, laying dome-shaped egg, grass-feeding larva, rotten fruits- or sap- feeding adults, were also linked to LCAS and Forests and some typical species included *Mycalesis francisca, Hasora badra*. These traits of butterflies are specific to specialist or conservation concern (threatened/range-restricted) species.

Thus, the traits specific to generalist butterflies corresponded with simplified ecosystems of MOAS and FAS, whereas, traits specific to specialists or land-use change vulnerable functional groups preferred the complex ecosystems of LCAS and Forests. The environmental filtering of butterfly traits along the agroecosystem-forest gradient in the present study was structured by the habitat and environmental variables (Steffan-Dewenter and Tscharntke, 2000; Öckinger *et al.*, 2010; Cleary *et al.*, 2009; Börschig *et al.*, 2013; Pavoine *et al.*, 2014; Hanspach *et al.*, 2015; Archaux *et al.*, 2018). However, based on fourth-corner analysis significant relationships of functional traits with the environmental variables were comparatively fewer for butterflies (6 out of 14 traits) than birds(14 out of 17 traits), thereby signifying the role of not only Forests and LCAS but also MOAS and FAS in maintaining functional composition of butterflies. In fact, the fourth-corner analysis presented highly significant relationships of butterfly functional composition with both landscape environmental variables, and species traits.

6.5. Conclusion

For birds, the declining pattern of FRic along the agroecosystem-forest gradient did not mirror the pattern of TD nor FDis, FEve and FDiv. On the contrary, for butterflies, FRic, FDis, and TD, showed similar declining pattern along the agroecosystem-forest gradient, but contrasted with the pattern of FDiv and FEve. Thus, the use of complementary FD indices allowed in correctly inferring that higher FD did not result into enhanced ecosystem functioning (specifically for birds) in the simplified agroecosystems (Montaño-Centellas *et al.*, 2020). The significant positive relationship of FRic and species richness for both observed and simulated communities will result in positive BEF relationship (Cardinale *et al.*, 2012) for birds (in LCAS and Forest) and butterflies (in MOAS, FAS). This implies that further landscape simplification in the region, will not only lead to decline in taxonomic diversity through local extinction and trait filtering but also eventually cause loss of FD, functional composition and disruption of critical ES provisioning in birds (Flynn *et al.*, 2009; Şekercioğlu, 2012; Bregman *et al.*, 2016; Maseko *et al.*, 2019), and butterfly communities (Barbaro and Van Halder, 2009; Hanspach *et al.*, 2015; Kühsel and Blüthgen, 2015; Dainese *et al.*, 2017; Goded *et al.*, 2019).

For both bird and butterfly functional composition, species traits showed significant associations with the environmental variables, and consequently there was environmental filtering of species traits along the agroecosystem-forest gradient. Functional traits specific to generalist or non-conservation concern species of birds and butterflies were preferably linked with simplified agroecosystems i.e., MOAS and FAS. On the other hand, traits specific to specialists or conservation concern functional groups favoured the complex ecosystems of LCAS and Forests for both the studied taxa. The functional composition for birds will be best maintained in the Forests and LCAS, whereas low strength and fewer significant relationship between species traits and environmental variables in case of butterflies implies the importance of MOAS and FAS as well in maintaining functional composition of butterflies. The agroecosystems of Sikkim can sustain not only high TD but also high FD, functional composition and associated ES of birds and butterflies. Further studies covering larger landscape and elevation gradient along with empirical studies on behavioural traits will provide better insights on FD and functional composition of birds and butterflies.

Chapter 7

CONSERVATION CONCERN AND INDICATOR SPECIES OF BIRDS AND BUTTERFLIES IN AGROECOSYSTEMS OF SIKKIM

Moore's Bushbrown *Mycalesis heri heri* Moore, 1857 (WPA Schedule II protected)

7.1. Introduction

The hyperdiverse tropical ecosystems are highly threatened by the anthropogenic land use and climate change leading to negative consequences for biodiversity and ecosystem services (Pandit, 2017; Barlow *et al.*, 2018; Peters *et al.*, 2019). The biodiversity loss has been particularly high amongst the endemic, specialists and threatened species across taxa (Pandit *et al.*, 2007; Flynn *et al.*, 2009; Waltert *et al.*, 2011; Newbold *et al.*, 2015; Pandit, 2017). The resulting biotic homogenization (Gámez-Virués *et al.*, 2016; Aguirre-Gutiérrez, 2020; Rurangwa *et al.*, 2021) has also disrupted biotic interactions such as herbivory, plant-pollinators and frugivory (Montoya and Raffaelli, 2010), as well as delivery of key ES (Peters *et al.*, 2019). Finally the disruptive effects has in turn greatly affected the species very highly specialized niche requirements, narrow diet and habitat breadth, and small range size, which are mainly endemic species or those threatened with extinctions (Koh, 2007; Newbold *et al.*, 2013; Bregman *et al.*, 2016).

Conservation strategies of land-sparing (through networks of PAs separated from intensive agriculture), have been the most preferred approach for conserving the biodiversity and ES (Phalan *et al.*, 2011). Although largely successful, land-sparing are not the only/most effective strategy (Coad *et al.*, 2019),since large bodies of recent research has reported effectiveness of land-sharing (wildlife friendly agriculture adjacent to forest/PAs) to complement the former (Michael *et al.*, 2016). Past studies from India (Sreekar *et al.*, 2015; Yashmita-Ulman *et al.*, 2016; Chettri *et al.*, 2018b) as well as from elsewhere (e.g., Ohwaki *et al.*, 2007; Bubova *et al.*, 2015; Hanle *et al.*, 2021) have highlighted that agroecosystems when managed under traditional/organic practices, can even retain and protect conservation concern species (range-restricted/endemic, globally/nationally protected species) of diverse taxa such as birds, butterflies, mammals and plants. The presence/retention of such significant species in the agricultural landscape signifies their high conservation potential and reflects the presence of suitable habitat in such ecosystems inviting more attention for its conservation from relevant policy makers (Bubova *et al.*, 2015; Sreekar *et al.*, 2015). Past studies on status and distribution of endemic/range-restricted and threatened birds in the Eastern Himalaya including Sikkim have emphasized the importance of forests (Ganguli-Lachungpa *et al.*, 2007; Acharya and Vijayan 2010; Kandel *et al.*, 2018; Grimmett *et al.*, 2019) but their status in agroecosystems of the region are poorly known. Similarly, except for some report on rediscovery of few conservation concern species of butterflies (Kunte, 2010; Rai *et al.*, 2012; Dewan *et al.*, 2018), there has not been any comprehensive study on the extent of butterflies of conservation concern in Sikkim (but see Chandra *et al.*, 2018) in general and from agroecosystems of the region in specific. Additionally, best conservation strategy for Sikkim Himalaya as well as indicator species of birds and butterflies for long-term monitoring in the region also remains to be evaluated.

The birds and butterflies are the two widely studied indicator taxa of vertebrates (endotherms) and invertebrates (ectotherms), respectively (Kremen *et al.*, 1992; Schulze *et al.*, 2004a; ILTEO, 2015; Herrando *et al.*, 2016). This chapter aims to understand the potentiality of the IFS and the adjoining Forest of Sikkim Himalaya for retention/protection of conservation concern species of birds and butterflies as well as evaluate indicator species and conservation strategy for Sikkim Himalaya. To achieve this, I have framed four objectives. First, I examined the species richness, distribution pattern and extent of bird and butterfly species of conservation concern in different IFS and adjoining Forests ecosystems in present study vis-à-vis Sikkim and India as a whole. Second, I attempted to understand the cross taxon congruence between bird, butterfly and tree diversity. Third, I identified indicator species of birds and butterflies for long-term ecological monitoring in the Eastern Himalaya. Finally, I attempted to identify the best conservation strategy (land-sparing, land-sharing or combination of both) with reference to birds and butterflies in the Eastern Himalaya.

7.2. Materials and methods

7.2.1. Study sites and design

As described in **chapter 3**, the present study covers three representative agroecosystems viz., (MOAS, FAS, and LCAS) and nearby natural Forest (as control) ecosystem identified along a gradient of shade tree diversity within the elevation of 600-2000 m AMSL in Sikkim, Eastern Himalaya, India. I laid 24 transects (6 per ecosystem of 1km length each, spaced ≥1 km apart) and 240 permanent sampling points (10 in each transect, spaced ≥100 m apart) across these four ecosystem.

7.2.2. Bird and butterfly sampling

As **detailed in chapter 4**, the birds were sampled following open-width point count method (Bibby *et al.*, 2000; Raman, 2003; Acharya *et al.*, 2011a; Acharya and Vijayan, 2017) along each transect, and at the 10 pre-established permanent point, I recorded the identity of the birds to the species level based on the standard field guides (Ali, 1962; Grimmett *et al.*, 2011) and their abundance [for 10 minutes, on clear days in the morning hours (06:00 hrs to 09:00 hrs)].

I sampled butterflies following fixed width point count method (Pollard, 1977; Acharya and Vijayan, 2015) along each transect and recorded the identity of butterflies to species level following standard field guides (Wynter-Blyth, 1957; Haribal, 1992; Kehimkar, 2008), and their abundances (for five minutes within the 5 m radius plot during 09:00 hrs to 12:00 hrs) at each of the pre-established permanent point (**detailed in chapter 5**).

7.2.3. Conservation concern species

Data on conservation concern species of birds and butterflies were collected from this study (primary data) as well as those collated from different standard literatures (secondary data). The birds were identified as conservation concern species if they were endemic/range-restricted to Eastern Himalaya (Acharya and Vijayan, 2010; Grimmett *et al.*, 2019; BirdLife International, 2020b), or based on their global threatened status in IUCN Red List 2019 (BirdLife International, 2020a; Praveen *et al.*, 2020b), listed in Appendices I, II of CITES (UNEP-WCMC, 2018), and national-level threatened status as per Indian wildlife protection Act 1972 Schedule I (WPA 1972) (Anonymous, 2010). Similarly, butterfly species under conservation concern were identified based on their protection status in India Red Data Book of butterflies (Gupta and Mondal, 2005), WPA 1972 Schedule I, II, IV (Anonymous, 2010), and the CITES Appendix II (UNEP-WCMC, 2018).

7.2.4. Data analysis

7.2.4.1. Conservation concern species

I assessed the relative contribution of conservation concern species to total bird and butterfly community in the present study, followed by comparison with those in the state of Sikkim and India.

7.2.4.2. Cross taxon congruence

I assessed the cross taxon congruence of birds (described in Chapter 4), butterflies (discussed in Chapter 5) and trees in the present study by running correlation tests for species richness and Shannon-Wiener diversity following Chettri (2010). The trees were sampled (by measuring stems>20cm gbh) following quadrat method (size 20m*20m at each permanent point; 10 per transect) to quantify tree species richness (tsr) for each transects (discussed at detail in Chapter 4). Therefore, I assessed cross-taxon congruence among the three taxonomic groups along the agroecosystem-forest gradient to examine the extent to which these three taxa can act as surrogate for each other, and consequently their potential in conservation planning and management.

7.2.4.3. Indicator value analysis

Species indicator value (IndVal) analysis was ran to evaluate the association of bird and butterfly species to single ecosystem as well as group of ecosystems (De Cáceres *et al.*, 2010) using ‘multipatt’ function of “indicspecies” package in R 3.1.4 (R Core Team, 2017). The IndVal is the product of two components, specificity value ‘A’ and sensitivity value ‘B’ (Dufrene and Legendre, 1997; De Cáceres *et al.*, 2012). The IndVal analysis helps in identifying characteristic species of birds and butterflies (Barlow *et al.*, 2007a, b; Plexida *et al.*, 2012; Weyland *et al.*, 2014; Herrando *et al.*, 2016) for use in monitoring studies to assess the health of an ecosystem. Finally, I analysed the pattern of identified indicator species of birds and butterflies across the four ecosystems in the present study.

7.3. Results

7.3.1. Conservation concern species of birds

I recorded 221 bird species from the different IFS and adjoining Forests of Sikkim, out of which 8.14 % are conservation concern species. The conservation concern species were protected under IUCN Red list 2019 (2 species each in Near Threatened, and Vulnerable category), CITES (1 species in Appendix I; 11 species in

Appendix II) and WPA 1972 (10 species in Schedule I); and endemic/range-restricted to the Eastern Himalaya (3 species)] totalling 18 species (Table 7.1). In Sikkim, 37 Bird species falls under threatened or near threatened categories of IUCN Red list 2019, of which five are critically endangered: three Endangered, 14 Vulnerable and 15 Near Threatened categories (Praveen *et al.*, 2020b; BirdLife International, 2020a). Contrary to the pattern of total bird community, conservation concern species of birds were most resilient in Forest and FAS, but declined in MOAS and LCAS, showing highest species richness (14 vs. 9 species each) and proportion of protected and endemic/range-restricted species (Table 7.1).

Table 7.1. Comparison of protected and endemic birds (PEBi) observed in the present study, with those of Sikkim, and India. CITES: Convention on International Trade in Endangered Species of Wild Fauna and Flora; WPA 1972: Indian wildlife protection Act 1972. Proportion of PEBi: Percentage of PEBi to total species.

Community parameters	Ecosystem				Total	Sikkim	India
	MOAS	FAS	LCAS	Forest			
Total bird species[1,3,6]	132	157	160	147	221	580	1335
IUCN Red list (Critically Endangered)[7]	-	-	-	-	-	5	17
IUCN Red list (Endangered)[7]	-	-	-	-	-	3	21
IUCN Red list (Vulnerable)[7]	-	2	-	1	2	14	63
IUCN Red list (Near Threatened)[7]	2	-	-	-	2	15	81
IUCN Red list (Data Deficient)[7]	-	-	-	-	-	0	3
Threatened and Near threatened[7, 8]	2	2	-	1	4	37	182
CITES (Appendix I)[4,7]	-	1	-	-	1	-	23
CITES (Appendix II)[4,7]	7	7	5	8	11	-	137
WPA 1972 (Schedule I)[2,7]	5	7	3	7	10	-	102
Endemic[1,5,7, 9]	2	3	3	3	3	10	73
Total PEBi species	9	14	9	14	18	-	323
Proportion of PEBi (%)	6.82	8.92	5.63	9.52	8.14	-	24.19

Note: [1]Acharya and Vijayan, 2011a; [2]Anonymous, 2010; [3]Mandal *et al.*, 2018; [4]UNEP-WCMC, 2018; [5]Grimmett *et al.*, 2019; [6]Praveen *et al.*, 2020a; [7]Praveen *et al.*, 2020b; [8]BirdLife International, 2020a; [9]BirdLife International, 2020b.

Half of the conservation concern species of birds were insectivores and frugivores/granivores (dominated by forest specialists), which were well represented in FAS and Forest (8 species each), followed by LCAS (6 species), but declined in MOAS (only 3 species). The other half constituting the carnivores/piscivores (dominated by openland/ forest generalists), were more speciose and abundant in Forest, FAS and MOAS, (6 species each), but declined in LCAS (only 3 species).

7.3.2. Conservation concern species of butterflies

Out of the 268 butterfly species recorded in the present study, around one-fifth (53 species) are conservation concern species comprised of two species protected under CITES Appendix II, and 51 species under WPA 1972 (9, 33 and 9 species respectively in Schedule I, II and IV).

Table 7.2. Comparison of protected butterfly species (PB) observed in the present study, with those of Sikkim and India. CITES: Convention on International Trade in Endangered Species of Wild Fauna and Flora; WPA 1972: Indian wildlife protection Act 1972. Proportion of CCS: Percentage of CCS to total species

Community parameters	**Ecosystem**				**Total**	**Sikkim**	**India**
	MOAS	FAS	LCAS	Forest			
Total butterfly species [1,5,6]	187	180	157	170	268	690	**1328**
WPA 1972 (Schedule I) [3,4]	6	1	4	4	9	38	128
WPA 1972 (Schedule II) [3,4]	20	15	17	18	33	138	303
WPA 1972 (Schedule IV) [3,4]	6	5	4	4	9	13	19
WPA 1972 (Schedule I, II and IV) [3,4]	31	21	25	26	51	189	450
India Red Data Book of butterflies [2]	6	1	4	4	9	38	128
CITES (Appendix II)[7]	2	1	-	2	2	4	6
Total PB species	33	22	25	28	53	191	454
Proportion of PB (%)	17.65	12.22	15.92	16.47	19.78	27.68	33.19

Note: [1]Haribal, 1992; [2]Gupta and Mondal, 2005; [3]Kunte, 2008; [4]Anonymous, 2010; [5]Varshney and Smetacek, 2015; [6]Kehimkar, 2016; [7]UNEP-WCMC, 2018.

The nine WPA 1972 Schedule I species are also protected under India Red Data Book of butterflies. Out of four CITES Appendix II butterflies found in Sikkim, the present study reports two species(*Troides aeacus aeacus*, *Troides helena cerberus*), while Kaiser-I-Hind and Bhutan Glory (not observed in present study) are also protected under WPA Schedule II. These conservation concern species of butterflies observed in the present study in turn represented 27.75% of the total protected butterfly species found in Sikkim and >11% reported from India (Table 7.2). The forest specialists (45 species) and monophagous (14 species) dominated the protected butterflies in the present study. Similar to the total butterfly community, MOAS were the most resilient ecosystem for conservation concern species of butterflies, showing highest species richness and proportion of WPA 1972 protected species closely followed by Forest and LCAS (Table 7.2). Family-wise analysis of the WPA 1972 species revealed that they were dominated by Nymphalidae, followed by Lycaenidae (constituting about 80% of total federally protected butterflies), resembling the patterns of Sikkim and India (Table 7.3).

Table 7.3. Family-wise comparison of butterflies protected under WPA 1972 observed in the present study, with those of Sikkim and India. Species richness (Species) and percentage coverage (Prop.) are also shown. [1]Kunte, 2008; [2]Anonymous, 2010.

Family	**Present study**		Sikkim		**India**[1,2]	
	Species	**Prop.**	Species	Prop.	**Species**	**Prop.**
Papilionidae	2	3.9 %	14	7.41%	35	7.8%
Pieridae	1	2.0 %	9	4.76%	31	6.9%
Nymphalidae	30	58.8 %	101	53.44%	206	45.8%
Lycaenidae	11	21.6 %	52	27.51%	161	35.8%
Riodinidae	2	3.9 %	3	1.59%	4	0.9%
Hesperiidae	5	9.8%	10	5.29%	12	2.7%
Total WPA 1972	51	100.0%	189	100.0%	450	100.0%

7.3.3. Cross taxon congruence

Analysis of cross-taxon congruence revealed non-significant weak negative relationship between birds and butterflies in case of both species richness and Shannon-Wiener diversity (Fig. 7.1a, b). However, relationship of species richness with tree species richness, was non-significant negative for birds (Fig. 7.1c), whereas, significant positive for butterflies (Fig. 7.1d).

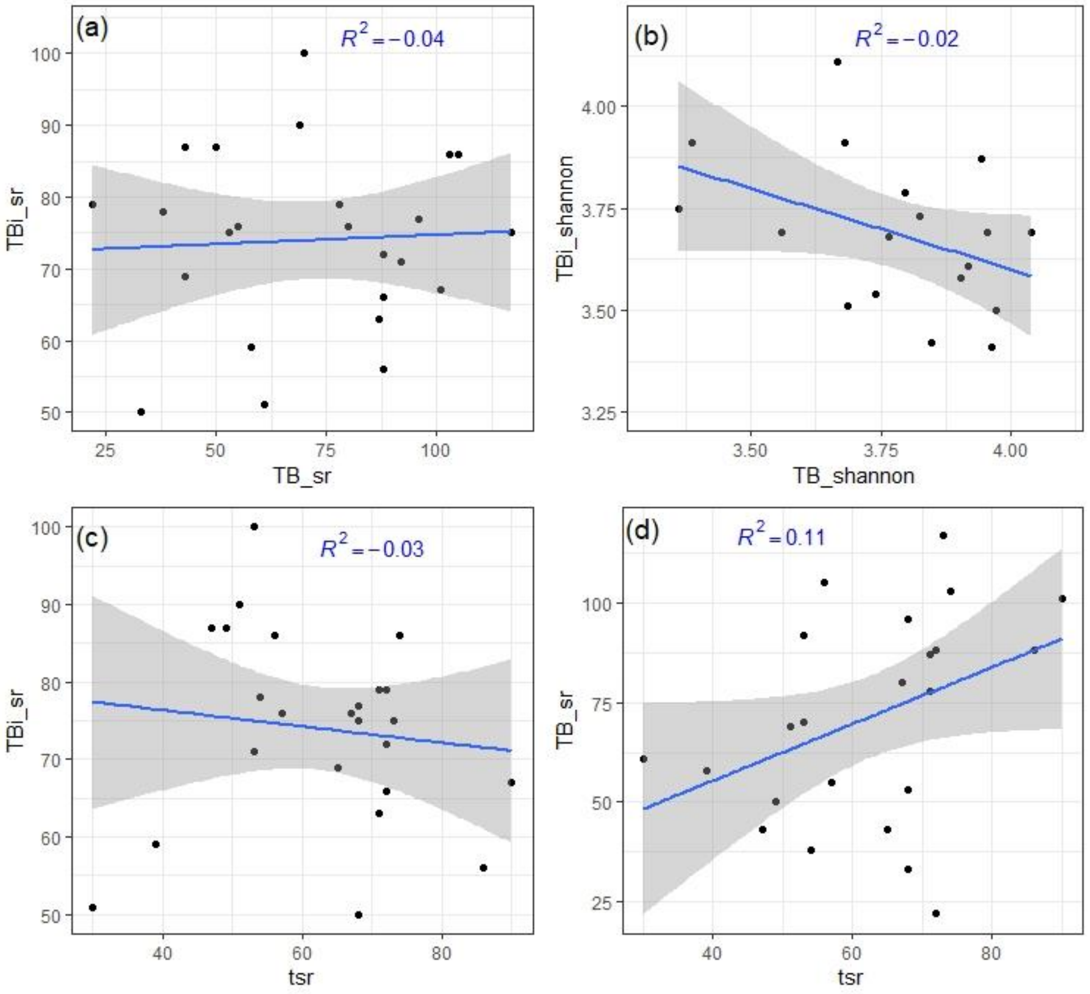


Fig. 7.1. Relationship of species richness (sr) and Shannon-Wiener diversity (shannon) between birds (TBi) and butterflies (TB) (a-b), and with tree species richness (tsr) (c-d) along agroecosystem-forest gradient of Sikkim, Eastern Himalaya.

7.3.4. Indicator and ecosystem exclusive species of birds and butterflies

7.3.3.1. Indicator and ecosystem exclusive bird species

The study identified 36 indicator bird species for specific/group of ecosystems, of which 28 species were associated with LCAS, far exceeding other two IFS (14, 16 species) and Forest (15 species). Nine species were associated with single ecosystem (MOAS: two species; FAS: one species; LCAS: six species and none for forests), 15 species to group of two ecosystems (eight species among the two of three IFS and Forests shared six with LCAS and one with MOAS) and 12 species to group of three ecosystems (five species were associated with three types of IFS vs. seven species for Forest with the IFS) (Appendix B).

Table 7.4. Patterns of species richness of identified indicator birds along agroecosystem-forest gradient of Sikkim, Eastern Himalaya, India.

Indicator birds	**MOAS**	**FAS**	**LCAS**	**Forest**	**Total**
Total indicator species	16	14	28	15	36
Habitat specialization guilds					
Forest specialist	7	2	16	12	17
Forest generalist	4	3	6	3	9
Grassland-openland	5	9	6	0	10
Land use sensitive guilds					
Protected & Endemic species	0	0	2	1	2
Forest specialist frugivores and nectarivores	0	1	2	2	2
Forest specialist understorey insectivores	2	1	6	6	7
Forest specialist canopy insectivores	5	0	8	4	8

Although, Forest had less indicator birds (15 species), however 12 of them were forest specialist, and only three forest generalist species. Similarly, indicator species of LCAS were dominantly forest specialist (16 species) but also had forest generalist, and openland species. In contrast, other two IFS had reduced (in MOAS: 7 species), or highly reduced (in FAS: 2 species) representation of Forest specialist indicator birds, and were mostly openland or forest generalist species, indicative of relatively disturbed habitat (Table 7.4; Appendix B). The indicator birds were dominated by forest specialist (17 species), that belonged to different land use sensitive guilds: forest specialist frugivores and nectarivores (two species); forest specialist canopy insectivores (8 species); and forest specialist understorey insectivores (7 species). Two conservation concern species which are globally protected under CITES Appendix II namely *Leiothrix argentauris* (Photo plate 7.1F), *Leoithrix lutea* were also among the indicator bird species identified during the present study. These land use sensitive guilds were represented well in LCAS (16 species), and Forest (12 species), but poorly in FAS and MOAS (2-7 species) (Appendix B; Table 7.4).

Sixty-one bird species were exclusively observed in a single type of IFS (11, 22 and 19 species in MOAS, FAS and LCAS, respectively) or in Forest (9 species) (Appendix B; Table 7.5). Among the the ecosystem exclusive birds, forest specialists (13 and 9 species out of 33) and forest generalists (5 each out of 17 species) were dominant in LCAS and FAS, whereas, grassland-openland birds in FAS (8 out of 11 species). About half of the ecosystem exclusive birds belonged to land use sensitive guilds: 13 FSUIBi, 5 FSMSIBi, 4 FSCIBi, 3 FSFNBi and 5 protected species, with dominance of former two in LCAS, whereas, latter three in FAS (Table 7.5). It also included five species protected by IUCN red list under NT (*Gyps himalayensis; Haliaetus humilis* in MOAS), and VU (*Buceros bicornis* in FAS), or by CITES appendix II: *Accipiter nisus* (in Forest) and *Glaucidium radiatum* (in FAS) and three identified indicator species: *Upupa epops* (for FAS), *Machlolophus xanthogenys,* and *Phylloscopus reguloides* (for LCAS) (Table 7.5; Appendix B).

Table 7.5. Patterns of species richness of ecosystem exclusive birds along agroecosystem-forest gradient of Sikkim, Eastern Himalaya, India.

Ecosystem exclusive birds	**MOAS**	**FAS**	**LCAS**	**Forest**	**Total**
Total ecosystem exclusive species	11	22	19	9	61
Habitat specialization					
Forest specialist	7	9	13	4	33
Forest generalist	3	5	5	4	17
Grassland-openland	1	8	1	1	11
Land use sensitive guilds					
Forest specialist understorey insectivores	3	1	7	2	13
Forest specialist midstorey insectivores	0	2	3	0	5
Forest specialist canopy insectivores	1	2	1	0	4
Forest specialist frugivores and nectarivores	0	2	1	0	3
Conservation concern species	2	2	0	1	5

7.3.3.2. Indicator and ecosystem exclusive butterfly species

I identified 15 indicator butterfly species for specific or group of ecosystems out of which 11 species were forest specialist including three monophagous and two WPA 1972 (Schedule II) protected species (Appendix G; Table 7.6). Nine species were associated with single ecosystem (MOAS: 6 species; LCAS: 3 species), one species to group of two ecosystems (MOAS and FAS) and five species to group of three ecosystems (4 species were associated with three IFS types vs. one species for Forest shared with LCAS and FAS). However, eight indicator butterflies identified for the different IFS were also encountered in Forests but less abundantly. The indicator butterfly species were dominated by land use sensitive guilds of forest specialist (11 species), monophagous (3 species) and WPA 1972 protected species (2 species).

Table 7.6. Patterns of species richness of identified indicator butterflies along agroecosystem-forest gradient of Sikkim, Eastern Himalaya, India.

Indicator butterflies	MOAS	FAS	LCAS	Forest	Total
Total indicator species	11	6	8	1	15
Habitat specialization					
Forest specialist	8	3	5	0	11
Forest generalist	3	3	3	1	4
Larval host specificity					
Monophagous	3	1	1	0	3
Polyphagous	8	5	7	1	12
Forest specialist and Monophagous	3	1	1	0	3
Protected species	2	1	1	3	4

Eighty-one butterfly species were exclusively observed in a single type of IFS (24, 21 and 15 species in MOAS, FAS and LCAS, respectively) or in Forest (21 species) (Appendix G; Table 7.7). The ecosystem exclusive butterflies included ~84% forest specialists (68 species), 27.16% monophagous (22 species), and ~31% protected species (25 WPA 1972 protected: 5 species each in Schedule I and IV; 15 in Schedule

II), thus reflecting the dominance of the land use sensitive sub-groups. These land use sensitive butterflies were most speciose in MOAS, whereas, forest generalists (5 out of 13 species) and polyphagous (11 out of 28 species) in FAS, and larval host data-deficient (11 out of 31 species) in Forest. They also included three identified indicator species: *Flos asoka* de Nicéville, 1884 (for MOAS), *Euthalia franciae franciae* Gray, 1846; *Abisara chela chela* de Nicéville, 1886 (for LCAS) (Table 7.7; Appendix G).

Table 7.7. Patterns of species richness of ecosystem exclusive butterflies along agroecosystem-forest gradient of Sikkim, Eastern Himalaya, India.

Ecosystem exclusive butterflies	**MOAS**	**FAS**	**LCAS**	**Forest**	**Total**
Total ecosystem exclusive species	24	21	15	21	81
WPA 1972 (Schedule I)	2	0	2	1	5
WPA 1972 (Schedule II)	5	3	3	4	15
WPA 1972 (Schedule IV)	3	1	0	1	5
Protected exclusive species	10	4	5	6	25
Habitat specialization					
Forest specialist	22	16	13	17	68
Forest generalist	2	5	2	4	13
Larval host specificity					
Monophagous	8	5	4	5	22
Polyphagous	8	11	4	5	28
Data deficient	8	5	7	11	31
Forest specialist and Monophagous	8	3	3	4	18

7.4. Discussion

7.4.1. Conservation concern species of birds and butterflies

The IFS and the adjoining forests supported 38.1 % of total birds (221 out of ~580 species) reported from Sikkim (Acharya and Vijayan, 2011a). It also harboured 18 conservation concern species. Out of the 35 endemic/range-restricted bird species found in the Himalaya, 10 have been reported from Sikkim out of which three species were observed during the present study (Acharya and Vijayan, 2010; Mandal *et al.*, 2018). The three endemic/range-restricted species namely *Actinodura nipalensis, Yuhina bakeri,* and *Phylloscopus cantator* (Photo plate 7.1A-C) were seen in both IFS and Forest ecosystems, with former two species abundantly in LCAS and Forest, whereas, latter in FAS and MOAS. Out of Sikkim's 37 IUCN Red List protected bird species, I observed four species viz., two each of Near Threatened species namely *Gyps himalayensis* and *Haliaetus humilis* encountered only in MOAS as well as Vulnerable species viz., *Buceros bicornis* (Photo plate 7.1D) in FAS, and *Sitta formosa* in FAS and Forest. In addition to a CITES Appendix I species (*Buceros bicornis*), I also observed 10 CITES Appendix II species such as *G. cuculoides* (Photo plate 7.1E) in all IFS and Forest, *G. radiatum* in FAS, whereas, other species like *Leiothrix lutea* and *Leiothrix argentauris* (Photo plate 7.1F), respectively were very abundant or only present in LCAS and Forest. I observed 10 out of 102 WPA Schedule I protected bird species, such as *Pavo cristatus* (Photo plate 4.1C) and *Lophura leucomelanos* in all ecosystems except MOAS, *Spilornis cheela* (Photo plate 4.1E), *Accipiter badius, Accipiter virgatus* in both IFS and Forest whereas, *Accipiter nisus* only in Forest. The PEBi species dominated by carnivores/piscivores as well as insectivores and frugivores/granivores declined respectively in LCAS and MOAS.

The agricultural-dominated landscapes of Sikkim also supported 38.8% of total butterflies (268 out of 690 species) of Sikkim (Haribal, 1992). They were very rich in conservation concern species of butterflies (53 species: 51 species in WPA 1972 Schedule I, II and IV; two species in CITES Appendix II). WPA 1972 Schedule I protected species were either only observed in a particular ecosystem for example *Deudorix epijarbas amatius, Euthalia telchinia* in MOAS, *Symbrenthia silana* in LCAS, *Delias sanaca* in Forest, whereas, others like *Lethe distans* (Photo plate 7.2A), *Papilio clytia clytia* were harboured in both IFS and Forest, as well as *Athyma jina jina* (Photo plate 7.2B) in two IFS. Many WPA 1972 Schedule II protected species were observed in single IFS type e.g., *Sinthusa nasaka amba, Phalanta alcippe alcippoides*, *Libythea lepita lepita* in MOAS, *Mycalesis heri heri, Euripus consimilis*

consimilis, Papilio epycides epycides in FAS, *Euthalia franciae franciae, Tanaecia lepidea lepidea, Lethe visrava* in LCAS, *Charaxes dolon centralis, Hypolimnas misippus, Parasarpa dudu dudu,* while others like *Elymnias vasudeva* and *Melanitis zitenius zitenius* were only seen in Forest. However, other Schedule II protected species like *Poritia hewitsoni hewitsoni* and *Dodona adonira adonira* (Photo plate 7.2C,D)**,** as well as *Lethe serbonis, Jamides elpis pseudodelpis, Euripus nyctelius* were encountered in both IFS and Forest ecosystems, whereas species like *Dodona egeon egeon* (Photo plate 7.2E) only in the IFS. WPA 1972 Schedule IV protected species were restricted to a particular IFS types such as *Pelopidas assamensis, Euploea radamanthus* in MOAS, *Baoris farri* in FAS, or *Pelopidas subochracea subochracea* in Forest, whereas, others such as *Euploea mulciber mulciber* (Photo plate 7.2F), *Euploea core core, Polytremis discreta discreta* were common between Forest and agroecosystems. Additionally, two CITES Appendix II species viz., *Troides aeacus aeacus* and *Troides helena cerberus* were seen in both IFS and Forest.

From the above discussion, it becomes clear that for protection of all conservation concern species, it is necessary to focus on the different IFS types as well as Forest. Therefore, the IFS along with adjacent Forests allowed the total community and conservation concern species of birds (described at detail in Chapter 4), and butterflies (please see Chapter 5 for details), to thrive at different spatial scale (local and landscape) and across seasons. The persistence of bird and butterfly communities, including conservation concern species in the different agroecosystems, were significantly higher (in LCAS for birds; in MOAS for butterflies) or comparable (birds in FAS, MOAS; butterflies in FAS, LCAS) than the adjoining Forests. This high conservation potential of the mosaic landscape of Sikkim could be due to greater landscape and habitat heterogeneity (Mas and Dietsch, 2003; Loos *et al.*, 2014;

Buechley *et al.*, 2015; Katayama *et al.*, 2019; Rurangwa *et al.*, 2021) and lack of chemical pesticide/herbicide application (Ohwaki *et al.*, 2007; Hanspach *et al.*, 2015) in the wildlife-friendly IFS of Sikkim. The availability of suitable foraging and breeding habitats as well as microclimatic refugia also benefited birds (Şekercioğlu *et al.*, 2007; Ghosh-Harihar, 2013) and butterflies (Bobo *et al.*, 2006; Kitahara *et al.*, 2008; Loos *et al.*, 2014). Lack of hunting (large frugivores/granivores such as Hornbill, Pheasants; Raptors, Owls, insectivores, or any bird groups) for domestic use or illegal wildlife trade, unique socio-economic practices influenced by Hinduism and Buddhism also helped protect these conservation concern species in Sikkim (Grimmett *et al.*, 2011, 2019). This high retention of biodiversity including protected and endemic species in the agricultural landscape of Sikkim reflects their high conservation potential, and suitability of land-sharing conservation framework to complement land-sparing in the Eastern Himalaya (Grass *et al.*, 2019). Thus, there is a need for suitable policy interventions by the state government to properly manage mosaic landscapes of the region for effectively conserving the rich biodiversity including conservation concern species (Bubova *et al.*, 2015; Sreekar *et al.*, 2015). The substitution components highly dominated the multiple-site beta diversity for both taxa (detailed in Chapter 4 & 5), further signifying the need for including both IFS and Forest ecosystems in biodiversity conservation framework in the Himalaya.

7.4.2. Cross taxon congruence of birds, butterflies and trees

Based on cross-congruence analysis, I found a near significant positive relationship of butterfly species richness with tree species richness, but failed to detect any significant relationships between birds vs. butterflies or bird species richness vs. tree species richness. This near significant congruence between butterflies and trees agrees with the previous reports of strong and significant cross-taxon congruence among birds, butterflies and tree diversity in the Himalaya (Chettri, 2010) and elsewhere (Ekroos *et al.*, 2013), which depended on habitat, scale and landscape context (Ekroos *et al.*, 2013). Along the Himalayan elevation gradient, species richness of birds, butterflies and angiosperms peaks between 600-1600 m, which overlaps with the highest number of hydro-power dams (Pandit, 2017) and other anthropogenic disturbances. However, there was lack of strong and significant congruence between birds, butterflies and trees along agroecosystem-forest gradient in Sikkim, similar to the observation made for these three taxa along land-use gradient in the tropical region (Kessler *et al.*, 2009). This weak congruence among these three taxa also points towards some form of disturbance in the agricultural landscapes of Sikkim. It also underlines the need for considering these vertebrate and invertebrate faunal taxa as well as angiosperm taxa in monitoring the vulnerable ecosystems of Eastern Himalaya (Kremen, 1992; Chettri, 2010; Herrando *et al.*, 2016).

7.4.3. Indicator and ecosystem exclusive species of birds and butterflies

Out of the 36 indicator bird species identified in the present study, 28 species were for LCAS, which was twice than that of other IFS and Forests. Although, Forests had comparable number of indicator species to that of MOAS and FAS (15 species vs. 14-16 species), but were dominated by Forest specialist. Contrastingly, indicator birds identified for MOAS and FAS were dominated by forest generalist and openland species such as *Acridotheres tristis, Upupa epops, Pycnonotus leucogenis,* and *Delichon nipalense* indicating their relatively disturbed habitat. The 17 forest specialist indicator species were represented well in LCAS (16 species), and Forest (12 species) such as *Machlolophus xanthogenys* (Photo plate 4.1D), *Phylloscopus* *reguloides, Grammatoptila striata, Actinodura strigula, Niltava grandis, Pteruthius aeralatus, Muscicapa ferruginea.* However, FAS and MOAS had very less forest specialist indicator birds (2-7 species) e.g., *Clamator coromandus,* as well as *Alcippe nipalensis, Pterorhinus pectoralis, Niltava sundara, Oriolus traillii* shared with Forest and/or LCAS. The two CITES Appendix II indicator species namely *Leoithrix argentauris* (Photo plate 7.1F) and *Leoithrix lutea* were specific to LCAS and Forest.

Among the 15 indicator butterfly species identified in the present study, only one species (i.e., *Aglais caschmirensis aesis*) was found as indicator of Forests (along with FAS and LCAS) vs.14 species for the three IFS. However, they were dominated by land use sensitive sub-group including 11 forest specialists such as *Euthalia franciae franciae, Flos asoka, Acytolepis puspa gisca, Papilio paris paris, Abisara neophron neophron,* and *Abisara chela chela;* three monophagous species viz., *Lethe mekara mekara, Lethe verma sintica* and *Ypthima asterope mahratta;* and two WPA 1972 protected species viz., *Euploea midamus rogenhoferi* and *Rapala varuna varuna.* Past studies have also reported higher number of indicator birds (Solomou and Sfougaris, 2011; Plexida *et al.*, 2012) and butterflies (Vu, 2007; Herrando *et al.*, 2016; Basset *et al.*, 2017) in the agroecosystems, other human-modified ecosystems or disturbed forests than undisturbed natural forest ecosystems.

The identified indicator species of birds (36 species), and butterflies (15 species) were dominated by the land use sensitive sub-groups of birds (17 forest specialist including two CITES Appendix II protected species), and butterflies (11 forests specialists including three monophagous, and two WPA 1972 Schedule II protected species). Therefore, they are suitable for long-term ecological monitoring program to assess health of threatened ecosystems in the Eastern Himalaya (ILTEO, 2015; Negi *et al.*, 2019). The ecosystem exclusive birds (61 species) were dominated by forest specialists (33 species), and included 5 protected, and 3 indicator species identified for FAS (*Upupa epops*) and LCAS (*Machlolophus xanthogenys, Phylloscopus reguloides*). Similarly, ecosystem exclusive butterflies (81 species) were dominated by forest specialists (68 species), and included 22 monophagous, 25 protected, and 3 indicator species identified for MOAS (*Flos asoka* de Nicéville, 1884) and LCAS (*Euthalia franciae franciae* Gray, 1846; *Abisara chela chela* de Nicéville, 1886). Hence, these ecosystem specific birds and butterflies can complement their indicator species counterpart for future ecological monitoring in the Eastern Himalaya.

7.5. Conclusion

The traditionally managed and organic agroecosystems along with adjoining Forests showed great potential for conservation of birds and butterflies, including land use sensitive guilds, and conservation concern species (Acharya and Vijayan, 2011a; Haribal, 1992). Among the different agroecosystems of Sikkim, the LCAS was found to be most resilient for bird communities including land use sensitive guilds and conservation concern species, even exceeding the adjacent Forest ecosystems. However, MOAS was the most resilient for the butterfly communities, including the land use sensitive guilds and conservation concern species. The study identified 36 indicator species of birds (and 61 ecosystem exclusive species) dominated by land use sensitive guilds, and conservation concern species. Similarly, 15 indicator species of butterflies (and 81 ecosystem exclusive species) were also identified dominated by land use sensitive guilds, and WPA 1972 protected species. The study recommends the use of these two most well studied vertebrate and invertebrate taxa in monitoring the natural forests and agroecosystems in the region.

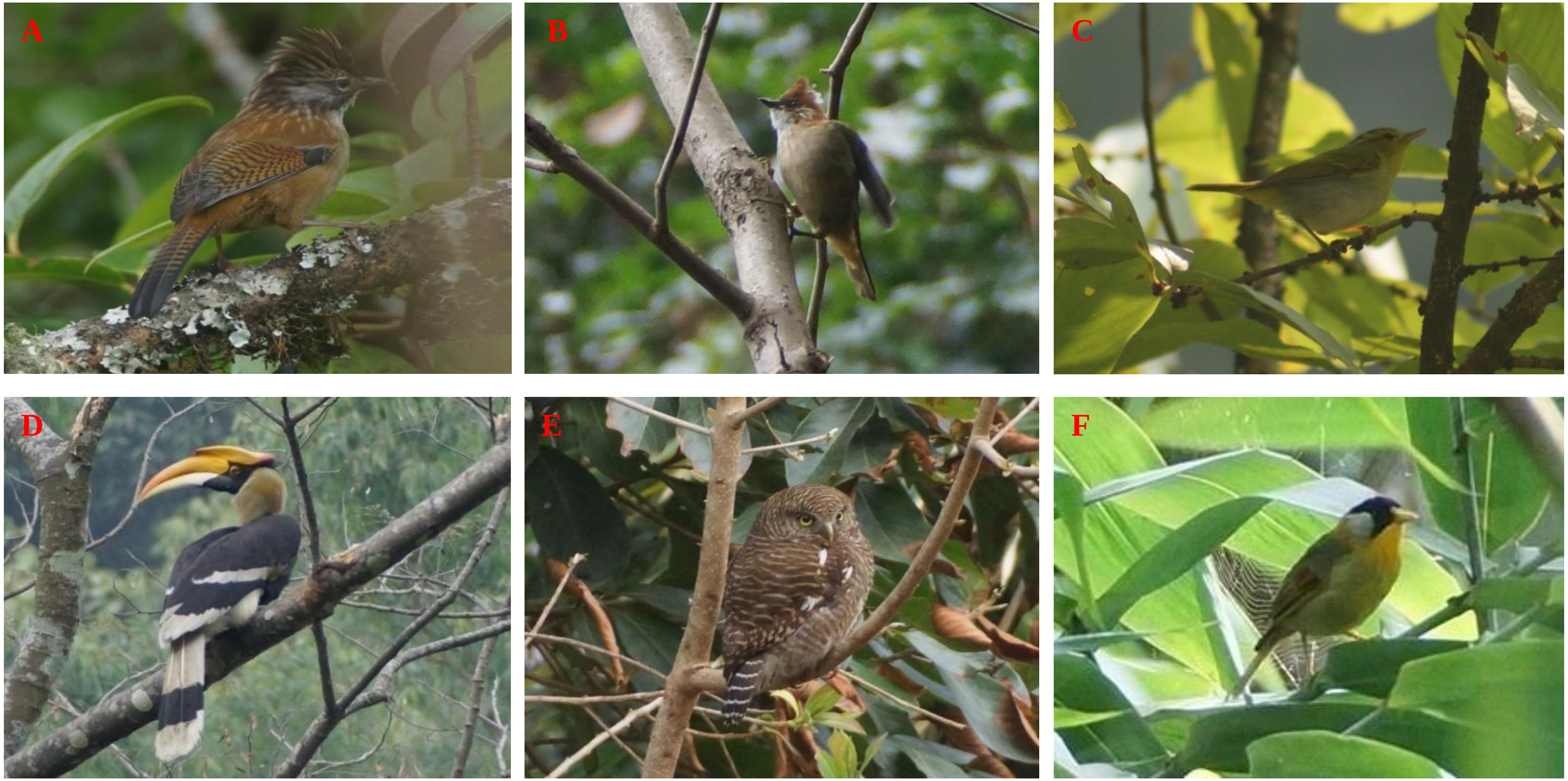


Photo plate 7.1. Representative protected and Eastern Himalayan endemic bird species observed in IFS and adjoining Forest ecosystems of Sikkim, Eastern Himalaya, India. Eastern Himalayan Endemic/ range-restricted: Hoary-throated Barwing *Actinodura nipalensis* (A), White-naped *Yuhina bakeri* (B), Yellow-vented Warbler *Phylloscopus cantator* (C)*;* IUCN Red List (Near Threatened), CITES Appendix I & WPA 1972 Schedule I: Great Hornbill *Buceros bicornis* (D); CITES Appendix II: Asian-Barred Owlet *Glaucidium cuculoides* (E); Silver-eared Mesia *Leiothrix argentauris* (F).

Photo plate 7.2. Representative protected butterfly species observed in IFS and adjoining Forest ecosystems of Sikkim, Eastern Himalaya, India. WPA Schedule I: Scarce Red Forester *Lethe distans* Butler, 1870 (A); Sullied Bhutan Sergeant *Athyma jina jina* Moore, 1857 (B); WPA Schedule II: Common Gem *Poritia hewitsoni hewitsoni* Moore 1865 (C); Striped Punch *Dodona adonira adonira* Hewitson, 1865 (D); Orange Punch *Dodona egeon egeon* Doubleday, 1851 (E); WPA Schedule IV: Striped Blue Crow *Euploea mulciber mulciber* Cramer, 1777 (F).

Chapter 8

SYNTHESIS AND RECOMMENDATIONS

Chestnut-headed Bee-eater *Merops leschenaulti*

8.1. Overview

Conservation of biodiversity in agroecosystems is a global challenge as conversion of forest to agroecosystems has been one of the major causes for biodiversity loss through habitat degradation and landscape simplification. To deal with global challenges of biodiversity conservation and food security, land-sparing/land-sharing conservation frameworks has been proposed and debated. Most studies have reported importance of land-sparing (requiring PAs separated from intensive agriculture), which has been largely successful and preferred strategy worldwide. However, increasing number of studies in recent decades have also supported effectiveness of land-sharing (forests adjacent to wildlife friendly agriculture) to complement land-sparing, while few other authors have advocated for mix of both of these conservation strategies (details provided in chapter 1).

Studies from the globe as well as India (albeit mostly from the Western Ghats region) have found that the agroecosystems, especially those traditionally managed or organic retain high biodiversity (birds, butterflies, plants, and other faunal and floral taxa) including endemic, specialists and conservation concern species, and sustain associated ecosystem service provisioning. Although bird and butterfly communities of Sikkim have been well studied in the past, however, these studies mostly covered the PAs or forest ecosystems. The agricultural-dominated landscapes in the low- and mid-hills (<1800 m elevation) of Sikkim in spite of low coverage of PAs harbours highest diversity of birds, butterflies and other faunal/floral taxa, which is experiencing high anthropogenic pressure. These organic and traditionally-managed agroecosystems of Sikkim, Eastern Himalaya has not been studied for their potential in biodiversity conservation, including the conservation concern species. Birds and butterflies are widely used as indicator taxa in biodiversity assessment and ecological monitoring studies. Therefore present study was conducted with three main objectives: 1) To assess the community structure of birds and butterflies in the indigenous farming systems; 2) To understand the functional diversity of birds and butterflies in the indigenous farming systems; and 3) To identify indigenous farming systems of high biodiversity conservation value (detailed in Chapters 1-2).

Sikkim is an important constituent of the global biodiversity hotspot of Himalaya, and possesses 43.45% of birds (580 out of 1335 species) and 51.96 % of butterflies (690 out of 1328 species) found in India, despite covering only 0.22% of the country’s TGA. It falls within the Eastern Himalaya, which is an EBA, a global ecoregion, and an agro-biodiversity hotspot. The field sampling for birds and butterflies were conducted in East and South districts of Sikkim during December 2012 to August 2017, covering three representative IFS (MOAS, FAS and LCAS) along with adjacent Forest ecosystem, which represents agroecosystem-forest gradients in the study area. I established 24 transects (6 transects per study ecosystems) and marked 240 permanent sampling points (60 per study ecosystems) within the elevation gradient of 600-2000 m AMSL (details in Chapter 3). I sampled birds following open-width point count method and completed 2050 point counts across 24 transect (details provided in chapter 4). Based on past studies, important habitat and environmental variables were quantified for each of the 24 transects (detailed in Chapter 4). The butterflies were sampled following fixed-width point count method and 1760 point counts were completed during the study period (details in chapter 5). The functional traits were collected for each of the bird and butterfly species observed during the study (please see chapter 6). From the observed bird and butterfly communities, I identified conservation concern species (and compared with the total for Sikkim and India), as well as indicator and ecosystem exclusive taxa for ecological monitoring in the Eastern Himalaya (details in Chapter 7). I have provided the brief results and recommendations for conservation and management in the following sections.

8.2. Bird alpha and beta diversity

In chapter 4, I report the patterns and determinants of taxonomic alpha and beta diversity of birds in the IFS (LCAS, MOAS, and FAS) along with adjacent natural Forests ecosystem of Sikkim, Eastern Himalaya, India. I recorded 221 total bird (TBi) species including 164 insectivores, 125 forest specialist, 38 forest specialist canopy insectivores (FSCIBi), 35 forest specialist understorey insectivores (FSUIBi), 20 forest specialist midstorey insectivores (FSMSIBi), 18 forest specialist frugivores and nectarivores (FSFNBi), and 18 protected and endemic (PEBi) species. Alpha diversity (also abundance per point and Shannon-wiener diversity) of TBi, and four out of five land use sensitive guilds (FSUIBi, FSCIBi, FSFNBi, PEBi) were significantly highest in LCAS, whereas, FSMSIBi in Forest. For the different avian-mediated ES, significantly highest alpha diversity (also abundance per point) was observed for invertebrate pest control, and pollination in LCAS and Forest, showing increasing pattern along the agroecosystem-forest gradient, whereas, seed dispersal, waste disposal, nutrient deposition, and ecosystem engineering (highest in MOAS, FAS) declined along the gradient. There was also marked seasonal dynamics (breeding vs. dispersal and wintering) in the bird communities, which in turn differed among the TBi, different land use sensitive and ES guilds. Pair-wise beta diversity was significantly highest in Forests (followed by FAS) that declined in MOAS for total birds and three land use sensitive guilds (FSUIBi FSMSIBi, and PEBi), four ES providers (Invertebrate pest control, Pollination, Seed dispersal, and Vertebrate pest control & Scavenging). However, FAS (followed by Forest) had significantly highest pair-wise beta diversity for the other two land use sensitive (FSCIBi, and FSFNBi) and ES (Nutrient deposition, and Ecosystem engineering) sub-groups.

Both alpha and beta diversity were determined by tree species richness, tree density, percentage canopy cover, elevation, MAP and MAT, however, the strength and direction differed between TBi, and different land use sensitive guilds, with significant relationships being few in the former but more clear in the latter. I report high bird conservation potential of these IFS (especially in LCAS) due to their greater resilience for birds including forest specialist and conservation concern species, and high multiple-site beta diversity dominated by substitution components.

8.3. Butterfly alpha and beta diversity

In addition to birds, I assessed land use effect on butterfly diversity along agroecosystem-forest gradient of Sikkim in the Eastern Himalaya. During the study, I recorded 268 species of butterflies representing six families which included two-third forest specialist, one-third monophagous and one-fifth conservation concern species. Along the agroecosystem-forest gradient, alpha diversity of butterflies declined for total, forest specialist, monophagous and protected species. However, pair-wise beta diversity increased and the multiple beta diversity was dominated by substitution components for both total butterflies and land use sensitive sub-groups. The butterfly community assemblage were significantly determined by ecosystems for total species, forest specialist and monophagous (but not for protected species), with transects from the two IFS clustering together distinct from LCAS and Forest.

The butterfly communities showed marked seasonal dynamics, which in turn were most speciose and abundant in monsoon and post monsoon season across the four ecosystems. Alpha diversity and abundance per point were determined by tree species richness, tree density, percentage canopy cover, elevation, MAP, season, whereas tree species richness, tree density, tree basal area, percentage canopy cover, elevation, mean annual temperature, and MAP influenced beta diversity. The IFS and adjoining Forest ecosystems forming the mosaic landscape in the Sikkim Himalaya sustained high taxonomic alpha and beta diversity as well as community composition of butterflies including specialist (forest specialist, monophagous) and protected species.

8.4. Functional diversity of bird and butterfly communities

Understanding the consequences of the land-use and climate change on biodiversity and ecosystem functions can be better achieved by focusing on functional diversity (FD) and trait composition instead of only taxonomic diversity (TD). The functional diversity can most appropriately represent the biodiversity-ecosystem - function relationships, and are better suited to understand community assembly mechanism and consequences of biodiversity loss due to their greater sensitivity to land-use change and intensification. The chapter 6 of this thesis provides an understanding on the pattern and plausible determinants of FD and functional composition of birds and butterflies along agroecosystem-forest gradient of Sikkim, Eastern Himalaya.

For birds, the declining pattern of functional richness (FRic) along agroecosystem-forest gradient did not mirror the pattern of TD (species richness, Shannon-Wiener diversity) or FDis (functional dispersion). On the contrary, FD (FRic, FDis) and TD (species richness, Shannon-Wiener diversity) showed similar declining pattern along agroecosystem-forest gradient for butterflies. For both the taxa, functional evenness (FEve) and functional divergence (FDiv) were comparatively higher (though non-significantly) in the LCAS and Forests than MOAS and FAS. There was a positive relationship between FRic and species richness for both observed and simulated communities (though comparatively more strongly in case of butterflies), which in turn resulted in positive biodiversity-ecosystem function relationships.

The FD indices showed strong and significant relationships with the different habitat and environmental variables for both the taxa. In particular, relationships of FRic and FDis (showing significant difference among the four ecosystems) with the different habitat and environmental variables differed for birds and butterflies. For birds, FRic had weak (non-significant) positive correlation with tree species richness, MAT and MAP, but negative correlation with elevation, pcc, tree density, tree basal area, whereas, butterflies’ FRic correlated significant positively with MAT and MAP, and significant negatively with elevation, tree density and basal area. For FDis of birds, different variables exerted strong significant effect (positive by tree species richness and MAT, but negative by elevation, pcc and tree basal area), whereas, for butterflies FDis showed weak negative correlation with tree species richness, density and basal area. However, both CWM_bodymass (of birds), and CWM_Wingspan (of butterflies) correlated significant positively with MAT and MAP but negatively with elevation, and additionally tree density exerted significant negative effect on CWM_Wingspan. The FDiv of birds correlated weak positively only with tree species richness, but negatively with elevation and pcc, whereas, for butterflies it correlated significantly negatively with MAT, but positively (non-significant) with elevation, tree basal area, tree density, and pcc. The negative effect of tree species richness on FEve, was strong significant for birds but weak/ non-significant for butterflies. The

FEve also showed positive (though non-significant) effect of percentage canopy cover (for both taxa) and tree basal area (for only birds).

This study found that community assembly for birds and butterflies in the heterogeneous mosaic landscapes dominated by the IFS (outside PA networks) of Sikkim were structured by both limiting similarity, and environmental filtering based on niche theory. The limiting similarity played more pivotal role in driving bird and butterfly communities in the relatively more simplified agroecosystems of MOAS and FAS, whereas environmental filtering was more important in structuring bird and butterfly communities in the complex ecosystems of LCAS and Forests.

The functional composition for birds and butterflies showed significant effect of landscape environmental variables and species' functional traits. For the bird and butterfly functional composition, species traits showed significant associations with the environmental variables, and there was filtering of species traits along the agroecosystem-forest gradient. The traits specific to generalists species were preferably linked with simplified agroecosystems of MOAS and FAS, whereas traits specific to specialists or land-use change vulnerable functional groups were preferably associated with the complex ecosystems of LCAS and Forests for both the studied taxa. The functional composition for birds will be best maintained in the Forests and LCAS, whereas low strength and fewer significant relationship between species traits and environmental variables in case of butterflies implies the importance of MOAS and FAS also in maintaining functional composition of butterflies. The present study shows that the traditional and organically managed agroecosystems of Sikkim, Eastern Himalaya can sustain high FD and functional composition of birds and butterflies and associated ES.

8.5. Protected, endemic and indicator species of birds and butterflies

The traditionally managed and organic agroecosystems and adjoining forests of Sikkim, Eastern Himalaya possessed 38.1 % of bird and 38.8% of butterfly species of the state. In the present study, bird communities were most resilient in LCAS and Forests, whereas butterflies in MOAS and FAS across spatio-temporal scales. These agricultural-dominated landscapes supported high diversity of birds and butterflies, dominated by land use sensitive guilds including protected and endemic species. About 8.14% of birds (18 species) and 19.78% of butterflies (53 species) observed in the present study are conservation concern species.

The three IFS types supported comparable species richness and diversity of protected and endemic/range-restricted bird species than Forest. The conservation concern bird species sustained in the IFS and adjoining Forest ecosystems was represented by 10 WPA Schedule I protected bird (out of 102 reported for India), such as *Pavo cristatus, Lophura leucomelanos, Accipiter* spp., and *Spilornis cheela;* four IUCN Red Listed species (out of 37 that occur in Sikkim) with two species each categorized as Near Threatened (*Gyps himalayensis* and *Haliaetus humilis*), and Vulnerable species (*Buceros bicornis* and *Sitta formosa*). They also included 11 CITES protected species representing one CITES Appendix I species i.e., *Buceros bicornis*, and 10 CITES Appendix II species such as *Glaucidium* spp., *Leiothrix argentauris* and *Leiothrix lutea*, as well as three Eastern Himalaya endemic/range-restricted species (out of the 10 found in Sikkim) viz., *Actinodura nipalensis*, *Yuhina bakeri*, and *Phylloscopus cantator*.

However, the species richness and diversity of protected butterfly species in the IFS types was higher (in MOAS) or comparable (in FAS) with Forest, similar to the pattern for total butterflies. About one-fifth of the total butterflies observed in the present study was conservation concern species protected, either at national level i.e., under WPA 1972 (9 Schedule I, 33 Schedule II and 9 Schedule IV) and India Red Data Book of butterflies (9 species), or at international level i.e., under CITES Appendix II (*Troides aeacus aeacus*, *Troides helena cerberus*).

Therefore, the IFS along with adjacent Forests allowed the total community and conservation concern species of birds (discussed at detail in Chapter 4 and 7), and butterflies (also see Chapter 5 and 7 for details), to thrive at different spatial scale (local and landscape) and across seasons. The persistence of birds and butterfly communities, including conservation concern species in the different agroecosystems, were significantly higher (in LCAS for birds; in MOAS for butterflies) or comparable (birds in FAS, MOAS; butterflies in FAS, LCAS) than the adjoining Forests. The multiple-site beta diversity for both taxa (discussed in Chapter 4 & 5) were also highly dominated by substitution components, further signifying the importance of all the ecosystems (all IFS types and Forest ecosystems) in biodiversity conservation framework in Sikkim. Hence, the different IFS types and Forest needs to be conserved and properly managed to protect the total communities as well as conservation concern species of birds and butterflies in the region.

I identified 36 indicator bird species specific to a single ecosystem (MOAS: two species; FAS: one species; LCAS: six species), or group of two ecosystems (IFS types: eight species; and Forests and LCAS: six species; Forest and MOAS: one species) and three ecosystems (three types of IFS: five species; Forest and two IFS: seven species). The highest indicator species were found for LCAS (28 species), which was about twice than other IFS (14-16 species) and Forests (15 species). The

17 forest specialist (FSFNBi: two species; FSCIBi: 8 species; and FSUIBi: 7 species) indicator species were represented well in LCAS (16 species), and Forest (12 species), but poorly in FAS and MOAS (2-7 species). The two CITES Appendix II protected indicator species namely *Leoithrix argentauris* and *Leoithrix lutea* were specific to LCAS and Forest (see for details in Chapter 7). Similarly, I also identified 15 indicator butterfly species for specific (MOAS: six species; LCAS: three species) or group of two (1 species: MOAS and FAS) or three (4species for three IFS types, 1 species for Forest, LCAS and FAS) ecosystems. Although only one species (*Aglais caschmirensis aesis*) was identified as indicator for Forest, but the total indicator butterflies (across the ecosystems) were dominated by forest specialist (11 species) including three monophagous and two WPA 1972 (Schedule II) protected species (Discussed in Chapter 7). The identified indicator species were dominated by land use sensitive birds (17 forest specialist including two CITES Appendix II protected species), and butterflies (11 forests specialists, including three monophagous, two WPA 1972 Schedule II protected species) and hence are suitable for long-term ecological monitoring program in the Eastern Himalaya.

Among the ecosystem exclusive birds (61 species) and butterflies (81 species), three species each were also identified as indicator species. Three identified indicator species of birds: *Upupa epops* (for FAS), *Machlolophus xanthogenys* and *Phylloscopus reguloides*d (for LCAS), as well as butterflies: *Flos asoka* de Nicéville, 1884 (for MOAS); *Euthalia franciae franciae* Gray, 1846; *Abisara chela chela* de Nicéville, 1886 (for LCAS) were specifically observed in these ecosystems. Furthermore, forest specialists, land use sensitive and conservation concern sub-groups dominated the ecosystem exclusive communities of birds (33 forest specialists; 5 protected species), and butterflies (68 forests specialists; 22 monophagous; 25 WPA

1972 protected: 5 each in Schedule I and IV; and 15 Schedule II protected species). Thus, from both the studied taxa, ecosystem exclusive species can be effectively employed alongside their indicator species counterparts for future monitoring of threatened biodiversity and ecosystems in the Eastern Himalaya.

8.6. Management implications

In the past, research on agroecosystems vis-a-vis biodiversity of India has been predominantly conducted in Western Ghats region and very few in the Himalaya and North East India. Most of these studies were conducted with single taxa or in one type of agroecosystem. Even in the studies reported from different parts of the world, very few were conducted in agroecosystems of montane region with traditional and organic management systems. The relative importance of forests, agroforests, and simplified agricultural systems in landscapes with varying forest cover and composition in biodiversity conservation especially with regards to FD and associated ES like pollination and natural pest control at different spatial and temporal scale are not well understood. The human-modified landscape in the low- and mid-hills consisting of mosaic of forest and cultivated systems with poor representation of PAs are facing the greatest threat from deforestation, land use change and agricultural intensification and have not been considered important in policy formulations. The first step towards conservation of wild biodiversity which primarily depend on or occasionally use agroecosystems in Himalaya will be to generate robust scientific information on them to facilitate the policy makers to take informed decision while formulating policies at the regional and national level. Therefore, the present study with multiple taxa in multi agroecosystems has allowed filling up this research gap in Himalayan region as

- It aided in understanding the diversity and community structure of birds and butterflies in the IFS and adjoining forests across spatial and temporal scale and to generate baseline information.
- The study also aided in understanding the FD and functional composition of birds and butterflies in IFS and adjacent Forests with respect to ES by birds such as natural pest control, pollination, seed dispersal, scavenging, ecosystem engineering, nutrient deposition, and by butterflies, e.g., pollination.
- The relative importance of the different IFS for biodiversity conservation was understood, ecosystems of high conservation value were identified, and indicator species (36 birds and 15 butterflies) were identified dominated by land use sensitive and conservation concern species for long-term ecological monitoring in future.
- The indigenous farming system-dominated agricultural landscapes can complement the PAs in fostering biodiversity conservation and ecosystem service provision, especially in the areas with high human pressure and low PA and forest coverage. Therefore, the present study highlights the important of adopting land-sharing conservation approach, especially in the low- and mid-hills of Sikkim to complement the land-sparing framework. This will provide win-win scenario for achieving food security and human well-being without compromising the efforts for biodiversity conservation and maintenance of ecosystem services in the natural and human-modified ecosystems of Eastern Himalaya.
- The findings of the study will be inputs for management and conservation of IFS and associated biodiversity in the Eastern Himalayan region and aid in holistic policy formulation. Given their rich biodiversity including conservation concern species, these findings can help in further developing the study sites, especially

Lingee-Payong, Sumbuk (South Sikkim), Bering-Tareythang, Pendam (East Sikkim) into ecotourism sites.

- In the present study, as discussed in the preceding chapters, I found that the organic and traditionally managed agroecosystems of Sikkim can conserve high taxonomic diversity including land use sensitive guilds of birds (see chapter 4) and butterflies (see chapter 5) across spatio-temporal scale. These agroecosystems also sustained functional diversity and functional composition for both taxa (see chapter 6). They also supported high diversity of conservation concern species from both taxa and the identified indicator species can be effectively used for monitoring health of these ecosystems in future (see chapter 7). Thus, agroecosystems along with the adjoining forests (that formed mosaic landscape) were found effective for complementing PAs in biodiversity conservation.

- The indigenous farming systems of Sikkim was officially designated as an associate candidate site under globally important agriculture heritage site by FAO, UNO way back in 2007. Now, this study has highlighted its very high conservation potential for biodiversity across vertebrate and invertebrate fauna, and trees. Therefore, a dedicated effort is urgently required for upgrading them into full site under GIAHS. This will help to properly safeguard wild biodiversity and agrobiodiversity of the state, and further enhance the potentiality of its ecotourism sector.

- The agroecosystems of Sikkim with rich biodiversity and supporting critical ecosystem service providers are highly threatened by anthropogenic climate change, habitat loss and land-use change. In the past two decades, the biodiverse tropical and sub-tropical belts have turned into the hub of developmental activities such as hydro-power dams, industrialization, road constructions and urbanization. For example, these important agricultural ecosystems have been rapidly converted to industrial belts such as pharmaceuticals (with 56 built till date and 48 currently operational), particularly in East and South districts of Sikkim, where the present study was conducted. During the same period, Sikkim also witnessed mushrooming of hydroelectric projects throughout the state earning the distinction of having the highest density of dams (4 dams/1000 km^2) in the world (Pandit, 2017), which have greatly damaged much of the low land terrace rice cultivation and adjacent natural forests. In the Himalaya, maximum number of hydro-electric dam sites overlaps with the species richness maxima (i.e., 600-1600 m elevation) across taxonomic groups, especially birds, butterflies and angiosperms (Pandit, 2017). These biodiversity will be further threatened with the ongoing construction of >300 hydro-power dams and >3000 km border roads each by India and China in the Himalaya. Therefore, a policy is needed to safeguard these vulnerable ecosystems and their biodiversity and there should be a ban on further conversion of traditional agricultural lands for non-agricultural purposes or commercial plantations crops such as oil palm being aggressively implemented in the North East India. A delay in doing so will have a devastating consequence for the rich wild biodiversity, agrobiodiversity, ES provisioning as well as livelihood of the local communities, and Sikkim will remain as an organic state only in paper. The planned policy formulation must take into consideration the sustainable development goals, ecosystem restoration, as well as long-term food security and human-health of the local populations, instead of focusing only on economic development.

8.7. Limitation of the study

Like most ecological studies, the present study also encountered some limitations. I could not cover all the agroecosystem types of Sikkim across much wider elevation gradient across the four districts of Sikkim. I also could not conduct field work for longer duration and track few indicator species populations, which would have been better. However, I designed robust study covering 24 transects, 240 permanent points across four major land-use types (three dominant agroecosystems and adjacent Forest ecosystems) for multi-year seasonal sampling of birds and butterflies. Therefore, the present study adequately allowed in understanding the biodiversity conservation potential of the organic and traditionally managed agroecosystems of Sikkim.

8.8. Way forward

The present study has provided baseline for understanding the importance of human modified ecosystems, especially IFS, in retention and conservation of birds and butterflies. Since the present study covered only three IFS along with forest ecosystems in Sikkim alone, the future research should focus on more number of ecosystems over larger landscape covering few other regions in the Eastern Himalaya and northeastern India assessing all facets of biodiversity including phylogenetic diversity. There is also need to track the population dynamics of the identified indicator species of birds and butterflies. Mark-recapture studies for selected endemic, threatened and other specialist bird and butterfly species will help in better understanding their distribution range and seasonal migration along spatio-temporal scales. Response of mixed-flocking vs. non-flocking bird species to land use change in the Eastern Himalaya can be studied. Additionally, research on other taxa such as odonates, beetles, spiders, herpetofauna, fishes and mammals for a longer period are necessary. The human dimensions of conservation and ecosystem services become an important area of research in future. Studies on effect of climate change on birds, butterflies, and other floral and faunal taxa, and niche modelling of selected endemic, threatened and indicator species using suitable geospatial tools and techniques can also be carried out. The government of India is aggressively pushing for oil palm and rubber cultivations in the northeastern states, and Andaman and Nicobar Islands. Sikkim is also likely to follow suit, which would threaten its biodiverse agroecosystems and forest ecosystems. Hence, there is urgent need for more scientific evidences in favour of sustainable and wildlife-friendly agriculture over commercial monocultures/ conventional agriculture. This will allow in greater recognition of land-sharing to complement land-sparing conservation frameworks by the relevant policy makers, local communities, and global scientific communities.

The tropical and subtropical belts in Sikkim has very low coverage of PAs where primary forests were converted into teak plantations or agriculture lands during the past centuries, and it still continue to be under huge pressure from the various developmental activities. However, remnant patches of original forest still exist in few places, which along with agroecosystems, a dominant land use type in the zone, forms the mosaic landscape (forest patches within agroecosystems). The mosaic landscape in the present study harboured high diversity and distinct composition of birds and butterflies (in terms of both taxonomic and functional facets), which might have been partly contributed by the remnant forest patches and the size of these forest patches. The current study was not designed to address the questions on the role played by size of nearby forest patch, spatial and temporal proximity between natural forest patches/ protected areas and agro-ecosystems in retaining biodiversity (i.e., bird and butterfly communities). The area of the forest patches (used as control) in the present study ranged between ~100 ha to more than 200 ha (some of which were continuous to larger reserve forests or PAs of >1000 ha). Forests with larger patch size did not necessarily resulted into higher diversity of birds and butterflies, as high diversity of birds and butterflies (>100 species each) were observed even in small forest patches, for instance, in Tareythang forest and its adjoining agroecosystems.

However, preliminary analysis revealed that the study sites across the three IFS, situated relatively closer to reserve forests or the protected areas had comparatively higher richness and diversity (particularly for birds) at the level of both fields and land-use types. The biodiversity was higher in the study sites adjacent to the reserve forests or PAs, for instance in LCAS (Sumik-Khamdong to Tumin forest and Fambunglho WLS; Pantharey-Simkharka to Payong forest and Maenam WLS; Luing to Fambunglho WLS), and in FAS (Bering to Tareythang forest; upper Payong to Kaw forest; Sumbuk and Sumbuk-Kamarey to Sumbuk forest and Kitam bird sanctuary). The study sites in the LCAS and FAS were mostly continuous to the reserve forests, or in relatively more proximity to protected areas.

Past studies has reported significant and positive role of forest patches in driving retention of biodiversity and ecosystem services in the agricultural landscapes (Mitchell *et al.*, 2014; Kalda *et al.*, 2015). Presence of nearby forest patches, large patch size, and connectivity between habitat patches in the landscape enhances habitat quality, which in turn has positively effects on taxonomic (e.g., species richness) and functional (e.g., FRic) diversity for total birds (Bovo *et al.*, 2018; Ehlers Smith *et al.*, 2018; Lee and Carrol, 2018), and Frugivorous species (Bovo *et al.*, 2017). Reduction in patch size leads to decline in CWM_bodymass in cavity-nesting (Ibarra *et al.*, 2017) and frugivorous birds (Bovo *et al.*, 2017). Similarly, proximity to natural Forest and PAs also enhances biodiversity retention in neighbouring agroecosystems, for instance, species richness and taxonomic diversity in birds (Anand *et al.* 2008; Karanth *et al.* 2016) and butterflies (Dolia at al. 2008). The proximity to forest and proportion of ‘wildlife-friendly’ habitat also enhances FRic of birds in neighbouring agroecosystems (Cannon *et al.*, 2019). However, these effects can differ across season (e.g., breeding vs. winter or dispersal) in birds (Yabuhara *et al.*, 2019). Similarly, ecosystem functioning, e.g., avian-mediated pest control services depends on remnant habitat size and proximity to habitat patch (Karp *et al.*, 2013; Jordani *et al.*, 2015).

However, a recent study by Bełcik *et al.* (2020) has also reported differential response of biodiversity matrices (taxonomic, functional and phylogenetic diversity) for bird communities to habitat fragmentation metrics (forest patch size, isolation) and forest stand characteristics (forest stand age, stand density). Bełcik *et al.* (2020) found increase in birds’ taxonomic diversity (most sensitive to fragmentation) with increasing forest patch size, which stabilized after reaching a thresholds level of fragmentation (~200 ha) for both total species and forest specialists. There was rapid decline in phylogenetic diversity with increasing size of forest patch, but it stabilized at low level of diversity and large patch area. There was linear decline in functional diversity (least sensitive to fragmentation) with decrease in isolation of forest patches. There was positive effects of forest stand age on total bird species’ taxonomic and functional diversity (but negative for phylogenetic diversity) and for forest specialists’ taxonomic diversity. Finally the study found positive effect of forest stand density on taxonomic and functional diversity (but negative for phylogenetic diversity) of total species and forest specialists.

In future, researchers can look into the role of patch size of the nearby forests, and spatial and temporal proximity between forests and agro-ecosystems, as well as stand characteristics (e.g., forest stage age and stand density) in determining the extent of biodiversity retention and ecosystem services at different scales. Studies on the pattern and drivers of taxonomic, functional and phylogenetic diversity for total species, land use sensitive guilds (e.g., forest specialists), and ecosystem function providers (for birds, butterflies, plants, and other taxa) along the gradient of forest patch size, proximity to PAs/ natural forest patches, forest stand age may be undertaken. The study may be designed to include pairs of agroecosystems and natural forest patches, with varying patch sizes (e.g., 5 ha, 10 ha, 20 ha, …, 200 ha, etc. or based on exact size in ha), proximity (e.g., 200m, 400m, 600m, 1km, …, 50 km, etc.) of agroecosystems to natural forest patches and PAs. This will further improve our understanding on the role of adjoining forest patches, critical forest patch size, and proximity to natural forests or PAs, for retention biodiversity and ecosystem functioning across spatial and temporal scales in agricultural landscapes of the Eastern Himalaya.

Appendix A. The details on coordinates, habitat and environmental variables of transects laid in different IFS (MOAS, FAS, LCAS) and adjoining Forest of Sikkim Himalaya, India. Mandarin orange-based agroforestry system (MOAS: T1-6), farm-based agroforestry system (FAS: T7-12), large cardamom-based agroforestry system (LCAS: T13-18), Natural forest (Forest: T19-24).

Transects	Latitude (°N)	Longitude (° E)	Elevation (elev: m)	Percentage canopy cover (pcc: %)	Mean annual temperature (MAT/temp: °C)	Mean annual precipitation (MAP/rain: mm)	Tree species richness (tsr)	Tree density (tden.: stems ha^{-1})	Tree basal area (tba: m^{2} ha^{-1})
T1	27.373	88.483	750	51.2	20.93	3683	73	767.5	34.61
T2	27.373	88.475	850	43.03	19.67	3048	90	862.5	29.75
T3	27.202	88.523	1000	59.19	20.95	3110	67	1057.5	66.36
T4	27.247	88.5	1100	46.9	20.42	3159	86	1072.5	26.49
T5	27.265	88.496	1250	48.83	18.98	2788	72	1027.5	28.09
T6	27.217	88.525	1540	69.48	18.81	2822	57	902.5	41.96
T7	27.21	88.659	700	72.16	20.78	4184	74	640	50.16
T8	27.213	88.654	800	65.8	20.93	4160	53	680	79.98
T9	27.12	88.389	980	50.23	21.63	2976	72	1055	34.35
T10	27.114	88.378	1030	53.23	22.1	3117	71	1000	39.38
T11	27.369	88.458	1450	60.3	18.52	2539.5	68	1257.5	48.49
T12	27.368	88.452	1550	55.08	17.32	2477	54	1270	31.33
T13	27.376	88.585	1200	66.13	17.25	2505	71	1442.5	33
T14	27.354	88.594	1280	63.53	17.94	2661	68	1122.5	36.46
T15	27.289	88.466	1500	80.65	17.92	2644	51	1052.5	45.41
T16	27.389	88.443	1610	65.19	18.45	2642	49	775	52.08
T17	27.371	88.44	1750	65.78	15.57	2315	47	847.5	55.96
T18	27.297	88.474	2000	72.6	16.82	2604	53	772.5	55.34
T19	27.107	88.367	640	77.43	23.76	3708	30	1142.5	66.61
T20	27.22	88.644	850	81.3	20.6	3919	56	812.5	68.75
T21	27.364	88.451	1597	75.17	17.95	2523	65	1302.5	45.44
T22	27.295	88.478	1800	84.31	15.73	2524	39	845	91.92
T23	27.224	88.549	1850	79.15	16.56	2717	68	1470	66.7
T24	27.371	88.434	2000	62.53	15.57	2120	72	1950	76.25

Appendix B. Bird species observed in different IFS (MOAS, FAS, LCAS) and adjoining Forest of Sikkim Himalaya, India.Bird taxonomy, classification and auto-ecological traits follows: Ali (1962; 1977); Ali and Ripley (2002); Şekercioğlu (2006); Grimmett *et al.* (2011, 2019); Praveen *et al.* (2016, 2020a, 2020b).

^{I}**Sl. no.**	^{II}**6-letter code**	^{III}**Scientific name**	^{IV}**Common name**	^{V}**Ecosystems**				^{VI}**PFG**	^{VII}**Migr.**	^{VIII}**HS**	^{IX}**PFS**	^{X}**Av.ES**	^{XI}**IndVal**
				MOAS	**FAS**	**LCAS**	**Forest**						
		ACCIPITRIDAE											
1$^{\#\#,+}$	ACCBAD	*Accipiter badius*	Shikra	4	5		2	Carni	R	OA	C	WD	
2$^{\#\#,+}$	ACCNIS	*Accipiter nisus*[4]	Eurasian Sparrowhawk				1	Carni	AlM	FG	NA	WD	
3$^{\#\#,+}$	ACCTRI	*Accipiter trivirgatus*	Crested Goshawk		2		1	Carni	R	FG	C	WD	
4$^{\#\#,+}$	ACCVIR	*Accipiter virgatus*	Besra	5	5	4	7	Carni	AlM	FG	U	WD	
5$^{NT,\#\#,+}$	GYPHIM	*Gyps himalayensis*[1]	Himalayan Vulture	6				Carni	AlM	FG	C	WD	
6$^{NT,\#\#,+}$	HALHUM	*Haliaetus humilis*[1]	Lesser Fish Eagle	2				Carni	R	FS	Aq	WD; ND	
7$^{\#\#,+}$	SPICHE	*Spilornis cheela*	Crested Serpent Eagle	24	13	21	13	Carni	AlM	FG	C	WD	
		AEGITHALIDAE											
8	AEGCON	*Aegithalos concinnus*	Black-throated Tit	4	6	50	62	Insect	R	FG	C	IPC	
9	AEGIOU	*Aegithalos iouschistos*[2]	Black-browed Tit		6			Insect	AlM	FS	C	IPC	
		AEGITHNIDAE											
10	AEGTIP	*Aegithina tiphia*	Common Iora	2	22		7	Insect	R	FG	C	IPC	
		ALCEDINIDAE											
11	HALSMY	*Halcyon smyrnensis*	White-throated Kingfisher	3	3			Carni	R	FG	C	WD; EE; ND	
		ARDEIDAE											
12	ARDBAC	*Ardeola bacchus*	Chinese Pond Heron	2	4			Carni	WV	OA	Aq	WD; ND	
		BUCEROTIDAE											
13$^{VU,\#,+}$	BUCBIC	*Buceros bicornis*[2]	Great Hornbill		1			Frug	R	FS	C	SD; EE	
		CAMPEPHAGIDAE											
14	CORMAC	*Coracina macei*	Large Cuckooshrike	6	1	2	2	Insect	AlM	FG	C	IPC	

15	LALMEL	*Lalage melaschistos*	Black-winged Cuckooshrike			2	1	Insect	AlM	FG	C	IPC	
16	PERBRE	*Pericrocotus brevirostris*	Short-billed Minivet	1		2		Insect	AlM	FS	C	IPC	
17	PERETH	*Pericrocotus ethologus*	Long-tailed Minivet	14	36	35	11	Insect	AlM	FG	C	IPC	
18	PERSPE	*Pericrocotus speciosus*	Scarlet Minivet	90	125	93	136	Insect	R	FS	C	IPC	
		CERTHIIDAE											
19	CERDIS	*Certhia discolor*	Sikkim Treecreeper	3	2	10	3	Insect	AlM	FS	MS	IPC	
20	CERNIP	*Certhia nipalensis*[3]	Rusty-flanked Treecreeper			5		Insect	AlM	FS	MS	IPC	
		CETTIIDAE											
21	ABRSUP	*Abroscopus superciliaris*	Yellow-bellied Warbler	12	4	2	4	Insect	AlM	FS	U	IPC	
22	CETCAS	*Cettia castaneocoronata*	Chestnut-headed Tesia	1	2	2	2	Insect	AlM	FS	U	IPC	
23	TESCYA	*Tesia cyaniventer*[1]	Grey-bellied Tesia	1				Insect	AlM	FS	U	IPC	
24	TESOLI	*Tesia olivea*[3]	Slaty-bellied Tesia			1		Insect	AlM	FS	U	IPC	
		CHLOROPSEIDAE											
25	CHLAUR	*Chloropsis aurifrons*	Golden-fronted Leafbird	3	3	4	10	Nectar-Frug	R	FS	C	P	
26	CHLHAR	*Chloropsis hardwickii*	Orange-bellied Leafbird	2	2	2		Nectar-Frug	R	FS	C	P	
		CINCLIDAE											
27	CINPAL	*Cinclus pallasii*	Brown Dipper	6	3		1	Insect	AlM	FG	U	IPC; ND	
		CISTICOLIDAE											
28	ORTSUT	*Orthotomus sutorius*	**Common Tailorbird**[123]	**118**	**90**	**43**	13	Insect	R	OA	NA	IPC	0.975**
29	PRIATR	*Prinia atrogularis*	Black-throated Prinia	6	17	37	11	Insect	AlM	OA	NA	IPC	
30	PRICRI	*Prinia crinigera*[2]	Striated Prinia		6			Insect	AlM	OA	NA	IPC	
31	PRIFLA	*Prinia flaviventris*[4]	Yellow-bellied Prinia				2	Insect	R	FG	NA	IPC	
32	PRIHOD	*Prinia hodgsonii*[2]	Grey-breasted Prinia		4			Insect	AlM	OA	NA	IPC	
33	PRIINO	*Prinia inornata*	Plain Prinia	8	2	13	6	Insect	AlM	OA	NA	IPC	

34	PRIRUF	*Prinia rufescens*^{2}	Rufescent Prinia		2			Insect	R	FG	NA	IPC	
		COLUMBIDAE											
35	MACUNC	*Macropygia unchall*	Barred Cuckoo Dove	2	20	12	14	Frug	AlM	FS	C	SD	
36	STRCHI	*Streptopelia chinensis*	**Spotted Dove**^{123}	**100**	**80**	**60**	11	Gran	AlM	OA	G	SD	0.922**
37	STRORI	*Streptopelia orientalis*	Oriental Turtle Dove	47	89	33	20	Gran	AlM	OA	G	SD	
38	TREAPI	*Treron apicauda*^{2}	Pin-tailed Green Pigeon		2			Frug	AlM	FS	C	SD	
39	TRESPH	*Treron sphenurus*	**Wedge-tailed Green Pigeon**^{234}		**22**	**35**	**41**	Frug	AlM	FS	C	SD	0.85*
		CORACIIDAE											
40	CORBEN	*Coracias benghalensis*^{4}	Indian Roller				2	Omni	AlM	FG	C	WD; EE	
		CORVIDAE											
41	CISCHI	*Cissa chinensis*	Common Green Magpie	69	62	32	22	Omni	R	FS	C	WD	
42	CORMAC	*Corvus macrorhynchos*	Large-billed Crow	15	39	2	11	Omni	R	FG	C	WD; ND	
43	CORSPL	*Corvus splendens*	House Crow	9	9	42	2	Omni	R	OA	C	WD; ND	
44	DENFOR	*Dendrocitta formosae*	Grey Treepie	391	226	275	136	Omni	R	FG	C	WD	
45	UROERY	*Urocissa erythroryncha*	Red-billed Blue Magpie		2		2	Omni	AlM	FG	C	WD	
46	UROFLA	*Urocissa flavirostris*	Yellow-billed Blue Magpie			4	4	Omni	AlM	FS	C	WD	
		CUCULIDAE											
47	CACPAS	*Cacomantis passerinus*^{2}	Grey-bellied Cuckoo		3			Insect	SV	FG	C	IPC	
48	CLACOR	*Clamator coromandus*	**Chestnut-winged Cuckoo**^{1}	**9**	2		1	Insect	SV	FS	C	IPC	0.791*
49	CUCCAN	*Cuculus canorus*	Common Cuckoo	4	3	1	8	Insect	SV	FS	C	IPC	
50	CUCPOL	*Cuculus poliocephalus*	Lesser Cuckoo	2			3	Insect	SV	FS	C	IPC	
51	CUCSAT	*Cuculus saturatus*	Himalayan Cuckoo		1	1	1	Insect	SV	FS	C	IPC	
52	HIENIS	*Hierococcyx nisicolor*^{3}	Whistling Hawk Cuckoo			2		Insect	SV	FS	U	IPC	

53	HIESPA	*Hierococcyx sparverioides*[4]	Large Hawk Cuckoo				1	Insect	SV	FS	C	IPC	
54	PHATRI	*Phaenicophaeus tristis*	Green-billed Malkoha	5	26	4	10	Insect	R	FS	MS	IPC	
55	SURLUG	*Surniculus lugubris*[2]	Drongo Cuckoo		2			Insect	SV	FG	C	IPC	
		DICAEIDAE											
56	DICIGN	*Dicaeum ignipectus*	**Fire-breasted Flowerpecker**[34]		7	**56**	**32**	Nectar-Frug	AlM	FG	C	P; SD	0.834*
		DICRURIDAE											
57	DICAEN	*Dicrurus aeneus*	Bronzed Drongo		5	4	2	Insect	R	FS	C	IPC	
58	DICHOT	*Dicrurus hottentottus*	Hair-crested Drongo	61	79	37	84	Insect-Nectar	R	FS	C	IPC; P	
59	DICLEU	*Dicrurus leucophaeus*	Ashy Drongo	84	82	91	60	Insect	AlM	FS	C	IPC	
60	DICMAC	*Dicrurus macrocercus*	Black Drongo	5			12	Insect	R	OA	C	IPC	
		ESTRILDIDAE											
61	LONPUN	*Lonchura punctulata*[3]	Scaly-breasted Munia			11		Gran	R	OA	NA	SD	
62	LONSTR	*Lonchura striata*	**White-rumped Munia**[123]	**178**	**166**	**41**	5	Gran	R	OA	NA	SD	0.876*
		EURYLAIMIDAE											
63	PSADAL	*Psarisomus dalhousiae*	Long-tailed Broadbill	11	6	5	16	Insect	R	FS	MS	IPC	
		FRINGILLIDAE											
64	CARERY	*Carpodacus erythrinus*	Common Rosefinch	4	16	12		Gran	AlM	FG	U	SD	
65	CARSIP	*Carpodacus sipahi*[4]	Scarlet Finch				28	Gran	AlM	FS	Gr	SD	
66	CARSUB	*Carpodacus subhimachalus*[3]	Crimson-browed Finch			2		Gran	AlM	FS	Gr	SD	
67	CHLSPI	*Chloris spinoides*[2]	Yellow-breasted Greenfinch		6			Gran	AlM	OA	C	SD	
68	PRONIP	*Procarduelis nipalensis*	Dark-breasted Rosefinch			2	12	Gran	AlM	FS	Gr	SD	
69	PYRERY	*Pyrrhula erythrocephala*[4]	Red-headed Bullfinch				2	Gran	AlM	FS	U	SD	

		HIRUNDINIDAE											
70	CECDAU	*Cecropis daurica*[2]	Red-rumped Swallow		1			Insect	AlM	OA	Ar	IPC; ND	
71	DELNIP	*Delichon nipalense*	**Nepal House Martin[1]**	**299**	10	22	19	Insect	AlM	FG	Ar	IPC; ND	0.924**
72	DELURB	*Delichon urbicum*	Northern House Martin		20	8	17	Insect	PM	FG	Ar	IPC; ND	
		LANIIDAE											
73	LANCRI	*Lanius cristatus*	Brown Shrike	3	2			Insect	WV	FG	NA	IPC	
74	LANSCH	*Lanius schach*	**Long-tailed Shrike[123]**	**41**	**80**	**22**	13	Insect	AlM	OA	NA	IPC	0.930**
75	LANTEP	*Lanius tephronotus*	**Grey-backed Shrike[23]**		**6**	**5**		Insect	SV	OA	NA	IPC	0.707*
		LEIOTHRICHIDAE											
76	ACTCYA	*Actinodura cyanouroptera*	Blue-winged Minla	100	91	162	111	Insect	AlM	FG	U	IPC	
77	ACTEGE	*Actinodura egertoni*[3]	Rusty-fronted Barwing			2		Insect	AlM	FS	U	IPC	
78[$]	ACTNIP	*Actinodura nipalensis*	Hoary-throated Barwing	2	2	25	28	Insect	R	FS	U	IPC	
79	ACTSTR	*Actinodura strigula*	**Bar-throated Siva[34]**			**33**	**26**	Insect	R	FS	U	IPC	0.707*
80	ALCNIP	*Alcippe nipalensis*	**Nepal Fulvetta[134]**	**36**	3	**105**	**40**	Insect	R	FS	U	IPC	0.843*
81	ARGSTR	*Argya striata*	Jungle Babbler	21	40	2		Insect	R	FG	U	IPC	
82	CUTNIP	*Cutia nipalensis*	Himalayan Cutia		13	2		Insect	R	FS	C	IPC	
83	GARLEU	*Garrulax leucolophus*	White-crested Laughingthrush	225	138	68	71	Insect	R	FS	U	IPC	
84	GARMON	*Garrulax monileger*	Lesser Necklaced Laughingthrush	42	35	9	30	Insect	R	FS	U	IPC	
85	GRASTR	*Grammatoptila striata*	**Striated Laughingthrush[3]**		8	**55**	7	Insect	R	FS	U	IPC	0.886**
86	HETCAP	*Heterophasia capistrata*	Rufous Sibia	88	64	228	97	Insect-Nectar	R	FS	U	IPC; P	
87[##]	LEIARG	*Leiothrix argentauris*	**Silver-eared Mesia[3]**			**74**	10	Insect	R	FG	C	IPC	0.766**
88[##]	LEILUT	*Leiothrix lutea*	**Red-billed Leoithrix[34]**	2	11	**189**	**62**	Insect	AlM	FS	U	IPC	0.890**
89	MINIGN	*Minla ignotincta*	Red-tailed Minla	99	55	178	134	Insect	AlM	FS	U	IPC	

90	PTEALB	*Pterorhinus albogularis*	White-throated Laughingthrush	16		31	21	Insect	AlM	FG	C	IPC	
91	PTEPEC	*Pterorhinus pectoralis*	**Greater Neckalaced Laughingthrush**[124]	**79**	**76**		**37**	Insect	R	FS	U	IPC	0.816*
92	PTERUF	*Pterorhinus ruficollis*[1]	Rufous-necked Laughingthrush	15				Insect	R	FG	U	IPC	
93	TROERY	*Trochalopteron erythrocephalum*	Chestnut-crowned Laughingthrush	4		41	11	Insect	R	FG	U	IPC	
94	TROLIN	*Trochalopteron lineatum*[2]	Streaked Laughingthrush		1			Insect	R	FG	U	IPC	
		MEGALAIMIDAE											
95	PSIASI	*Psilopogon asiaticus*	Blue-throated Barbet	64	57	15	18	Frug	R	FG	C	SD; EE	
96	PSIFRA	*Psilopogon franklinii*	Golden-throated Barbet			3	2	Frug	R	FS	C	SD; EE	
97	PSIVIR	*Psilopogon virens*	Great Barbet	103	94	87	66	Frug	R	FS	C	SD; EE	
		MEROPIDAE											
98	MERLES	*Merops leschenaulti*[2]	Chestnut-headed Bee-eater		7			Insect	SV	FS	C	EE; IPC	
		MOTACILLIDAE											
99	ANTHOD	*Anthus hodgsoni*	Olive-backed Pipit	34	72	47	22	Insect	SV	FG	Gr	IPC	
100	MOTCIN	*Motacilla cinerea*	Grey Wagtail	2	1	2	2	Insect	AlM	OA	Gr	IPC; ND	
101	MOTCIT	*Motacilla citreola*[2]	Citrine Wagtail		1			Insect	PM	OA	Gr	IPC; ND	
102	MOTMAD	*Motacilla maderaspatensis*[2]	White-browed Wagtail		1			Insect	R	OA	Gr	IPC; ND	
		MUSCICAPIDAE											
103	ANTMON	*Anthipes monileger*	White-gorgeted Flycatcher		1	1		Insect	AlM	FG	U	IPC	
104	BRACRU	*Brachypteryx cruralis*[3]	Himalayan Shortwing			6		Insect	AlM	FS	U	IPC	
105	COPMAL	*Copsychus malabaricus*	White-rumped Shama	1			1	Insect	R	FS	U	IPC; EE	

106	COPSAU	*Copsychus saularis*	Oriental Magpie Robin	33	28	9	4	Insect	AlM	FG	U	EE; IPC	
107	CYORUB	*Cyornis rubeculoides*[4]	Blue-throated Flycatcher				2	Insect	R	OA	NA	IPC	
108	CYOTIC	*Cyornis tickelliae*[2]	Tickell's Blue Flycatcher		2			Insect	R	FG	U	IPC	
109	ENIIMM	*Enicurus immaculatus*[1]	Black-backed Forktail	1				Insect	R	FS	Aq	IPC; ND	
110	ENISCH	*Enicurus schistaceus*	Slaty-backed Forktail		2	4	3	Insect	AlM	FS	Aq	IPC; ND	
111	ENISCO	*Enicurus scouleri*[2]	Little Forktail		1			Insect	R	FS	Aq	IPC; ND	
112	EUMTHA	*Eumyias thalassinus*	Verditer Flycatcher	40	54	72	30	Insect	AlM	FG	C	IPC	
113	FICALB	*Ficedula albicilla*	Taiga Flycatcher	1	4			Insect	WV	OA	U	IPC	
114	FICHYP	*Ficedula hyperythra*[3]	Snowy-browed Flycatcher			3		Insect	AlM	FS	U	IPC	
115	FICSAP	*Ficedula sapphira*[2]	Sapphire Flycatcher		2			Insect	AlM	FS	U	IPC	
116	FICSTR	*Ficedula strophiata*	Rufous-gorgeted Flycatcher	11	29	32	23	Insect	AlM	FG	U	IPC	
117	FICTRI	*Ficedula tricolor*[3]	Slaty-blue Flycatcher			1		Insect	SV	FS	U	IPC	
118	FICWES	*Ficedula westermanni*	Little Pied Flycatcher	20	19	15	14	Insect	R	FG	MS	IPC	
119	MONCIN	*Monticola cinclorhyncha*	Blue-capped Rock Thrush	7	7	5	1	Insect	SV	FS	U	IPC	
120	MONRUF	*Monticola rufiventris*	Chestnut-bellied Rock Thrush		2	2	1	Insect	AlM	FS	U	IPC	
121	MONSOL	*Monticola solitarius*[1]	Blue Rock Thrush	1				Insect	WV	OA	U	IPC	
122	MUSFER	*Muscicapa ferruginea*	**Ferruginous Flycatcher**[34]		1	**13**	**3**	Insect	SV	FS	U	IPC	0.741*
123	MYOCAE	*Myophonus caeruleus*	Blue Whistling Thrush	115	83	107	55	Insect	AlM	FS	U	IPC; ND	
124	NILGRA	*Niltava grandis*	**Large Niltava**[34]	2	5	**16**	**25**	Insect	AlM	FS	U	IPC	0.844*
125	NILMAC	*Niltava macgrigoriae*	Small Niltava	9	14	18	20	Insect	AlM	FG	U	IPC	
126	NILSUN	*Niltava sundara*	**Rufous-bellied**	**8**	**13**	**13**	5	Insect	AlM	FS	U	IPC	0.880*

			Niltava^{123}										
127	PHOAUR	*Phoenicurus auroreus^{2}*	Daurian Redstart		1			Insect	WV	OA	U	IPC	
128	PHOCOE	*Phoenicurus coeruleocephala^{4}*	Blue-capped Redstart				1	Insect	WV	FG	U	IPC	
129	PHOFRO	*Phoenicurus frontalis*	Blue-fronted Redstart	3	1	11	5	Insect	AlM	FG	U	IPC	
130	PHOFUL	*Phoenicurus fuliginosus*	Plumbeous Water Redstart	5	9	5	8	Insect	AlM	OA	U	IPC; ND	
131	PHOHOD	*Phoenicurus hodgsoni*	Hodgson's Redstart		3	1		Insect	WV	OA	Aq	IPC	
132	PHOLEU	*Phoenicurus leucocephalus*	White-capped Water Redstart	10	9	8	7	Insect	AlM	OA	Aq	IPC; ND	
133	PHOOCH	*Phoenicurus ochruros*	Black Redstart	2	2	5		Insect	PM	OA	NA	IPC	
134	SAXFER	*Saxicola ferreus*	Grey Bushchat	16	33	25	16	Insect	AlM	OA	U	IPC	
135	SAXMAU	*Saxicola maurus*	Siberian Stonechat		3	4	2	Insect	SV	OA	Gr	IPC	
136	TARRUF	*Tarsiger rufilatus*	Himalayan Blue-tail	1		9		Insect	AlM	FS	U	IPC	
		NECTARINIIDAE											
137	AETGOU	*Aethopyga gouldiae*	Mrs Gould's Sunbird	2		12	12	Nectar	AlM	FS	C	P	
138	AETIGN	*Aethopyga ignicauda^{3}*	Fire-tailed Sunbird			5		Nectar	R	FS	C	P	
139	AETNIP	*Aethopyga nipalensis*	Green-tailed Sunbird			13	10	Nectar	R	FS	C	P	
140	AETSAT	*Aethopyga saturata*	Black-throated Sunbird	4	8	28	19	Nectar	R	FS	C	P	
141	AETSIP	*Aethopyga siparaja*	Crimson Sunbird	26	73	4	11	Nectar	R	FG	U	P	
142	ARAMAG	*Arachnothera magna*	Streaked Spiderhunter	2	6		3	Nectar	AlM	FS	C	P	
		ORIOLIDAE											
143	ORITRA	*Oriolus traillii*	**Maroon Oriole^{134}**	**9**	2	**24**	**14**	Insect	AlM	FS	C	IPC	0.894**
		PARADOXORNITHIDAE											
144	FULVIN	*Fulvetta vinipectus*	White-browed Fulvetta	11		12	10	Insect	AlM	FG	U	IPC	
145	SUTNIP	*Suthora nipalensis^{4}*	Black-throated Parrotbill				12	Insect	AlM	FS	U	IPC	

		PARIDAE											
146	MACSPI	*Machlolophus spilonotus*[3]	Yellow-cheeked Tit			6		Insect	AlM	FG	C	EE; IPC	
147	MACXAN	*Machlolophus xanthogenys*[3]	**Black-lored Tit**[3]			**34**		Insect	AlM	FG	C	EE; IPC	0.707*
148	MELSUL	*Melanochlora sultanea*[1]	Sultan Tit	7				Insect	R	FS	U	EE; IPC	
149	PARMON	*Parus monticolus*	Green-backed Tit	296	290	365	205	Insect	AlM	FG	C	EE; IPC	
		PASSERIDAE											
150	PASCIN	*Passer cinnamomeus*	Russet Sparrow	10	13	18		Gran	AlM	OA	NA	SD	
151	PASDOM	*Passer domesticus*	**House Sparrow**[23]	9	**58**	**76**		Gran	R	OA	NA	SD	0.790**
152	PASMON	*Passer montanus*	**Eurasian Tree Sparrow**[23]		**16**	**46**		Gran	R	OA	NA	SD	0.764*
		PELLORNEIDAE											
153	PELRUF	*Pellorneum ruficeps*	Puff-throated Babbler	18	9	4	11	Insect-Nectar	R	FS	C	IPC	
154	SCHCAS	*Schoeniparus castaneceps*	Rufous-winged Fulvetta	2		17	4	Insect	R	FS	C	IPC	
		PHASIANIDAE											
155	GALGAL	*Gallus gallus*[1]	Red Junglefowl	4				Gran	R	FS	U	SD	
156^{+}	LOPLEU	*Lophura leucomelanos*	Kalij Pheasant		6	2	6	Gran	R	FS	U	SD	
157^{+}	PAVCRI	*Pavo cristatus*	Indian Peafowl		48		32	Gran	R	FS	U	SD	
		PHYLLOSCOPIDAE											
158	PHYAFF	*Phylloscopus affinis*	Tickell's Leaf Warbler		2	4		Insect	SV	FG	U	IPC	
159	PHYBUR	*Phylloscopus burkii*	Green-crowned Warbler	4	17	21		Insect	AlM	FS	C	IPC	
160$^{\$}$	PHYCAN	*Phylloscopus cantator*	Yellow-vented Warbler	18	19	6	12	Insect	AlM	FS	U	IPC	
161	PHYCAS	*Phylloscopus castaniceps*	**Chestnut-crowned Warbler**[13]	**14**		**74**		Insect	AlM	FS	U	IPC	0.764*
162	PHYCHL	*Phylloscopus chloronotus*	Lemon-rumped Warbler	18	49	39	24	Insect	AlM	FS	C	IPC	
163	PHYINT	*Phylloscopus intermedius*	**White-spectacled Warbler**[13]	**18**	4	**80**	12	Insect	AlM	FS	U	IPC	0.846*

164	PHYMAC	*Phylloscopus maculipennis*	Ashy-throated Warbler	10	16	28	57	Insect	AlM	FS	C	IPC	
165	PHYMAG	*Phylloscopus magnirostris*[3]	Large-billed Leaf Warbler			8		Insect	SV	FS	C	IPC	
166	PHYPOL	*Phylloscopus poliogenys*	**Grey-cheeked Warbler[3]**	3	2	**89**	9	Insect	AlM	FS	C	IPC	0.930**
167	PHYPUL	*Phylloscopus pulcher*	Buff-barred Warbler		10	18	2	Insect	AlM	FS	C	IPC	
168	PHYREG	*Phylloscopus reguloides*[3]	**Blyth's Leaf Warbler[3]**			**16**		Insect	AlM	FG	C	IPC	0.707*
169	PHYTRO	*Phylloscopus trochiloides*	Greenish Warbler	22	22	32	8	Insect	SV	FS	C	IPC	
170	PHYWHI	*Phylloscopus whistleri*	Whistler's Warbler	48	16	27	46	Insect	AlM	FS	U	IPC	
171	PHYXAN	*Phylloscopus xanthoschistos*	Grey-hooded Warbler	192	130	163	128	Insect	AlM	FS	C	IPC	
		PICIDAE											
172	CHRGUT	*Chrysocolaptes guttacristatus*	Greater Flameback	9	6	1	2	Insect	R	FS	MS	EE; IPC	
173	CHRFLA	*Chrysophlegma flavinucha*	Greater Yellownape	26	22	24	26	Insect	R	FS	MS	EE; IPC	
174	DENDAR	*Dendrocopos darjellensis*[3]	Darjeeling Woodpecker			2		Insect	AlM	FS	MS	EE; IPC	
175	DINSHO	*Dinopium shorii*	Himalayan Flameback	2	1	1	11	Insect	R	FS	MS	EE; IPC	
176	DRYCAT	*Dryobates cathpharius*[3]	Crimson-breasted Woodpecker			2		Insect	R	FS	MS	EE; IPC	
177	GECGRA	*Gecinulus grantia*[2]	Pale-headed Woodpecker		5			Insect	R	FS	MS	EE; IPC	
178	MICBRA	*Micropternus brachyurus*[2]	Rufous Woodpecker		2			Insect	R	FS	MS	EE; IPC	
179	PICINN	*Picumnus innominatus*	Speckled Piculet			2	5	Insect	R	FS	MS	EE; IPC	
180	PICCAN	*Picus canus*	Grey-headed Woodpecker	6	11	4	5	Insect	R	FS	MS	EE; IPC	
181	PICCHL	*Picus chlorolophus*	**Lesser Yellownape[23]**	5	**16**	**10**	3	Insect	R	FS	MS	EE; IPC	0.798*
182	SASOCH	*Sasia ochracea*	White-browed Piculet		1	3		Insect	R	FS	MS	EE; IPC	
183	YUNCAN	*Yungipicus canicapillus*	**Grey-capped Pygmy Woodpecker[34]**	7	3	**40**	**13**	Insect	R	FG	MS	EE; IPC	0.837*

		PYCNONOTIDAE											
184	HEXFLA	*Hemixos flavala*	Ashy Bulbul	8	2		30	Frug	AlM	FS	C	SD	
185	HYPLEU	*Hypsipetes leucocephalus*	Black Bulbul	230	239	410	185	Frug	AlM	FS	C	SD	
186	IXOMCC	*Ixos mcclellandii*	Mountain Bulbul		2	29	28	Frug	AlM	FG	C	SD	
187	PYCCAF	*Pycnonotus cafer*	Red-vented Bulbul	365	342	150	71	Frug	R	OA	C	SD	
188	PYCLEU	*Pycnonotus leucogenis*	**Himalayan Bulbul^{12}**	**244**	**322**	34	60	Frug	AlM	OA	C	SD	0.926**
189	PYCSTR	*Pycnonotus striatus*	Striated Bulbul			12	3	Frug	AlM	FS	C	SD	
		RHIPIDURIDAE											
190	RHIALB	*Rhipidura albicollis*	White-throated Fantail	6	16	11	6	Insect	AlM	FG	C	IPC	
191	RHIAUR	*Rhipidura aureola*	White-browed Fantail	3	5	5	8	Insect	R	FG	C	IPC	
		SITTIDAE											
192	SITCAS	*Sitta cashmirensis*	**Kashmir Nuthatch^{234}**		**13**	**31**	**28**	Insect	R	FS	MS	EE; IPC	0.85*
193	SITCIN	*Sitta cinnamoventris*	Chestnut-bellied Nuthatch	10	26	25	34	Insect	R	FS	MS	EE; IPC	
194^{VU}	SITFOR	*Sitta formosa*	Beautiful Nuthatch		1		2	Insect	AlM	FS	MS	EE; IPC	
195	SITFRO	*Sitta frontalis*	Velvet-fronted Nuthatch	49	24	6	12	Insect	R	FS	MS	EE; IPC	
196	SITHIM	*Sitta himalayensis*	White-tailed Nuthatch			4	9	Insect	R	FS	MS/C	EE; IPC	
		STENOSTIRIDAE											
197	CHEHYP	*Chelidorhynx hypoxanthus*	**Yellow-bellied Fantail^{134}**	**25**	2	**61**	10	Insect	AlM	FS	C	IPC	0.904**
198	CULCEY	*Culicicapa ceylonensis*	Greyheaded CanaryFlycatcher	91	93	65	144	Insect	AlM	FS	C	IPC	
		STRIGIDAE											
199$^{\#\#}$	GLACUC	*Glaucidium cuculoides*	Asian Barred Owlet	10	18	5	8	Carni	AlM	FS	C	WD; EE	
200$^{\#\#}$	GLARAD	*Glaucidium radiatum^{2}*	Jungle Owlet		1			Carni	R	FS	C	WD; EE	

		STURNIDAE											
201	ACRTRI	*Acridotheres tristis*	**Common Myna[12]**	**82**	**265**	43	4	Omni	R	OA	NA	WD; EE	0.938**
		TICHODROMIDAE											
202	TICMUR	*Tichodroma muraria*	Wallcreeper		2		1	Insect	AlM	FG	Gr	EE; IPC	
		TIMALIIDAE											
203	ERYERY	*Erythrogenys erythrogenys[1]*	Rusty-cheeked ScimitarBabbler	5				Insect	R	FS	U	IPC	
204	POMRUF	*Pomatorhinus ruficollis[1]*	Streak-breasted ScimitarBabbler	8				Insect	R	FS	C	IPC	
205	STANIG	*Stachyris nigriceps*	**Grey-throated Babbler[14]**	**26**		9	**24**	Insect	R	FG	U	IPC	0.797**
		TROGLODYTIDAE											
206	TROTRO	*Troglodytes troglodytes[3]*	Eurasian Wren			1		Insect	AlM	FG	U	IPC	
		TURDIDAE											
207	GEOCIT	*Geokichla citrina*	Orange-headed Thrush	2		1		Insect	SV	FS	U	IPC	
208	TURATR	*Turdus atrogularis[3]*	Black-throated Thrush			12		Insect	WV	FG	C	IPC	
209	TURBOU	*Turdus boulboul*	Grey-winged Blackbird	3	22	69	14	Insect	AlM	FS	C	IPC	
210	ZOODAU	*Zoothera dauma*	Scaly Thrush			6	1	Insect	AlM	FS	C	IPC	
211	ZOODIX	*Zoothera dixoni[3]*	Long-tailed Thrush			2		Insect	AlM	FS	U	IPC	
		UPUPIDAE											
212	UPOEPO	*Upupa epops[2]*	**Common Hoopoe[2]**		**3**			Insect	AlM	OA	Gr	IPC; EE	0.707*
		VANGIDAE											
213	HEMPIC	*Hemipus picatus[1]*	Bar-winged Flycatcher-shrike	2				Insect	AlM	FG	C	IPC	
		VIREONIDAE											
214	PTEAER	*Pteruthius aeralatus*	**Blyth's Shrike-Babbler[3]**	5	2	**23**	8	Insect	AlM	FS	C	IPC	0.778*

215	PTERUF	*Pteruthius rufiventer*	Black-headed Shrike-Babbler		8	6	4	Insect	R	FS	C	IPC	
		ZOSTEROPIDAE											
216$^{\$}$	YUHBAK	*Yuhina bakeri*	White-naped Yuhina		16	20	46	Insect	R	FS	U	IPC	
217	YUHFLA	*Yuhina flavicollis*	Whiskered Yuhina	85	65	154	96	Insect-Nectar	AlM	FS	C	IPC; P	
218	YUHGUL	*Yuhina gularis*	Stripe-throated Yuhina	5		21	13	Insect-Nectar	AlM	FS	C	IPC; P	
219	YUHNIG	*Yuhina nigrimenta*	**Black-chinned Yuhina**^{134}	**50**	6	**47**	**86**	Insect-Nectar	R	FS	C	IPC; P	0.898**
220	YUHOCC	*Yuhina occipitalis*	Rufous-vented Yuhina	2	10	12	8	Insect-Nectar	AlM	FS	C	IPC; P	
221	ZOSPAL	*Zosterops palpebrosus*	Indian White-eye	106	112	56	68	Insect-Nectar	R	FG	C	IPC; P	

Notes:

[I] **Sl. no.** in bold font: Bird species Endemic/range-restricted to Eastern Himalaya ($), and protected under IUCN 2019 Near Threatened (NT), IUCN 2019 Vulnerable (VU), CITES Appendix I (#), CITES Appendix II (##), Indian Wildlife Protection Act 1972 Schedule I (+);

[II] **6-letter code**: bird species code composed of first three letters each of genus name and species epithet.

[III] **Scientific name**: bird families (50) in uppercase bold font. The ecosystem exclusive bird species species are also marked as 1 (MOAS), 2 (FAS), 3 (LCAS), and 4 (Forests);

[IV] **Common name**: indicator species of land use types, MOAS (1), FAS (2), LCAS (3), Forests (4);

[V] **Ecosystems with abundance**: Bird abundance in large cardamom-based agroforestry system (LCAS), farm-based agroforestry system (FAS), mandarin orange-based agroforestry system (MOAS), Natural forest (Forest). The abundances of indicator species are depicted in bold, whereas that of ecosystem exclusive species are underlined;

[VI] **Primary feeding guild (PFG)**: carnivores (Carni), insectivores (Insect), nectarivores (Nectar), frugivores (Frug), granivores (Gran), omnivores (Omni);

[VII] **Migratory status (Migr.)**: resident (R), altitudinal migrant (AlM), summer visitor (SV), winter visitor (WV), passage migrant (PM);

[VIII] **Habitat specialization (HS)**: forest specialist (FS), forest generalist (FG), openland (OA);

[IX] **Preferred foraging stratum(PFS)**: canopy (C), midstorey (MS), understorey (U), ground (Gr), aquatic (Aq), air (A);

[X] **Avian-mediated ecosystem services (Av.ES)**: pollination (P), seed dispersal (SD), insect pest control (IPC), vertebrate pest control & scavenging (WD), ecosystem engineering (EE), nutrient deposition (ND);

[XI] **Indicator value (IndVal)**:p<0.01 (**), p<0.05 (*); (alpha=0.05, func = "IndVal.g", duleg = FALSE, nperm = 999, At = 0.6, Bt=0.25, indvalcomp=TRUE).

Appendix C. Results of NMDS ordination based on Bray Curtis distance matrix for total bird communities and land use sensitive sub-groups along agroecosystem- forest gradient of Sikkim, Eastern Himalaya, India: total bird species (2D stress = 0.11), forest specialist understorey insectivores (FSUIBi: 2D stress = 0.16), forest specialist midstorey insectivores (FSMSIBi: 2D stress = 0.18), forest specialist canopy insectivores (FSCIBi: 2D stress = 0.15).

	Total bird species			**FSUIBi**			**FSMSIBi**			**FSCIBi**		
	NMDS1	**NMDS2**	**R^2**	**NMDS1**	**NMDS2**	**R^2**	**NMDS1**	**NMDS2**	**R^2**	**NMDS1**	**NMDS2**	**R^2**
Vectors												
Elevation	0.99853	-0.05425	**0.7934*****	0.64301	-0.76586	**0.7221*****	0.87809	0.47850	**0.5346****	0.84263	0.53849	**0.8128*****
Mean annual temperature	-0.88950	0.45694	**0.7913*****	-0.45982	0.88801	**0.7462*****	-0.91424	-0.40517	**0.3351***	-0.98817	-0.15339	**0.7035*****
Mean annual precipitation	-0.99413	-0.10821	**0.5923*****	-0.70421	0.70999	**0.6641*****	-0.81069	-0.58547	**0.3842****	-0.88246	-0.47039	**0.7335*****
Percentage canopy cover	0.99877	0.04953	0.1785	0.39338	-0.91937	0.0954	0.92690	-0.37530	0.1134	0.59821	0.80134	0.0450
Tree basal area	0.84978	0.52713	0.1723	0.58094	-0.81395	0.0462	0.98898	0.14808	0.1711	-0.01185	0.99993	0.0598
Tree density	0.61239	0.79056	**0.2903***	0.99790	-0.06471	**0.2959***	0.39415	0.91904	0.1867	0.65239	0.75788	0.1167
Tree species richness	-0.83092	-0.55639	0.1591	-0.95810	0.28644	0.0685	-0.91390	0.40595	**0.2405***	-0.37106	-0.92861	0.0686
Ecosystem			**0.4541****			**0.4023****			**0.2616***			**0.2729***
MOAS	-0.2633	-0.0709		-0.5785	0.0523		-0.5003	0.0867		-0.0933	-0.0679	
FAS	-0.3228	0.1164		0.0919	0.3845		0.0140	0.0841		-0.2781	-0.0694	
LCAS	0.2717	-0.1795		0.1545	-0.4319		0.1763	-0.2457		0.3760	0.0377	
Forest	0.3144	0.1340		0.3321	-0.0049		0.3100	0.0749		-0.0046	0.0996	

Monte-Carlo randomization test with 999 permutations was used to analyse significance (‘***’:<0.001; ‘**’:<0.01; ‘*’:<0.05) of p values.

Appendix D. Results of NMDS ordination based Bray Curtis distance matrix for the land use sensitive and ecosystem service provider sub-groups of bird communities along agroecosystem- forest gradient of Sikkim, Eastern Himalaya, India: forest specialistfrugivores and nectarivores (FSFNBi: 2D stress = 0.14), protected and endemic species (PEBi: 2D stress = 0.18), Invertebrate pest control (2D stress = 0.13), and Pollination (2D stress = 0.15).

	FSFNBi			**PEBi**			**Invertebrate pest control**			**Pollination**		
	NMDS1	**NMDS2**	**R^2**	**NMDS1**	**NMDS2**	**R^2**	**NMDS1**	**NMDS2**	**R^2**	**NMDS1**	**NMDS2**	**R^2**
Vectors												
Elevation	0.88135	0.47247	**0.5014*****	0.97537	-0.22058	**0.6479*****	0.99970	0.02434	**0.8711*****	0.99988	0.01520	**0.7624*****
Mean annual temperature	-0.87820	-0.47830	**0.6648*****	0.96527	0.26124	**0.7379*****	-0.93593	-0.35218	**0.8123*****	-0.98587	0.16750	**0.7559*****
Mean annual precipitation	-0.99976	-0.02195	**0.4062****	-0.94645	0.32284	**0.5660*****	-0.87712	0.48027	**0.6948*****	-0.95009	-0.31196	**0.6522*****
Percentage canopy cover	0.34517	0.93854	0.0945	0.67659	0.73636	**0.3240***	0.61201	0.79085	**0.3350***	0.93774	-0.34733	0.0329
Tree basal area	0.00484	0.99999	**0.4450****	0.42270	0.90627	0.1654	0.63261	0.77447	**0.2402***	0.43712	-0.89940	0.0437
Tree density	0.87123	0.49088	0.0807	0.99635	0.08538	0.1041	0.93601	-0.35198	0.1835	0.67436	0.73840	0.1684
Tree species richness	0.80188	-0.59748	0.0552	-0.46325	-0.88623	0.1899	-0.76921	0.63899	1723	-0.57893	0.81538	0.0281
Ecosystem			0.2264				**0.5096*****		**0.3859****			**0.2656***
MOAS	-0.2319	-0.0691		-0.5568	-0.5142		-0.2513	0.0446		-0.0705	0.1829	
FAS	-0.2173	0.0015		-0.4401	0.0917		-0.2630	-0.0834		-0.4539	-0.1676	
LCAS	0.3787	-0.0526		0.7324	0.1392		0.2782	-0.0927		0.4060	-0.0740	
Forest	0.0705	0.1202		0.2645	0.2833		0.2362	0.1315		0.1185	0.0586	

Monte-Carlo randomization test with 999 permutations was used to analyse significance (‘***’:<0.001; ‘**’:<0.01; ‘*’:<0.05) of p values.

Appendix E. Results of NMDS ordination based Bray Curtis distance matrix for bird communities classified based on ecosystem services along agroecosystem-forest gradient of Sikkim, Eastern Himalaya, India: Seed dispersal (2D stress = 0.13), Vertebrate pest control&Scavenging (2D stress = 0.07), Nutrient deposition (2D stress =0.15), and Ecosystem engineering (2D stress =0.12).

	Seed dispersal			**Vertebrate pest control & Scavenging**			**Nutrient deposition**			**Ecosystem engineering**		
	NMDS1	**NMDS2**	**R^2**	**NMDS1**	**NMDS2**	**R^2**	**NMDS1**	**NMDS2**	**R^2**	**NMDS1**	**NMDS2**	**R^2**
Vectors												
Elevation	0.96132	-0.27543	**0.7159*****	0.91345	-0.40695	**0.4012****	0.36853	-0.92962	**0.5081*****	0.60080	0.79940	**0.6160*****
Mean annual temperature	-0.89639	0.44326	**0.6976*****	-0.71071	0.70348	**0.4727*****	-0.09781	0.99521	**0.3957****	-0.78407	-0.62067	**0.5213*****
Mean annual precipitation	-0.84948	0.52763	**0.5476****	-0.99973	-0.02306	0.2224	-0.40989	0.91214	**0.2838***	-0.33049	-0.94381	**0.6587*****
Percentage canopy cover	0.77969	0.62617	**0.2921***	0.70476	-0.70945	**0.2618***	0.59283	-0.80533	0.0825	0.99143	-0.13064	**0.3215***
Tree basal area	0.44644	0.89481	**0.3314***	0.99989	0.01496	**0.3709***	0.56989	-0.82172	0.2010	0.86670	-0.49882	**0.3637 ***
Tree density	0.65232	0.75794	0.2151	0.61233	0.79060	**0.2863***	0.92626	-0.37688	**0.2647***	0.59885	0.80086	0.0931
Tree species richness	-0.99778	-0.06659	0.1555	-0.97295	0.23100	0.1773	-0.83947	0.54341	0.1618	-0.67808	-0.73498	**0.2722***
Ecosystem			**0.5656*****			**0.4421*****			**0.3589****			**0.5104*****
MOAS	-0.3430	-0.0331		-0.3805	-0.1942		-0.7463	-0.1750		-0.1032	-0.1215	
FAS	-0.4486	-0.0003		-0.3123	0.3128		0.3766	0.3388		-0.3833	-0.0287	
LCAS	0.3063	-0.3030		-0.0134	-0.1746		-0.0543	0.0795		0.0754	0.2537	
Forest	0.4852	0.3364		0.7062	0.0560		0.4239	-0.2433		0.4111	-0.1034	

Monte-Carlo randomization test with 999 permutations was used to analyse significance (‘***’:<0.001; ‘**’: <0.01;‘*’:<0.05) of p values.

Appendix F. Literature referred for meta-analysis of bird species richness and Shannon-Wiener diversity of Agroecosystems and other human-modified ecosystems and Forests including PAs in the biodiversity hotspots of Himalaya, Indo-Burma, Western Ghats and Sri Lanka, and elsewhere covering tropical, temperate, and neotropics regionsof the World.

Region	**Agroecosystems and other human-modified ecosystems**	**Forests including PAs**
Himalaya	Laiolo, 2004; Ahmad and Yahya, 2010; Basnet *et al.*, 2016; Elsen *et al.*, 2016, 2018; Katuwal *et al.*, 2016a, 2016b; Jolli, 2017; Srinivasan *et al.*, 2019; **Present study#**.	Chettri *et al.*, 2001, 2005; Chettri, 2010; Laiolo, 2004; Acharya *et al.*, 2010, 2011a, Acharya and Vijayan, 2011a; Joshi and Bhatt, 2011; van der Poel, 2013; Dahal *et al.*, 2014; Elsen *et al.*, 2016, 2018; Dey *et al.*, 2017; Jolli 2017; Chettri *et al.*, 2018a; Ding *et al.*, 2019; Menon *et al.*, 2019; Srinivasan *et al.*, 2019; Surya and Keitt, 2019; **Present study#**.
Indo Burma	Raman *et al.*, 1998; Raman 2001; Zhijun and Young, 2003; Kry *et al.*, 2009; Li *et al.*, 2013; Ahmed and Dey, 2014; Cottee-Jones *et al.*, 2015; Mandal and Raman, 2016; Yashmita-Ulman *et al.*, 2016; Syiem *et al.*, 2018; Warren-Thomas *et al.*, 2019.	Raman *et al.*, 1998; Barua and Sharma, 1999, 2005; Raman 2001; Eames *et al.*, 2002; Zhijun and Young, 2003; Choudhury, 2006; Thinh, 2006; Li *et al.*, 2013; Ahmed and Dey, 2014; Mandal and Raman, 2016; Zhang *et al.*, 2017a, 2017b; Syiem *et al.*, 2018; Warren-Thomas *et al.*, 2019.
Western Ghats and Sri Lanka	Daniels *et al.*, 1990; Shahabuddin, 1997; Kunte *et al.*, 1999; Raman and Sukumar, 2002; Bhagwat *et al.*, 2005a; Raman, 2006; Anand *et al.*, 2008; Ranganathan *et al.*, 2008; Sidhu *et al.*, 2010; Sreekar *et al.*, 2013; Goodale *et al.*, 2014; Kottawa-Arachchi *et al.*, 2015; Karanth *et al.*,	Daniels *et al.*, 1990; Shahabuddin, 1997; Kunte *et al.*, 1999; Raman and Sukumar, 2002; Bhagwat *et al.*, 2005a; Raman *et al.*, 2005; Raman, 2006; Ranganathan *et al.*, 2008; Sidhu *et al.*, 2010; Sreekar *et al.*, 2013; Goodale *et al.*, 2014; Subasinghe *et al.*, 2014; Karanth *et al.*, 2016; Chandran and

	2016; Chandran and Vishnudas, 2018; Chang *et al.*, 2018.	Vishnudas, 2018.
Elsewhere	Thiollay, 1995, 1997, 1999; Greenberg *et al.*, 1997a, 1997b; Daily *et al.*, 2001; Beecher *et al.*, 2002; Şekercioğlu, 2002; Mas and Dietsch, 2004; Tejeda-Cruz and Sutherland, 2004; Laiolo 2005; Peh *et al.*, 2005; Waltert *et al.*, 2004, 2005; Faria *et al.*, 2006; Genghini *et al.*, 2006; Harvey *et al.*, 2006; Santos *et al.*, 2006; Soini, 2006; Barlow *et al.*, 2007a; Beukema *et al.*, 2007; Harvey and Villalobos, 2007; Philpott *et al.*, 2007; Radford and Bennet, 2007; Van Bael *et al.*, 2007; Haslem and Bennet, 2008; Gove *et al.*, 2008; Clough *et al.*, 2009; Doxa *et al.*, 2010; Johnson *et al.*, 2010; Milder *et al.*, 2010; Azman *et al.*, 2011; Solomou and Sfougaris, 2011; Martin *et al.*, 2012; Plexida *et al.*, 2012; Hernandez *et al.*, 2013; Muhamad *et al.*, 2013; Ekroos *et al.*, 2013; Weyland *et al.*, 2014; Buechley *et al.*, 2015; Greenler and Ebersole, 2015; Maas *et al.*, 2015; Myers *et al.*, 2015; Tanalgo *et al.*, 2015; Katayama, 2016; Knight *et al.*, 2016; Prabowo *et al.*, 2016; Chiawo *et al.*, 2018; Darras *et al.*, 2018; Takeuchi, 2019.	Berg 1997; Greenberg *et al.*, 1997b; Thiollay, 1997, 1999; Daily *et al.*, 2001; Reitsma *et al.*, 2001; Lindenmayer *et al.*, 2002; Şekercioğlu, 2002; Mas and Dietsch, 2004; Tejeda-Cruz and Sutherland, 2004; Díaz *et al.*, 2005; Waltert *et al.*, 2004, 2005; Peh *et al.*, 2005; Faria *et al.*, 2006; Harvey *et al.*, 2006; Santos *et al.*, 2006; Soini, 2006; Barlow *et al.*, 2007a; Beukema *et al.*, 2007; Harvey and Villalobos, 2007; Philpott *et al.*, 2007; Van Bael *et al.*, 2007; Gove *et al.*, 2008; Milder *et al.*, 2010; Azman *et al.*, 2011; Martin *et al.*, 2012; Plexida *et al.*, 2012; Ekroos *et al.*, 2013; Hernandez *et al.*, 2013; Jankowski *et al.*, 2013; Pakkala *et al.*, 2014; Carrara *et al.*, 2015; Greenler and Ebersole, 2015; Buechley *et al.*, 2015; Mammides *et al.*, 2015; Tsai *et al.*, 2015; Knight *et al.*, 2016; Prabowo *et al.*, 2016; Asefa *et al.*, 2017; Chiawo *et al.*, 2018; Yabuhara *et al.*, 2019.

Appendix G. Butterfly species recorded in different IFS (MOAS, FAS and LCAS) and adjoining Forest of Sikkim Himalaya, India. Butterfly taxonomy, classification and auto-ecological traits follows: Haribal (1992); Karmakar *et al.* (2018); Kehimkar (2008, 2016); Sengupta *et al.* (2014); and Kunte *et al.* (2018).

[I]Sl. no.	[II]6-letter code	[III]Scientific name	[IV]Common name	[V]Ecosystems with abundance				[VI]HS	[VII]LHS	Flight period	[VIII]IndVal
				MOAS	FAS	LCAS	Forest				
		HESPERIIDAE									
		Coeliadinae									
1	BURANA	*Burara anadi anadi* de Nicéville, 1883[b]	Plain Orange Awlet		2			FS	NA	April- June	
2	BURJAI	*Burara jaina jaina* Moore, 1856	Orange Awlet	1	1			FS	Poly.	April - October	
3	BUROED	*Burara oedipodea balesis* Mabille, 1876[b]	Branded Orange Awlet		1			FS	Mono.	January - November	
4	CHOBEN	*Choaspes benjaminii japonica* Murray, 1875[b]	Indian Awlking		2			FS	Poly.	March - November	
5	HASBAD	*Hasora badra badra* Moore, 1858	Common Awl	1	3	2	1	FS	Mono.	March - November	
		Hesperiinae									
6	AMPDIO	*Ampittia dioscorides dioscorides* Fabricius, 1793 [b]	Bush Hopper		1			FG	Poly.	August - November	
7	ANCNIG	*Ancistroides nigrita diocles* Moore, 1865 [b]	Chocolate Demon		1			FG	Mono.	March - December	
8[+]	BAOFAR	*Baoris farri* Moore, 1878 [b]	Paint-brush Swift		2			FS	Poly.	March - November	
9	BORBEV	*Borbo bevani* Moore, 1878	Bevan's Swift	6	7	1		FS	Poly.	March - November	
10	BORCIN	*Borbo cinnara* Wallace, 1866	Rice Swift	5	2			FG	Poly.	March - December	
11	CALKUM	*Caltoris kumara moorei* Evans, 1926[a]	Blank Swift	1				FS	Mono.	January - December	
12	CUPPUR	*Cupitha purreea* Moore, 1877	Wax Dart	13	9	5	5	FS	Poly.	March - October	
13	HALZEM	*Halpe zema zema* Hewitson, 1877	Banded Ace	1	1			FS	NA	February - October	
14[+]	HYAADR	*Hyarotis adrastus praba* Moore, 1866[a]	Tree Flitter	1				FS	Mono.	February - October	
15	IAMSAL	*Iambrix salsala salsala* Moore, 1866	Chestnut Bob	1	2		2	FS	Poly.	March - November	
16	KORBUT	*Koruthaialos butleri* de Nicéville, 1883	Dark Velvet Bob	2	2	3		FS	NA	January - November	
17	MATARI	*Matapa aria* Moore, 1866	Common Branded Redeye	2	2		2	FS	Mono.	January - December	
18	MATSAS	*Matapa sasivarna* Moore, 1865[d]	Black-veined Branded Redeye				1	FS	NA	January - November	

19	NOTCUR	*Notocrypta curvifascia curvifascia* Felder & Felder, 1862	Restricted Demon	1		1		FS	Poly.	March - November
20	NOTFEI	*Notocrypta feisthamelii alysos* Moore, 1832	Spotted Demon	19	10	10	8	FS	Mono.	March - November
21	NOTPAR	*Notocrypta paralysos asawa* Fruhstorfer 1911[c]	Common Banded Demon			2		FS	Mono.	April - November
22	OCHBRA	*Ochlodes brahma* Moore, 1878[d]	Himalayan Darter				2	FS	NA	April - October
23	OCHSUB	*Ochus subvittatus subradiatus* Moore, 1878 [d]	Tiger Hopper				1	FS	NA	May - November
24	ORISPP	*Oriens* spp.	Dartlet	8	6	4	3	FS	Poly.	January-November
25	PARGUT	*Parnara guttatus mangala* Moore, 1865	Himalayan Straight Swift	10	2		7	FS	Poly.	March - October
26^{+}	PELSUB	*Pelopidas subochracea subochracea* Moore, 1878[d]	Large branded Swift				1	FS	NA	March-November
27^{+}	PELASS	*Pelopidas assamensis* de Nicéville, 1882[a]	Great Swift	1				FS	NA	January - November
28^{+}	POLDIS	*Polytremis discreta discreta* Elwes & Edwards, 1897	Himalayan White-fringed Swift		1	1	1	FS	NA	April - November
29	POTSPP	*Potanthus* spp.[a]	Darts	2				FG	Poly.	January-December
30	SUAGRE	*Suastus gremius gremius* Fabricius, 1798	Indian Palm Bob	1	2			FG	Poly.	January - December
		Pyrginae								
31	CELLEU	*Celaenorrhinus leucocera* Kollar, 1844	Common Spotted Flat	1	4		1	FS	Poly.	March - October
32	COLIND	*Coladenia indrani indrani* Moore, 1866[a]	Tricolour Pied Flat	2				FS	Poly.	March - November
33	GERPHI	*Gerosis phisara phisara* Moore, 1884	Dusky Yellow-breasted Flat	6	5	3	5	FS	NA	March - November
34	GERSIN	*Gerosis sinica narada* Moore, 1884[a]	White Yellow-breasted Flat	2				FS	NA	March - December
35	MOOTRI	*Mooreana trichoneura pralaya* Moore, 1865[d]	Yellow-veined Flat				1	FS	NA	April - November
36	PSEDAN	*Pseudocoladenia dan fabia* Evans, 1949	Himalayan Fulvous Pied Flat	8	6	6		FS	Mono.	March - November
37	SARDAS	*Sarangesa dasahara dasahara* Moore, 1866	Common Small Flat	18	11	5	5	FG	Poly.	February - November
38	TAGLIT	*Tagiades litigiosa litigiosa* Möschler, 1878	Water Snow Flat	4	8	3		FS	Poly.	March - November
39	TAGMEN	*Tagiades menaka menaka* Moore, 1865[a]	Spotted Snow Flat	4				FS	Mono.	March - November
40	TAGPAR	*Tagiades parra gala* Evans, 1949[d]	Sikkim Multi-spotted Flat				1	FS	NA	March-November

		LYCAENIDAE									
		Aphnaeinae									
41[#]	SPILOH	*Spindasis lohita himalayanus* Moore, 1884[a]	Long-banded Silverline	4				FS	Mono.	March - October	
		Lycaeninae									
42	HELBRA	*Heliophorus brahma brahma* Moore, 1858	Golden Sapphire	39	67	78	43	FS	Poly.	April -December	
43	HELEPI	*Heliophorus epicles latilimbata*Fruhstorfer, 1908	Purple Sapphire	23	35	68	54	FG	Poly.	February - October	
44[$]	HELHYB	*Heliophorus hybrida* Tytler, 1912[c]	Hybrid Sapphire			1		FS	NA	October	
45[#]	HELMOO	*Heliophorus moorei moorei* Hewitson, 1865	Azure Sapphire	2	2	2		FS	Mono.	March-October	
46	HELTAM	*Heliophorus tamu tamu* Kollar, 1844	Himalayan Powdery Green Sapphire			1	12	FS	Poly.	April - November	
		Miletinae									
47	TARHAM	*Taraka hamada mendesia* Fruhstorfer, 1918	Forest Pierrot	1	1			FS	NA/ Insect.	March - December	
		Polyommatinae									
48	ACYPUS	*Acytolepis puspa gisca* Fruhstorfer, 1910	**Common Hedge Blue**[1, 2, 3]	**21**	**20**	**12**	1	FS	Poly.	January - December	0.874*
49	ANTEMO	*Anthene emolus emolus* Godart, 1824[a]	Common Ciliate Blue	1				FS	Poly.	March - November	
50	CASROS	*Castalius rosimon rosimon* Fabricius, 1775[d]	Common Pierrot				1	FG	Mono.	January - December	
51	CATPAN	*Catochrysops panormus exiguus* Distant, 1886[b]	Silver Forget-me-not		2			FS	Poly.	April-November	
52	CELARG	*Celastrina argiolus sikkima* Moore, 1883	Hill Hedge Blue		6		1	FS	Mono.	April -December	
53	CELHUE	*Celastrina huegelii oreana* Moore, 1883	Large Hedge Blue	7	10	3	1	FG	Mono.	March -October	
54	CELLAV	*Celastrina lavendularis limbatus* Moore, 1879[a]	Plain Hedge Blue	1				FS	NA	January - December	
55	CELMAR	*Celatoxia marginata marginata* de Nicéville, 1884[c]	Sikkim Margined Hedge Blue			1		FS	NA	February - December	
56[#]	EUCCNE	*Euchrysops cnejus cnejus* Fabricius, 1798	Gram Blue		1	5		FG	Poly.	January - December	
57	FRETRO	*Freyeria trochylus orientalis* Forster, 1980	Grass Jewel	1	1			FG	Poly.	January - December	
58	JAMALE	*Jamides alecto eurysaces* Fruhstorfer, 1915	Metallic Cerulean		5	4	4	FS	Poly.	January - November	

59	JAMBOC	*Jamides bochus bochus* Stoll, 1782	Dark Cerulean	21	18	15	15	FS	Poly.	January - December	
60	JAMCEL	*Jamides celeno celeno* Cramer, 1775	Common Cerulean	73	104	37	20	FG	Poly.	January - December	
61[#]	JAMELP	*Jamides elpis pseudodelpis* Butler, 1879	Glistening Cerulean	1	1	5	1	FS	Poly.	March - November	
62[#]	LAMBOE	*Lampides boeticus* Linnaeus, 1767	Pea Blue	2	4		4	FG	Poly.	January - December	
63	LEPPLI	*Leptotes plinius plinius* Fabricius, 1793[d]	Zebra Blue				4	FG	Poly.	January - December	
64	LESTRA	*Lestranicus transpectus* Moore, 1879[c]	White-banded Hedge Blue			1		FS	NA	March - December	
65[#]	PORHEW	*Poritia hewitsoni hewitsoni* Moore, 1866	Common Gem	1	3	1	1	FS	Poly.	March - December	
66	PSEMAH	*Pseudozizeeria maha maha* Kollar, 1844	Himalayan Pale Grass Blue	13	13	11	3	FG	Poly.	January - December	
67	UDADIL	*Udara dilectus dilectus* Moore, 1879	Pale Hedge Blue	10		6	1	FS	NA	Januray - December	
68	ZIZKAR	*Zizeeria karsandra* Moore, 1865	Dark Grass Blue	2	6	12	1	FG	Mono.	January - December	
69	ZIZOTI	*Zizina otis otis* Fabricius, 1787[b]	Lesser Grass Blue		3			FG	Mono.	January - December	
70	ZIZHYL	*Zizula hylax hylax* Fabricius, 1775	Tiny Grass Blue	3		1		FG	Poly.	March - November	
		Theclinae									
71	ANCCTE	*Ancema ctesiactesia* Hewitson, 1865	Himalayan Bi-spot Royal	1	3	1		FS	Mono.	April - October	
72	ARHABS	*Arhopala abseus indicus* Riley, 1923	Aberrant Oakblue		1	2	2	FS	Mono.	March - December	
73	ARHATR	*Arhopala atrax* Hewitson, 1862[b]	Indian Oakblue		2			FS	Poly.	January - December	
74	ARHCEN	*Arhopala centaurus pirithous* Moore 1884[c]	Centaur Oakblue			1		FS	Poly.	February - December	
75	ARHRAM	*Arhopala rama rama* Kollar, 1844	Himalayan Dark Oakblue	1	1			FS	Mono.	February - November	
76	CHEFRE	*Cheritra freja evansi* Cowan, 1965	Khasi Common Imperial	6	6	3	2	FS	Poly.	April - November	
77[$]	DEUEPI	*Deudorix epijarbas amatius* Fruhstorfer, 1912[a]	Cornelian	7				FG	Poly.	March - December	
78	FLOADR	*Flos adriana* de Nicéville, 1884	Variegated Plushblue		1	2		FS	NA	March - November	
79[#]	FLOARE	*Flos areste* Hewitson, 1862	Tailless Plushblue	1		2	2	FS	NA	February - November	
80	FLOASO	*Flos asoka* de Nicéville, 1884[a]	**Spangled Plushblue[1]**	**4**				FS	NA	April -November	0.707*
81	HYPERY	*Hypolycaena erylus himavantus* Fruhstorfer, 1912	Sikkim Common Tit		1	2		FS	Poly.	March - December	
82	LOXATY	*Loxura atymnus continentalis* Fruhstorfer, 1912	Yamfly	1	1			FS	Poly.	April - November	
83	RAPPHE	*Rapala pheretima petosiris* Hewitson, 1863	Copper Flash	14			1	FS	Mono.	March - December	

84[#]	RAPVAR	*Rapala varuna varuna* Hewitson, 1863	**Indigo Flash**[1]	**12**	2		1	FS	Poly.	October - May	0.816*
85[#]	SINNAS	*Sinthusa nasaka amba* Kirby, 1878[a]	Narrow Spark	7				FS	NA	April -September	
86	SURQUE	*Surendra quercetorum quercetorum* Moore, 1857	Himalayan Common Acacia Blue	1	5		4	FS	Mono.	February - December	
87	TICACT	*Ticherra acte acte* Moore, 1857[a]	Blue Imperial	1				FS	Mono.	February - November	
88	ZELAMA	*Zeltus amasa amasa* Hewitson, 1865	Fluffy Tit			1	4	FS	Mono.	April - December	
		NYMPHALIDAE									
		Acraeinae									
89	ACRISS	*Acraea issoria issoria* Hübner, 1818	Himalayan Yellow Coster	1	2	11	29	FS	Poly.	March - October	
90	CETBIB	*Cethosia biblis tisamena* Fruhstorfer, 1912	Himalayan Red Lacewing	24	8	19	9	FS	Mono.	February - November	
91	CETCYA	*Cethosia cyanecyane* Drury, 1770	Leopard Lacewing	9	1	4	7	FS	Mono.	January - November	
		Apaturinae									
92[#]	EURCON	*Euripus consimilis consimilis* Westwood, 1851[b]	Painted Courtesan		2			FS	Mono.	January - December	
93[#]	EURNYC	*Euripus nyctelius* Doubleday, 1845	Courtesan	2	1	2	1	FS	Poly.	March - November	
94	HERMAR	*Herona marathus marathus* Doubleday, 1848[b]	Pasha		1			FS	NA	April - October	
95	HESNAM	*Hestinalis nama* Doubleday, 1844	Circe	24	8	7	8	FS	Mono.	February - November	
96	MIMAMB	*Mimathyma ambica ambica* Kollar, 1844[a]	Himalayan Purple Emperor	1				FS	Mono.	April -October	
97	ROHPAR	*Rohana parisatis parisatis* Westwood, 1851	Black Prince	3	3			FS	Mono.	March - November	
98[$]	SEPCHA	*Sephisa chandra chandra* Moore, 1857	Eastern Courtier	2			1	FS	Mono.	April - November	
		Biblidinae									
99	ARIARI	*Ariadne ariadne pallidior* Fruhstorfer, 1899	Large Angled Castor		32		1	FS	Poly.	January - December	
100	ARIMER	*Ariadne merione tapestrina* Moore, 1884	Intricate Common Castor	4	25	5	4	FG	Poly.	January - December	
		Charaxinae									
101	CHABER	*Charaxes bernardus hierax* Felder & Felder, 1866[b]	Variable Tawny Rajah		1			FS	Poly.	March - November	
102	CHABHA	*Charaxes bharata* Felder & Felder, 1867	Common Nawab	13	9	6	1	FS	Poly.	January-December	
103[#]	CHADOL	*Charaxes dolon centralis* Rothschild, 1899[d]	Sikkim Stately Nawab				1	FS	NA	April - August	

		Cyrestinae									
104	CHERIS	*Chersonesia risa risa* Doubleday, 1848	Common Maplet	3	1	1	1	FS	NA	March - December	
105	CYRTHY	*Cyrestis thyodamas thyodamas* Doyère, 1840	Common Map	24	15	10	10	FS	Mono.	January - December	
106	PSEWED	*Pseudergolis wedah wedah* Kollar, 1844[c]	Himalayan Tabby			5		FS	Mono.	March - October	
107	STINIC	*Stibochiona nicea nicea* Gray, 1846	Himalayan Popinjay	2		1	1	FS	Mono.	March -October	
		Danainae									
108	DANCHR	*Danaus chrysippus chrysippus* Linnaeus, 1758	Plain Tiger	1	6		3	FG	Poly.	January-December	
109	DANGEN	*Danaus genutia genutia* Cramer, 1779	Striped Tiger	18	32	2	10	FG	Poly.	January-December	
110	EUPALG	*Euploea algea deione* Westwood, 1848[b]	Long Branded Blue Crow		1			FS	Poly.	April -November	
111+	EUPCOR	*Euploea core core* Cramer, 1780	Common Crow	11	6	2	1	FG	Poly.	January - December	
112+	EUPKLU	*Euploea klugii klugii* Moore, 1857	Blue King Crow	4	1	1		FS	Poly.	April - November	
113#	EUPMID	*Euploea midamus rogenhoferi* Felder & Felder, 1865	**Blue-spotted Crow**[1, 2]	**14**	**11**	4		FS	Poly.	January - December	0.889*
114+	EUPMUL	*Euploea mulciber mulciber* Cramer, 1777	Striped Blue Crow	40	16	13	12	FG	Poly.	January - December	
115+	EUPRAD	*Euploea radamanthus* Fabricius, 1793[a]	Magpie Crow	1				FS	Poly.	March - November	
116	EUPSYL	*Euploea sylvester hopei* Felder & Felder, 1865	Double-branded Blue Crow	6	1	1	2	FS	Poly.	March-October	
117	PARAGL	*Parantica aglea melanoides* Moore, 1883	Himalayan Glassy Tiger	23	17	7	8	FS	Poly.	January - December	
118	PARMEL	*Parantica melaneus plataniston* Fruhstorfer, 1910	Himalayan Chocolate Tiger	1	4		5	FS	Mono.	February - November	
119	PARSIT	*Parantica sita sita* Kollar, 1844	Chestnut Tiger	4	10	3	4	FS	Poly.	March - December	
120	TIRLIM	*Tirumala limniace exoticus* Gmélin, 1790	Blue Tiger	15	14	7	7	FG	Poly.	January - December	
121	TIRSEP	*Tirumala septentrionis septentrionis* Butler, 1874	Dark Blue Tiger	15	2	2	2	FS	Poly.	January - December	
		Heliconinae									
122	ARGCHI	*Argynnis childreni childreni* Gray, 1831	Large Silverstripe	3	2	5		FG	Mono.	April - November	
123	ARGHYP	*Argynnis hyperbius hyperbius* Linnaeus, 1763	Tropical Fritillary	10	8	5	1	FG	Poly.	January - December	
124	ISSISS	*Issoria issaea* Doherty, 1886[c]	Himalayan Queen Fritillary			1		FG	Mono	January - December	

125	CIRAOR	*Cirrochroa aoris aoris* Doubleday, 1847	Large Yeoman	54	23	21	15	FS	Mono.	March - November	
126	CIRTYC	*Cirrochroa tyche mithila* Moore, 1872	**Common Yeoman**[1]	**18**		1	4	FS	NA	March - November	0.808*
127[#]	PHAALC	*Phalanta alcippe alcippoides* Moore, 1900[a]	Small Leopard	1				FS	Poly.	March - December	
128	PHAPHA	*Phalanta phalantha phalantha* Drury, 1773	Common Leopard	6	1	3		FG	Poly.	January - December	
		Libytheinae									
129	LIBMYR	*Libythea myrrha sanguinalis* Fruhstorfer, 1898	Club Beak	8	2			FS	Mono.	March - October	
130[#]	LIBLEP	*Libythea lepita lepita* Moore, 1858[a]	Himalayan Common Beak	1				FS	Poly.	March - September	
		Limenitinae									
131	ABRGAN	*Abrota ganga ganga* Moore, 1857	Sergeant Major	4	1		2	FS	NA	March - September	
132	ATHCAM	*Athyma cama cama* Moore, 1857	Himalayan Orange Staff Sergeant	7	3	1	1	FS	Mono.	March -December	
133	ATHINA	*Athyma inara inara* Westwood, 1850	Himalayan Colour sergeant	1		1	1	FS	Mono.	January - December	
134[$]	ATHJIN	*Athyma jina jina* Moore, 1857	Sullied Bhutan Sergeant	1		2		FS	NA	February -December	
135	ATHPER	*Athyma perius perius* Linnaeus, 1758	Common Sergeant			1	1	FG	Mono.	February - December	
136[#]	ATHRAN	*Athyma ranga ranga* Moore, 1857[d]	Blackvein Sergeant				1	FS	Poly.	March - November	
137	ATHOPA	*Athyma opalina opalina* Kollar, 1844[c]	Himalayan Hill Sergeant			1		FS	Poly.	March - November	
138	ATHSEL	*Athyma selenophora selenophora* Kollar, 1844	Staff Sergeant	3			1	FS	Mono.	February - December	
139	ATHZER	*Athyma zeroca zeroca* Moore, 1872[a]	Khasi Small Staff Sergeant	1				FS	NA	March - December	
140	EUTACO	*Euthalia aconthea garuda* Moore, 1857	Common Baron	2	1	1	1	FG	Poly.	January - December	
141[#]	EUTDUD	*Euthalia duda duda* Straudinger, 1886	Himalayan Blue Duchess	3		2	3	FS	Mono.	July - November	
142[#]	EUTFRA	*Euthalia franciae franciae* Gray, 1846[c]	**Himalayan French Duke**[3]			**4**		FS	NA	March - September	0.707*
143	EUTMON	*Euthalia monina kesava* Moore, 1859[d]	Powdered Baron				2	FS	Mono.	March - December	
144	EUTPHE	*Euthalia phemius phemius* Doubleday, 1848	White-edged Blue Baron	5	1	2	1	FS	Mono.	March - November	
145[$]	EUTTEL	*Euthalia telchinia* Ménétriés, 1857[a]	Blue Baron	1				FS	NA	March - October	
146	MODPRO	*Moduza procris procris* Cramer, 1777	Commander		3		3	FG	Poly.	February - December	
147	NEPPSE	*Neptis pseudovikasi* Moore, 1899	False Dingy Sailer	4		2	3	FS	NA	March - November	

148[#]	NEPSOM	*Neptis soma soma* Moore, 1858	Himalayan Sullied Sailer		2		1	FS	Mono.	February - November	
149[#]	NEPANA	*Neptis ananta ochracea* Evans, 1924[a]	Yellow Sailer	1				FS	Mono.	March - December	
150	NEPCLI	*Neptis clinia susruta* Moore 1872	Cleamy Sailer		2	5		FS	Mono.	February - November	
151	NEPHYL	*Neptis hylas varmosa* Moore, 1872	Common Sailer	131	124	45	66	FG	Poly.	January - December	
152	NEPMIA	*Neptis miah miah* Evans, 1857[d]	Small Yellow Sailer				2	FS	NA	March - November	
153	PANHOR	*Pantoporia hordonia hordonia* Stoll, 1790	Common Lascar	24	12	9	8	FS	Poly.	March - November	
154[#]	PARDUD	*Parasarpa dudu dudu* Westwood, 1850[d]	White Commodore				1	FS	Mono.	March- November	
155	SUMDAR	*Sumalia daraxa daraxa* Doubleday, 1848	Green Commodore	1	18	23	5	FS	Poly.	March - November	
156	TANJAH	*Tanaecia jahnu jahnu* Moore, 1857	Plain Earl	2	9		5	FS	NA	February - October	
157	TANJUL	*Tanaecia julii appiades* Ménétriés, 1857	Common Earl	36	43	6	31	FS	NA	March - November	
158[#]	TANLEP	*Tanaecia lepidea lepidea* Butler, 1868[c]	Himalayan Grey Count			2		FG	Poly.	February - November	
		Nymphalinae									
159	AGLCAS	*Aglais caschmirensis aesis* Fruhstorfer, 1912	**Indian Tortoiseshell**[2, 3, 4]		**75**	**72**	**36**	FG	Poly.	February - November	0.943**
160	DOLBIS	*Doleschallia bisaltide indica* Moore, 1899	Himalayan Autumn Leaf	3	1	4		FS	Poly.	January-November	
161	HYPBOL	*Hypolimnas bolina jacintha* Drury, 1773	Great Eggfly	10	8		2	FG	Poly.	January - December	
162[#]	HYPMIS	*Hypolimnas misippus* Linnaeus, 1764	Danaid Eggfly	11	9	1	1	FG	Poly.	January - December	
163	JUNALM	*Junonia almana almana* Linnaeus, 1758	Peacock Pansy	10	4			FG	Poly.	January - December	
164	JUNATL	*Junonia atlites atlites* Linnaeus, 1763	Grey Pansy	6	14	1	1	FG	Poly.	January - December	
165	JUNHIE	*Junonia hierta magna* Evans, 1923	Yellow Pansy		2	1	1	FG	Poly.	January - December	
166	JUNIPI	*Junonia iphita iphita* Cramer, 1779	Chocolate Pansy	45	7	24	5	FS	Poly.	January - December	
167	JUNLEM	*Junonia lemonias lemonias* Linnaeus, 1758	Lemon Pansy	73	14	1	14	FG	Poly.	January - December	
168	JUNORI	*Junonia orithya swinhoei* Butler, 1885[b]	Pale Blue Pansy		1			FG	Poly.	January - December	
169	KALINA	*Kallima inachus inachus* Doyere, 1840	Orange Oakleaf	2	1	1	2	FS	Mono.	April -November	
170	SYMHYP	*Symbrenthia hypselis cotanda* Moore, 1874	Himalayan Spotted Jester		1	5		FS	Poly.	March - October	
171	SYMLIL	*Symbrenthia lilaea khasiana* Moore, 1874	Khasi Common Jester	46	31	15	16	FS	Mono.	January - December	
172[#]	SYMNIP	*Symbrenthia niphanda niphanda* Moore, 1872	Blue-tailed Jester			5	1	FS	Mono.	March - November	

173[$]	SYMSIL	*Symbrenthia silana* de Nicéville, 1885[c]	Scarce Jester			1		FS	NA	March - November	
174	VANCAR	*Vanessa cardui* Linnaeus, 1758	Painted Lady	7	29	31	25	FG	Poly.	January - December	
175	VANIND	*Vanessa indica indica* Herbst, 1794	Indian Red Admiral	49	26	20	5	FG	Poly.	March - October	
		Satyrinae									
176	ELYHYP	*Elymnias hypermnestra undularis* Drury, 1779	Common Palmfly		1		1	FG	Poly.	January - December	
177	ELYMAL	*Elymnias malelas malelas* Hewitson, 1863	Spotted Palmfly		2		2	FS	Mono.	March - November	
178	ELYPAT	*Elymnias patna patna* Westwood, 1851	Blue-striped Palmfly	1	3	1		FS	Mono.	March-November	
179[#]	ELYVAS	*Elymnias vasudeva* Moore, 1857[d]	Jezebel Palmfly				1	FS	NA	March-November	
180	LETCHA	*Lethe chandica chandica* Moore, 1857	Angled Red Forester	2	1	7	3	FS	NA	February - November	
181	LETCON	*Lethe confusa confusa* Aurivillius, 1898	Himalayan Banded Treebrown	41	29	48	26	FS	Mono.	February - November	
182[$]	LETDIS	*Lethe distans* Butler, 1870	Scarce Red Forester	1		1	1	FS	NA	March - May	
183	LETEUR	*Lethe europa niladana* Fruhstorfer, 1911[b]	Himalaya Bamboo Treebrown		2			FS	Mono.	March-November	
184	LETMEK	*Lethe mekara mekara* Moore, 1857	**Common Red Forester**[1]	**64**	1			FS	Mono.	April - November	0.992**
185[#]	LETSER	*Lethe serbonis* Hewitson, 1876	Brown Forester	1			1	FS	Mono.	June - November	
186	LETSID	*Lethe sidonis* Hewitson, 1863	Common Woodbrown	1	4	8		FS	Mono.	April - October	
187[#]	LETSIN	*Lethe sinorix sinorix*Hewitson, 1863	Tailed Red Forester	6	2	11	6	FS	NA	April - November	
188	LETSUR	*Lethe sura* Doubleday, 1849[c]	Lilacfork			1		FS	Mono.	April - November	
189	LETVER	*Lethe verma sintica* Fruhstorfer, 1911	**Straight Banded Treebrown**[1, 2, 3]	**30**	**13**	**29**	5	FS	Mono.	April - November	0.967**
190[#]	LETVIS	*Lethe visrava* Moore, 1865[c]	White-edged Woodbrown			2		FS	NA	May - October	
191	MELLED	*Melanitis leda leda* Linnaeus, 1758	Common Evening Brown	12	18	8	9	FG	Poly.	January - December	
192	MELPHE	*Melanitis phedima bela* Moore, 1857	Bengal Dark Evening Brown		12	1	5	FS	Poly.	April - December	
193[#]	MELZIT	*Melanitis zitenius zitenius* Herbst, 1796	Himalayan Great Evening Brown	2	18	5	6	FS	Poly.	April - December	
194	MYCFRA	*Mycalesis francisca sanatana* Moore, 1857	Himalayan Lilacine Bushbrown	1	1		1	FS	Mono.	March - October	
195[#]	MYCHER	*Mycalesis heri heri* Moore, 1857[b]	Moore's Bushbrown		1			FS	NA	August -October	

196	MYCINT	*Mycalesis intermedia* Moore, 1892	Intermediate Bushbrown	4	1	1	2	FS	NA	November	
197	MYCMIN	*Mycalesis mineus mineus* Linnaeus, 1758	Dark- Brand Bushbrown	3	1	1	3	FG	Poly.	April - December	
198	MYCPER	*Mycalesis perseus blasius* Fabricius, 1798	Himalayan Common Bushbrown	46	48	37	22	FG	Poly.	January - December	
199	MYCVIS	*Mycalesis visala visala* Moore, 1857	Long-Brand Bushbrown	1	6	1	5	FS	Mono.	January - December	
200	ORIDAM	*Orinoma damaris damaris* Gray, 1846	Tiger Brown	1		5		FS	Mono.	April - November	
201	ORSMED	*Orsotriaena medus medus* Fabricius, 1775	Medus Brown	59	102	15	34	FS	Poly.	January - December	
202	TELMAL	*Telinga malsara* Moore, 1857	White-line Bushbrown	1		5	1	FG	NA	March - November	
203	TELNIC	*Telinga nicotia* Westwood, 1850[a]	Bright-eye Bushbrown	1				FS	NA	April -August	
204	YPTAST	*Ypthima asterope mahratta* Moore, 1884	**Common Three-ring[1]**	**25**	4	1	1	FG	Mono.	January - December	0.820*
205	YPTBAL	*Ypthima baldus baldus* Fabricius, 1775	Common Five-ring	239	157	102	79	FS	Mono.	February - October	
206	YPTHUE	*Ypthima huebneri huebneri* Kirby, 1871	Common Four-ring	8	2	10	2	FG	Mono.	March - November	
207	YPTNEW	*Ypthima newara newara* Moore, 1874[d]	Himalayan Newar Three-ring				2	FG	NA	April - November	
208	YPTSAK	*Ypthima sakra sakra* Moore, 1857	East Himalayan Five-ring	34	19	98	109	FS	Mono.	February - November	
		PAPILIONIDAE									
		Papilioninae									
209	ATRAID	*Atrophaneura aidoneus*Doubleday, 1845[a]	Lesser Batwing	4				FS	Mono.	April - November	
210	ATRVAR	*Atrophaneura varuna astorion* Westwood, 1842	Common Batwing	2	1		1	FS	Mono.	March - November	
211	BYADAS	*Byasa dasarada dasarada* Moore, 1858[d]	East Himalayan Great Windmill				1	FS	Mono.	March - October	
212	BYAPOL	*Byasa polyeuctes polyeuctes* Doubleday, 1842[d]	Common Windmill				2	FS	Mono.	March-November	
213	GRAAGA	*Graphium agamemnon agamemnon* Linnaeus, 1758	Tailed Jay	13	4		5	FG	Poly.	January - December	
214	GRACLO	*Graphium cloanthus cloanthus* Westwood, 1841	Himalayan Glassy Bluebottle	4	5	3	1	FS	Poly.	March - November	
215	GRADOS	*Graphium doson axionides* Page & Treadaway, 2014[b]	Himalayan Common Jay		1			FG	Poly.	January - December	

216	GRAEUR	*Graphium eurypylus acheron* Moore, 1885[a]	Great Jay	1				FS	Poly.	March - September	
217	GRAMAC	*Graphium macareus indicus* Rothschild, 1895[b]	East Himalayan Lesser Zebra		1			FS	NA	March - May	
218	GRASAR	*Graphium sarpedon sarpedon* Linnaeus, 1758	Common Bluebottle	8	9	2	1	FS	Poly.	January - December	
219	PACARI	*Pachliopta aristolochiae geniopeltis* Rothschild, 1908	Indo-Chinese Common Rose			3	1	FG	Mono.	January - December	
220	PAPALC	*Papilio alcmenor alcmenor* C. & R. Felder, 1864	Redbreast	20	7	2	4	FS	NA	March - October	
221	PAPARC	*Papilio arcturus arcturus* Westwood, 1842[d]	East Himalayan Blue Peacock				1	FS	Poly.	March - August	
222	PAPBIA	*Papilio bianor ganesa* Doubleday, 1842	East Himalayan Common Peacock	16	7	13	8	FS	Poly.	March - October	
223	PAPAGE	*Papilio agestor agestor* Gray, 1831[c]	Tawny Mime			3		FS	Poly.	March - August	
224[$]	PAPCLY	*Papilio clytia clytia* Linnaeus, 1758	Common Mime	1	2		2	FG	Poly.	March - December	
225[#]	PAPEPY	*Papilio epycides epycides* Hewitson, 1862[b]	Himalayan Lesser Mime		1			FS	Poly.	March - May	
226	PAPHEL	*Papilio helenus helenus* Linnaeus, 1758	Red Helen	39	24	10	8	FS	Poly.	March - November	
227	PAPKRI	*Papilio krishna krishna* Moore, 1858	Himalayan Krishna Peacock	1	1	3		FS	Poly.	April - November	
228	PAPMAC	*Papilio machaon* Linnaeus, 1758[d]	Common Yellow Swallowtail				1	FG	Poly.	March - September	
229	PAPMEM	*Papilio memnon agenor* Linnaeus, 1758	Great Mormon	29	12	9	14	FG	Poly.	March - November	
230	PAPNEP	*Papilio nephelus chaon* Westwood, 1845	Yellow Helen	47	18	16	10	FS	Poly.	February - October	
231	PAPPAR	*Papilio paris paris* Linnaeus, 1758	**Paris Peacock**[1]	**21**	1	6	5	FS	Poly.	March - November	0.798*
232	PAPPOL	*Papilio polytes romulus* Cramer, 1775	**Common Mormon**[1, 2, 3]	**59**	**32**	**14**	6	FG	Poly.	January - December	0.945**
233	PAPPRO	*Papilio protenor euprotenor* Fruhstrorfer, 1908	Himalayan Spangle	29	7	9	5	FS	Poly.	March - November	
234[C]	TROAEA	*Troides aeacus aeacus* C. & R. Felder, 1860	Golden Birdwing	1			1	FS	Mono.	May - October	
235[C]	TROHEL	*Troides helena cerberus* C. & R. Felder, 1865	Common Birdwing	6	1		2	FS	Poly.	March - November	
		PIERIDAE									
		Coliadinae									
236	CATPOM	*Catopsilia pomona pomona* Fabricius, 1775	Lemon Emigrant	3	11	3	3	FG	Poly.	January - December	
237	CATPYR	*Catopsilia pyranthe* Linnaeus, 1758	Mottled Emigrant	1	12	4	6	FG	Poly.	January - December	

238	COLFIE	*Colias fieldii*Ménétriés, 1855	Himalayan Dark Clouded Yellow	1		1	5	FG	Poly.	February - December	
239	EURAND	*Eurema andersoni jordani* Corbet & Pendlebury, 1932	Sikkim One Spot Grass Yellow	4	5		1	FS	Mono.	March - December	
240	EURBLA	*Eurema blanda silhetana* Wallace, 1867	Three Spot Grass Yellow	204	192	50	56	FS	Poly.	January - December	
241	EURBRI	*Eurema brigitta rubella* Wallace, 1867	Small Grass Yellow		1	1	11	FG	Poly.	January - December	
242	EURHEC	*Eurema hecabe hecabe* Linnaeus, 1758	Common Grass Yellow	35	16	14	14	FG	Poly.	January - December	
243	EURLAE	*Eurema laeta sikkima* Moore, 1906	Sikkim Spotless Grass Yellow	4	9	2	11	FG	Mono.	January - December	
244	GANHAR	*Gandaca harina assamica* Moore, 1906	Assam Tree Yellow	6	2	1	5	FS	NA	March - December	
		Pierinae									
245	APPLAL	*Appias lalage lalage* Doubleday, 1842[d]	Himalayan Spot Puffin				2	FS	NA	February - November	
246	APPLYN	*Appias lyncida eleonora* Boisduval, 1836	Chocolate Albatross	4	1			FS	Mono.	March - December	
247	CEPNAD	*Cepora nadina nadina* Lucas, 1852	Lesser Gull	5	4	3	2	FS	Mono.	January - December	
248	CEPNER	*Cepora nerissa nerissa* Fabricius, 1775	Common Gull	3	1	1	2	FG	Poly.	January - December	
249	DELACA	*Delias acalis pyramus* Wallace, 1867	Himalayan Red-breast Jezebel	5	14	2	5	FS	NA	March - November	
250	DELAGO	*Delias agostina agostina* Hewitson, 1852	Yellow Jezebel	22	16		3	FS	Mono.	February - November	
251	DELBEL	*Delias belladonna ithiela* Butler, 1869	Sikkim Hill Jezebel	4		9	7	FS	Poly.	April - July; September-November	
252	DELDES	*Delias descombesi descombesi* Boisduval, 1836	Red-spot Jezebel	59	34	1	11	FS	Mono.	March - December	
253	DELHYP	*Delias hyparete indica* Wallace, 1867	Painted Jezebel	1	4		3	FS	Mono.	January - December	
254	DELPAS	*Delias pasithoe pasithoe* Linnaeus, 1767	Red-base Jezebel	5	15		1	FS	Poly.	February - December	
255[$]	DELSAN	*Delias sanaca* Moore, 1857[d]	Pale Jezebel				2	FS	Poly.	March - July	
256	HEBGLA	*Hebomoia glaucippe glaucippe* Linnaeus, 1758	Great Orange-tip	19	13	1	2	FS	Mono.	February - December	
257	IXIPYR	*Ixias pyrene familiaris* Butler, 1874	Yellow Orange-tip	14	11	7	7	FS	Mono.	Januray - December	
258	PIEBRA	*Pieris brassicae nepalensis* Doubleday, 1846	**Large Cabbage White**[1, 2, 3]	**35**	**10**	**13**	2	FG	Poly.	March - December	0.867*
259	PIECAN	*Pieris canidia indica* Evans, 1926	Himalayan Cabbage White	196	186	174	64	FG	Poly.	January - December	

260	PIEMEL	*Pieris melete ajaka* Moore, 1865[b]	Himalayan Green-veined White		2			FS	NA	March - November	
261	PRITHE	*Prioneris thestylis thestylis* Doubleday, 1842[b]	Spotted Sawtooth		2			FS	Poly.	April - November	
		RIODINIDAE									
		Riodininae									
262	ABICHE	*Abisara chela chela* de Nicéville, 1886[c]	**Sikkim Spot Judy[3]**			**5**		FS	NA	April - November	707*
263	ABIFYL	*Abisara fylla* Doubleday, 1851	Dark Judy	21	5	26	12	FS	Mono.	January - December	
264	ABINEO	*Abisara neophron neophron* Hewitson, 1861	**Tailed Judy[3]**	1	2	**10**		FS	NA	April - December	0.801*
265[#]	DODADO	*Dodona adonira adonira* Hewitson, 1866	Striped Punch	2		1	1	FS	Mono.	April - November	
266[#]	DODEGE	*Dodona egeon egeon* Doubleday, 1851	Orange Punch	2		2		FS	Mono.	February - December	
267	DODOUI	*Dodona ouida ouida* Moore, 1865	Mixed Punch			4	1	FS	Mono.	April - December	
268	ZEMFLE	*Zemeros flegyas flegyas* Cramer, 1780	Punchinello	22	13	13	37	FS	Mono.	February - November	

Notes:

[I] **Sl. no.** in bold font: butterfly species protected under Indian Wildlife Protection Act 1972: Schedule I ($), Schedule II (#), Schedule IV (**+)**; CITES Appendix II (C);

[II] **6-letter code:** butterfly species code composed of first three letters each of genus name and species epithet;

[III] **Scientific name**: butterfly families (six) in uppercase, sub-family (23) in lowercase bold font; ecosystem exclusive butterfly species are marked by superscript (a: MOAS; b: FAS; c: LCAS; and d: Forests);

[IV] **Common name in bold font:** indicator butterfly species identified for specific/group of habitats (1: MOAS, 2: FAS, 3: LCAS, 4: Forest);

[V] **Ecosystems with abundance**: Butterfly abundance in large cardamom-based agroforestry system (LCAS), farm-based agroforestry system (FAS), mandarin orange-based agroforestry system (MOAS), Natural forest (Forest). The abundances of indicator butterfly species are depicted in bold, whereas that of ecosystem exclusive species are underlined;

[VI] **Habitat specialization (HS)**: forest specialist (FS), forest generalist (FG);

[VII] **Larval host specificity (LHS)**: monophagous (Mono.), polyphagous (Poly.), data deficient (NA); insectivorous (Insect.);

[VIII] **IndVal**: indicator value and significance (p): '**' 0.01, '*' 0.05; (alpha=0.05, func = "IndVal.g", duleg = FALSE, nperm = 999, At = 0.6, Bt=0.25, indvalcomp=TRUE).

Appendix H. Results of generalized linear mixed-effects models (GLMMs) for butterfly communities based on species per point along agroecosystem-forest gradient of Sikkim, Eastern Himalaya. The coefficients and associated standard errors of different predictors for best models (ΔAICc<4: Model 1 to 6), full model (ΔAICc=8.2), and null model (ΔAICc=95.99) are presented. AICc: Second-order Akaike Information Criterion; AICw: Akaike weights; df: Number of parameters; LogLik: log likelihood; ΔAICc: delta Second-order Akaike Information Criterion; MAP: Mean Annual Precipitation; MOAS: Mandarin orange-based agroforestry systems; LCAS: large cardamom-based agroforestry systems; and Forest: Natural forests.

	1	**2**	**3**	**4**	**5**	**6**	**Full**	**Null**
Intercept	1.19 *** (0.04)	1.19 *** (0.04)	1.19 *** (0.04)	1.19 *** (0.04)	1.11 *** (0.07)	1.19 *** (0.04)	1.10 *** (0.07)	1.17 *** (0.07)
SeasonPost Monsoon	0.19 *** (0.04)	0.19 *** (0.04)	0.19 *** (0.04)	0.19 *** (0.04)	0.19 *** (0.04)	0.19 *** (0.04)	0.19 *** (0.04)	
SeasonPre Monsoon	-0.02 (0.04)	-0.02 (0.04)	-0.02 (0.04)	-0.02 (0.04)	-0.02 (0.04)	-0.02 (0.04)	-0.02 (0.04)	
SeasonWinter	-0.18 *** (0.04)	-0.18 *** (0.04)	-0.18 *** (0.04)	-0.18 *** (0.04)	-0.18 *** (0.04)	-0.18 *** (0.04)	-0.18 *** (0.04)	
EcosystemForest					0.08 (0.11)		0.04 (0.14)	
EcosystemLCAS					0.11 0.10)		0.16 (0.12)	
EcosystemMOAS					014 (0.09)		0.17 (0.10)	
Elevation	-0.20 *** (0.04)	-0.20 *** (0.04)	-0.20 *** (0.04)	-0.24 *** (0.04)	-0.21 *** (0.04)	-0.20 *** (0.04)	-0.15 * (0.08)	
MAP							0.10 (0.10)	
Percentage canopy cover			0.01 (0.05)			0.01 (0.05)	-0.01 (0.07)	
Tree basal area		0.01(0.04)				0.01 (0.05)	0.01 (0.05)	
Tree density	-0.08 * (0.04)	-0.08 * (0.04)	-0.08 * (0.04)		-0.07 (0.04)	-0.08 * (0.04)	-0.03 (0.05)	
Tree species richness	0.12 *** (0.04)	0.13 ** (0.04)	0.13 ** (0.05)	0.10 ** (0.04)	0.11 * (0.04)	0.13 ** (0.05)	0.11* (0.05)	
df	9	10	10	8	12	11	15	1
LogLik	-3611.60	-3611.56	-3611.57	-3613.78	-3610.36	-3611.55	-3609.6	-3665.6
AICc	7241.29	7243.25	7243.26	7243.65	7244.89	7245.26	7249.49	7337.28
ΔAICc	0.00	1.96	1.97	2.36	3.60	3.97	8.2	95.99
AICw	0.42	0.16	0.16	0.13	0.07	0.06	-	-

*** p<0.001; ** p<0.01; * p<0.05.

Appendix I. Results of generalized mixed-effects models (GLMMs) for butterfly communities based on abundance per point along agroecosystem-forest gradient of Sikkim, Eastern Himalaya. The coefficients and associated standard errors of different predictors for best models (ΔAICc<4: Model 1 to 12), full model (ΔAICc=7.29), and null model (ΔAICc=112.74) are presented. AICc: Second-order Akaike Information Criterion; AICw: Akaike weights; df: Number of parameters; LogLik: log likelihood; ΔAIC: delta Second-order Akaike Information Criterion; MAP: Mean Annual Precipitation.

	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**Full**	**Null**
Intercept	1.47 *** (0.05)	1.46 *** (0.04)	1.47 *** (0.05)	1.38 *** (0.08)	1.46 *** (0.05)	1.46 *** (0.05)	1.46 *** (0.05)	1.46 *** (0.05)	1.46 *** (0.04)	1.35 *** (0.08)	1.46 *** (0.05)	1.46 *** (0.05)	1.37 *** (0.08)	1.42 *** (0.08)
SeasonPost Monsoon	0.21 *** (0.05)	0.21 *** (0.05)	0.21 *** (0.05)	0.21 *** (0.05)	0.21 *** (0.05)	0.21 *** (0.05)	0.22 *** (0.05)	0.22 *** (0.05)	0.21 *** (0.05)	0.21 *** (0.05)	0.21 *** (0.05)	0.21 *** (0.05)	0.21 *** (0.05)	
SeasonPre Monsoon	-0.07 (0.05)	-0.07 (0.05)	-0.07 (0.05)	-0.06 (0.05)	-0.07 (0.05)	-0.07 (0.05)	-0.07 (0.05)	-0.07 (0.05)	-0.07 (0.05)	-0.06 (0.05)	-0.07 (0.05)	-0.07 (0.05)	-0.07 (0.05)	
SeasonWinter	- 0.24*** (0.05)	-0.24*** (0.05)	-0.24*** (0.05)	-0.24*** (0.05)	-0.24*** (0.05)	-0.24*** (0.05)	- 0.24*** (0.05)	-0.24*** (0.05)	-0.24*** (0.05)	-0.24*** (0.05)	-0.24*** (0.05)	-0.24*** (0.05)	- 0.24*** (0.05)	
EcosystemForest				-0.04 (0.11)						0.06 (0.14)			0.05 (0.15)	
EcosystemLCAS				0.15 (0.11)						0.21 (0.12)			0.18 (0.12)	
EcosystemMOAS				0.20 * (0.10)						0.18 (0.10)			0.16 (0.10)	
Elevation	-0.21 *** (0.04)	-0.21 *** (0.04)	-0.21 *** (0.04)					-0.26 *** (0.04)	-0.21 *** (0.04)				-0.12 (0.08)	
MAP				0.27 *** (0.04)	0.26 *** (0.04)	0.22 *** (0.05)	0.26 *** (0.04)			0.29 *** (0.04)	0.23 *** (0.05)	0.27 *** (0.04)	0.15 (0.10)	
Percentage canopy cover			0.02 (0.05)		-0.08 (0.05)				-0.00 (0.05)	-0.07 (0.06)	-0.07 (0.05)		-0.03 (0.07)	
Tree basal area		0.04 (0.04)							0.04 (0.05)			-0.06 (0.05)	0.03 (0.05)	
Tree density	-0.10 * (0.04)	-0.10 * (0.04)	-0.10 * (0.04)			-0.07 (0.05)			-0.10 * (0.04)		-0.05 * (0.05)		-0.04 (0.05)	
Tree species richness	0.15 *** (0.04)	0.17 *** (0.05)	0.16 ** (0.05)	0.15 *** (0.04)	0.12 * (0.05)	0.19 *** (0.04)	0.18 *** (0.04)	0.12 ** (0.04)	0.17 *** (0.05)	0.12 ** (0.05)	0.14 ** (0.05)	0.14 ** (0.05)	0.14 ** (0.05)	
df	9	10	10	11	9	9	8	8	11	12	10	9		
LogLik	-4175.17	-4174.77	-4175.10	-4174.45	-4176.51	-4176.65	-4177.67	-4177.80	-4174.77	-4173.82	-4175.86	-4176.97	-4172.7	-4237.59
AICc	8368.45	8369.66	8370.32	8371.05	8371.12	8371.40	8371.43	8371.68	8371.69	8371.81	8371.85	8372.04	8375.74	8481.19
ΔAICc	0	1.21	1.87	2.60	2.66	2.95	2.98	3.23	3.24	3.36	3.40	3.59	7.29	112.74
AICw	0.26	0.14	0.10	0.07	0.07	0.06	0.06	0.05	0.05	0.05	0.05	0.04	-	-

*** p<0.001; ** p<0.01; * p<0.05.

Appendix J. Results of NMDS ordination based on Bray Curtis distance matrix for butterfly communities along agroecosystem- forest gradient of Sikkim Himalaya: total butterfly species (2D stress = 0.08), Forest specialist butterflies (2D stress = 0.09), Monophagous butterflies (2D stress = 0.10), and Protected butterflies (2D stress = 0.21).

	Total butterfly species			**Forest specialist butterflies**			**Monogamous butterflies**			**Protected butterflies**		
	NMDS1	**NMDS2**	**R^2**	**NMDS1**	**NMDS2**	**R^2**	**NMDS1**	**NMDS2**	**R^2**	**NMDS1**	**NMDS2**	**R^2**
Vectors												
Elevation	0.97906	-0.20358	**0.8904*****	0.98873	- 0.14971	**0.8851*****	0.99993	-0.01149	**0.9020*****	0.93157	0.36356	**0.7010*****
Mean annual temperature	-0.86423	0.50309	**0.7445*****	-0.85902	0.51194	**0.7968*****	-0.94445	0.32866	**0.7701*****	-0.99221	-0.12458	**0.7435*****
Mean annual precipitation	-0.82965	0.55828	**0.7211* ***	-0.98824	- 0.15289	**0.6366*****	-0.99260	-0.12144	**0.6337*****	-0.95496	-0.29674	**0.5229*****
Percentage canopy cover	0.35063	0.93651	**0.3767* ***	0.99014	0.14006	0.1510	0.70957	0.70463	0.1951	0.72022	0.69375	0.1049
Tree basal area	0.29834	0.95446	**0.3387***	0.59676	-0.80242	0.1550	0.97339	- 0.22917	0.1064	0.93378	0.35785	0.0841
Tree density	0.99769	-0.06794	**0.2952 ***	0.96165	0.27427	**0.2651***	0.96954	0.24494	0.2294	0.93404	0.35716	**0.3492***
Tree species richness	-0.30383	-0.95273	**0.3030***	-0.45318	-0.89142	0.1931	-0.41665	-0.90907	0.1821	-0.08031	-0.99677	0.1261
Ecosystem			**0.3239***			**0.2862***			**0.3254***			0.1914
MOAS	-0.4166	-0.1138		-0.3828	-0.1922		-0.3294	-0.2878		-0.2317	-0.2956	
FAS	-0.2621	-0.0509		-0.3116	0.1756		-0.3023	0.2098		-0.2033	0.1259	
LCAS	0.2838	-0.0662		0.3141	0.0434		0.2999	0.0429		0.2215	0.0263	
Forest	0.3950	0.2309		0.3803	-0.0267		0.3318	0.0351		0.2134	0.1434	

I used Monte-Carlo randomization test with 999 permutations to analyze significance (‘**’:p<0.01; ‘*’: p<0.05) of p values.

Appendix K. Literature referred for meta-analysis of butterfly species richness and Shannon-Wiener diversity of Agroecosystems and other human-modified ecosystems, and Forests including PAs in the biodiversity hotspots of Himalaya, Indo-Burma, Western Ghats and Sri Lanka, and elsewhere including tropical, temperate, and neotropics regionsof the World.

Region	Agroecosystems and other human-modified ecosystems	Forests including PAs
Himalaya	Khanal, 2013; Chettri *et al.*, 2018b; Tamang *et al.*, 2019; **Present study#**.	Uniyal and Mathur, 1998; Uniyal, 2004, 2007; Singh, 2009; Bhardwaj *et al.*, 2012; Acharya and Vijayan, 2011b, 2015; Sengupta *et al.*, 2014; Chettri, 2015; Singh, 2016, 2017b; Pandey *et al.*, 2017; Chettri *et al.*, 2018b; Shrestha *et al.*, 2018; Dewan *et al.*, 2019; Tamang *et al.*, 2019; **Present study#**.
Indo Burma	Vu, 2009, 2015.	Majumder *et al.*, 2012; Nidup *et al.*, 2014; Vu *et al.*, 2015; Singh *et al.*, 2015; Lodh and Agarwala, 2016.
Western Ghats and Sri Lanka	Kunte 1997; Kunte *et al.*, 1999; Shahabuddin and Ali, 2001; Padhye *et al.*, 2006; Dolia *et al.*, 2008.	Kunte 1997; Kunte *et al.*, 1999; Devy and Davidar, 2001; Shahabuddin and Ali, 2001; Padhye *et al.*, 2006; Dolia *et al.*, 2008; Mihindukulasooriya *et al.*, 2014.
Elsewhere	Mas and Dietsch, 2003; Horner-Devine *et al.*, 2003; Bobo *et al.*, 2006; Barlow *et al.*, 2007b; Kitahara *et al.*, 2008; Rákosy and Schmitt, 2011; Munyuli, 2012; Ekroos *et al.*, 2013; Francesconi *et al.*, 2013; Curtis *et al.*, 2015; Jew *et al.*, 2015; Myers *et al.*, 2015; Villemey *et al.*, 2015; Konvicka *et al.*, 2016; Basset *et al.*, 2017; Šálek *et al.*, 2018.	Horner-Devine *et al.*, 2003; Mas and Dietsch, 2003; Barlow *et al.*, 2007b; Uehara-Prado *et al.*, 2007; Kitahara *et al.*, 2008; Rundlof *et al.*, 2008; Munyuli, 2012; Robinson *et al.*, 2012; Ekroos *et al.*, 2013; Francesconi *et al.*, 2013; Loos *et al.*, 2014; Jew *et al.*, 2015; Konvicka *et al.*, 2016; Filgueiras *et al.*, 2016, 2019; Basset *et al.*, 2017; Puig-Montserrat *et al.*, 2017.

REFERENCES

Abs, M., Eckhoff, P., Fiebig, J., Frahnert, S., 2010a. The bird collections in the Museum für Naturkunde Berlin resulting from Ernst Schfer’s three expeditions to Tibet and Sikkim. Zoosyst. Evol. 86 (1), 49-80.

Abs, M., Eckhoff, P., Fiebig, J., Frahnert, S., 2010b. Erratum: The bird collections in the Museum für Naturkunde Berlin resulting from Ernst Schäfer's three expeditions to Tibet and Sikkim. Zoosyst. Evol. 86 (2), 351-354.

Acharya, B.K., 2008. Bird communities and their distribution pattern along the elevation gradient of Teesta Valley, Sikkim. Ph. D. Thesis. Bharathiar University, Coimbatore, India.

Acharya, B.K., Vijayan, L., Chettri, B., 2010. The bird community of Shingba Rhododendron wildlife sanctuary, Sikkim, Eastern Himalaya, India. Trop. Ecol. 51, 149-59.

Acharya, B.K., Sanders, N., Vijayan, L. Chettri, B., 2011a. Elevational gradients in bird diversity in the Eastern Himalaya: an evaluation of distribution patterns and their underlying mechanisms. PLoS One 6, e29097.

Acharya, B.K., Chettri, B., Vijayan, L., 2011b. Distribution pattern of trees along an elevation gradient of Eastern Himalaya, India. Acta Oecol. 37, 329-336.

Acharya, B.K., Sharma, G. 2012. The Traditional Dzumsa System and their role in Resource Management in Cultural Landscape in North Sikkim. In: Ramakrishnan, P.S., Saxena, K.G., Rao, K.S., Sharma, G. (Eds.) Cultural Landscapes: The Basis for Linking Biodiversity Conservation with the Sustainable Development. UNESCO and National Institute of Ecology, New Delhi, pp. 175-186.

Acharya, B.K., Sharma, G. 2013. Forests and biodiversity in Sikkim. In: Gazetteer of Sikkim. Home Department, Government of Sikkim, Gangtok, India, pp. 42-70.

Acharya, B.K., Vijayan, L., 2007. Range extension of Rusty-bellied Shortwing *Brachypteryx hyperythra* in Sikkim, India. Birding Asia *7*, 50-51.

Acharya, B.K., Vijayan, L., 2010. Status and distribution of endemic and threatened birds of the Eastern Himalaya in Sikkim, India. J. Threat. Taxa 2(2), 685-689.

Acharya, B.K., Vijayan, L., 2011a. Birds of Sikkim: an analysis of elevational distribution, endemism and threats. In: Arawatia, M.L. and Tambe, S. (Eds.) Biodiversity of Sikkim: Exploring and Conserving Global Hotspots. IPR Department, Government of Sikkim, Gangtok, India, pp. 255-281.

Acharya, B.K., Vijayan, L., 2011b. Butterflies of Sikkim with reference to elevational gradient in species, abundance, composition, similarity and range size distribution. In: Arawatia, M.L.,Tambe, S. (Eds.) Biodiversity of Sikkim: Exploring and Conserving Global Hotspots. IPR Department, Government of Sikkim, Gangtok, India, pp. 207-222.

Acharya, B.K., Vijayan, L., 2015. Butterfly diversity along the elevation gradient of Eastern Himalaya, India. Ecol. Res. 30, 909-919.

Acharya, B.K., Vijayan, L., 2017. Vertical stratification of birds in different vegetation types along an elevation gradient in the Eastern Himalaya, India. Ornithol. Sci. 16(2), 131-140.

Aguirre-Gutiérrez, J., WallisDeVries, M.F., Marshall, L., van't Zelfde, M., Villalobos-Arámbula, A.R., Boekelo, B., Bartholomeus, H., Franzén, M., Biesmeijer, J.C., 2017. Butterflies show different functional and species diversity in relationship to vegetation structure and land use. Glob. Ecol. Biogeogr. 26(10), 1126-1137

Ahmad, K., Yahya, H.S.A., 2010. Winter diversity of birds in Makaibari Tea Estate, Kurseong, Darjeeling, India. Ind. For. 136(1), 69-87.

Ahmed, A., Dey, M., 2014. A checklist of the winter bird community in different habitat types of Rosekandy Tea Estate of Assam, India. J. Threat. Taxa 6(2), 5478-5484.

Ali, A., Rizvi, P.Q., 2007. Developmental response of cabbage butterfly, *Pieris brassicae* L. (Lepidoptera: Pieridae) on different cole crops under laboratory and field condition. Asian J. Plant Sci., 6(8), 1241-1245.

Ali, S., 1962. The Birds of Sikkim. Oxford University Press, New Delhi, India.

Ali, S., 1977. Field guide to the birds of eastern Himalayas, Oxford University Press, Delhi.

Ali,S., Ripley, S.D., 2002. Handbook of the Birds of India and Pakistan (Vol 1-10). Oxford University Press, New Delhi, India.

Almeida, S.M., Silva, L.C., Cardoso, M.R., Cerqueira, P.V., Juen, L., Santos, M.P.D., 2016. The effects of oil palm plantations on the functional diversity of Amazonian birds. J. Trop. Ecol. 32, 510-525.

Alström, P., Rasmussen, P.C., Zhao, C., Xu, J., Dalvi, S., Cai, T., Guan, Y., Zhang, R., Kalyakin, M.V., Lei, F., Olsson, U., 2016. Integrative taxonomy of the Plain-backed Thrush (*Zoothera mollissima*) complex (Aves, Turdidae) reveals cryptic species, including a new species. Avian Res. 7(1), 1-39.

Anand, M.O., Krishnaswamy, J., Das, A., 2008. Proximity to forests drives bird conservation value of coffee plantations: Implications for certification. Ecol. Appl. 18, 1754-1763.

Anand, M.O., Krishnaswamy, J., Kumar, A., Bali, A., 2010. Sustaining biodiversity conservation in human-modified landscapes in the Western Ghats: Remnant forests matter. Biol. Conserv.143, 2363-2374.

Andersson, S., Nilsson, L.A., Groth, I., Bergstrom, G., 2002. Floral scents in butterfly-pollinated plants: possible convergence in chemical composition. Bot. J. Linn. Soc.140(2), 129-153.

Anonymous, 2010. The Wildlife (Protection) Act 1972(as amended up to 2006). Natraj Publishers, Dehra Dun.

Archaux, F., Lorel, C., Villemey, A., 2018. Landscape drivers of butterfly and burnet moth diversity in lowland rural areas. Landsc. Ecol. 33(10), 1725-1739.

Asefa, A., Davies, A.B., McKechnie, A.E., Kinahan, A.A., van Rensburg, B.J., 2017. Effects of anthropogenic disturbance on bird diversity in Ethiopian montane forests. Condor 119(3), 416-430.

Ash, D., Khanna, A., Bagchi, H., 2017. Northern Long-eared Owl *Asio otus* in East Sikkim. Indian BIRDS 13 (2), 44-45.

Azman, N.M., Latip, N.S.A., Sah, S.A.M., Akil, M.A.M.M., Shafie, N.J., Khairuddin, N.L., 2011. Avian diversity and feeding guilds in a secondary forest, an oil palm plantation and a paddy field in Riparian areas of the Kerian River Basin, Perak, Malaysia. Trop. Life Sci. Res. 22(2), 45-64.

Baiser, B., Olden, J.D., Record, S., Lockwood, J.L., McKinney, M.L. 2012. Pattern and process of biotic homogenization in the New Pangaea. Proc. R. Soc. B 279(1748), 4772-4777.

Balasubramanian, M.V., 1990. Studies on the ecology of butterfly pollination in South India. Part-II. Pollination of *Pavetta indica* Linn. (Rubiaceae). Ann. Entomol. 8(2), 71-78.

Balmford, B., Green, R.E., Onial, M., Phalan, B., Balmford, A., 2019. How imperfect can land sparing be before land sharing is more favourable for wild species?. J. Appl. Ecol. 56(1), 73-84.

Banerjee, A., Chakraborty, P., Bandopadhyay, R., 2019. Urgent conservation needs in the Sikkim Himalaya biodiversity hotspot. Biodiversity 20(2-3), 88-97.

Barbaro, L., Van Halder, I., 2009. Linking bird, carabid beetle and butterfly life-history traits to habitat fragmentation in mosaic landscapes. Ecography 32(2), 321-333.

Barbaro, L., Giffard, B., Charbonnier, Y., Van Halder, I., Brockerhoff, E.G., 2014. Bird functional diversity enhances insectivory at forest edges: a transcontinental experiment. Divers. Distrib. 20(2), 149-159.

Barbaro, L., Rusch, A., Muiruri, E.W., Gravellier, B., Thiery D., Castagneyrol B., 2017. Avian pest control in vineyards is driven by interactions between bird functional diversity and landscape heterogeneity. J. Appl. Ecol. 54, 500-508.

Barbaro, L., Allan, E., Ampoorter, E., Castagneyrol, B., Charbonnier, Y., De Wandeler, H., Kerbiriou, C., Milligan, H.T., Vialatte, A., Carnol, M., Deconchat, M., 2019. Biotic predictors complement models of bat and bird responses to climate and tree diversity in European forests. Proc. R. Soc. B286(1894), 20182193. DOI: 10.1098/rspb.2018.2193

Barnagaud, J.Y., Barbaro, L., Papaix, J., Deconchat, M., Brockerhoff, E.G., 2014. Habitat filtering by landscape and local forest composition in native and exotic New Zealand birds. Ecology 95(1), 78-87.

Barlow, J., Mestre, L.A., Gardner, T.A., Peres, C.A., 2007a. The value of primary, secondary and plantation forests for Amazonian birds. Biol. Conserv. 136(2), 212-231.

Barlow, J., Overal, W.L., Araujo, I.S., Gardner, T.A., Peres, C.A., 2007b. The value of primary, secondary and plantation forests for fruit-feeding butterflies in the Brazilian Amazon. J. Appl. Ecol. 44, 1001-1012.

Barlow, J., França, F., Gardner, T.A., Hicks, C.C., Lennox, G.D., Berenguer, E., Castello, L., Economo, E.P., Ferreira, J., Guénard, B., Gontijo Leal, C., Isaac, V., Lees, A.C., Parr, C.L., Wilson, S.K., Young, P.J., & Graham, N.A.J. (2018). The future of hyperdiverse tropical ecosystems. Nature, 559(7715), 517-526.

Bartomeus, I., Cariveau, D.P., Harrison, T., Winfree, R., 2018. On the inconsistency of pollinator species traits for predicting either response to land-use change or functional contribution. Oikos 127, 306-315.

Barua, M., Sharma, P., 1999. Birds of Kaziranga national park, India. Forktail 15, 47-60.

Barua, M., Sharma, P., 2005. The birds of Nameri National Park, Assam, India. Forktail, 21, 15-26.

Baselga, A., 2010. Partitioning the turnover and nestedness components of beta diversity. Glob. Ecol. Biogeogr. 19, 134-143.

Baselga, A., 2013a. Separating the two components of abundance-based dissimilarity: balanced changes in abundance vs. abundance gradients. Methods Ecol. Evol. l4, 552-557.

Baselga, A., 2013b. Multiple site dissimilarity quantifies compositional heterogeneity among several sites, while average pairwise dissimilarity may be misleading. Ecography 36, 124-128.

Baselga, A., 2017. Partitioning abundance-based multiple-site dissimilarity into components: balanced variation in abundance and abundance gradients. Methods. Ecol. Evol. 8, 799-808.

Basnet, T.B., Rokaya, M.B., Bhattarai, B.P., Münzbergová, Z., 2016. Heterogeneous landscapes on steep slopes at low altitudes as hotspots of bird diversity in a Hilly Region of Nepal in the Central Himalayas. PLoS One 11(3), e0150498.

Basset, Y., Barrios, H., Ramirez, J.A., Lopez, Y., Coronado, J., Perez, F., Arizala, S., Bobadilla, R, Leponce, M., 2017. Contrasting the distribution of butterflies and termites in plantations and tropical forests. Biodivers. Conserv. 26, 151-176.

Batáry, P., Báldi, A., Kleijn, D., Tscharntke, T., 2011. Landscape-moderated biodiversity effects of agri-environmental management: a meta-analysis. Proc. R. Soc. B. 278(1713), 1894-1902.

Bawa, K.S., Sengupta, A., Chavan, V., Chellam, R., Ganesan, R., Krishnaswamy, J., Mathur, V.B., Nawn, N., Olsson, S.B., Pandit, N., Quader, S., 2021. Securing biodiversity, securing our future: A national Mission on biodiversity and human well-being for India. Biol. Conserv.253, 108867.

Beecher, N.A., Johnson, R.J., Brandle, J.R., Case, R.M., Young, L.J., 2002. Agroecology of birds in organic and nonorganic farmland. Conserv. Biol. 16(6), 1620-1631.

Behera, M.D., Kushwaha, S.P.S., Roy, P.S., 2002. High plant endemism in an Indian hotspot - eastern Himalaya. Biodivers Conserv 11, 669-682.

Bełcik, M., Lenda, M., Amano, T., Skórka, P., 2020. Different response of the taxonomic, phylogenetic and functional diversity of birds to forest fragmentation. Sci. Rep. 10, 20320. DOI: https://doi.org/10.1038/s41598-020-76917-2

Beletsky, L., 2006. Birds of the world. The John Hopkins University Press, Baltimore.

Berg, A., 1997. Diversity and abundance of birds in relation to forest fragmentation, habitat quality and heterogeneity. Bird Study 44, 355-366.

Beukema, H., Danielsen, F., Vincent, G., Hardiwinoto, S., van Andel, J., 2007. Plant and bird diversity in rubber agroforests in the lowlands of Sumatra, Indonesia. Agrofor. Syst. 70(3), 217-242.

Bhagwat, S.A., Kushalappa, C.G., Williams, P.H., Brown, N.D., 2005a. The role of informal protected areas in maintaining biodiversity in the Western Ghats of India. Ecol. Soc. 10(1), 8. https://www.ecologyandsociety.org/vol10/iss1/art8/

Bhagwat, S.A., Kushalappa, C.G., Williams, P.H., Brown, N.D., 2005b. A landscape approach to biodiversity conservation of sacred groves in the Western Ghats of India. Conserv. Biol. 19(6), 1853-1862.

Bhagwat, S.A., Willis, K.J., Birks, H.J.B. Whittaker, R.J., 2008. Agroforestry: a refuge for tropical biodiversity? Trends Ecol. Evol.23, 261-267.

Bhardwaj, M., Uniyal, V.P., Sanyal, A.K., Singh, A.P., 2012. Butterfly communities along an elevational gradient in the Tons valley, Western Himalayas: implications of rapid assessment for insect conservation. J. Asia-Pac. Entomol. 15, 207-217.

Bhutia, D.T., 2015. Sikkim Organic Mission "Journey of a fully organic state". Department of Food Security & Agriculture Development and Horticulture and Cash Crops Development, Government of Sikkim, Gangtok.

Bibby, C.J., Burgess, N.D., Hill, D.A., Hillis, D.M. Mustoe, S., 2000. *Bird census techniques.* 2nd edition, Elsevier.

Bihn, J.H., Gebauer, G., Brandl, R., 2010. Loss of functional diversity of ant assemblages in secondary tropical forests. Ecology, 91(3), pp.782-792.

Billeter, R., Liira, J., Bailey, D., Bugter, R., Arens, P., Augenstein, I., Aviron, S., Baudry, J., Bukacek, R., Burel, F., Cerny, M., 2008. Indicators for biodiversity in agricultural landscapes: a pan-European study. J. Appl. Ecol. 45(1), 141-150.

BirdLife International, 2020a. IUCN Red List for birds. Downloaded from https://www.birdlife.org on 20/06/2020.

BirdLife International, 2020b. Endemic Bird Areas factsheet: Eastern Himalayas. Downloaded from https://www.birdlife.org on 20/06/2020.

Bobo, K.S., Waltert, M., Fermon, H., Njokagbor, J., Mühlenberg, M., 2006. From forest to farmland: butterfly diversity and habitat associations along a gradient of forest conversion in Southwestern Cameroon. J. Insect Conserv. 10, 29-42.

Boesing, A.L., Nichols, E., Metzger, J.P., 2017. Effects of landscape structure on avian-mediated insect pest control services: a review. Landsc. Ecol. 32(5), 931-944.

Böhm, S.M., Wells, K., Kalko, E.K., 2011. Top-down control of herbivory by birds and bats in the canopy of temperate broad-leaved oaks (*Quercus robur*). PLoS One 6(4), e17857.

Bommarco, R., Marini, L., Vaissièrè, B.E., 2012. Insect pollination enhances seed yield, quality, and market value in oilseed rape. Oecologia 169(4), 1025-1032.

Borges, R.M., Gowda, V., Zacharias, M., 2003. Butterfly pollination and high-contrast visual signals in a low-density distylous plant. Oecologia 136(4), 571-573.

Börschig, C., Klein, A.M., von Wehrden, H., Krauss, J., 2013. Traits of butterfly communities change from specialist to generalist characteristics with increasing land-use intensity. Basic Appl. Ecol. 14(7), 547-554.

Bovo, A.A., Ferraz, K.M., Magioli, M., Alexandrino, E.R., Hasui, É., Ribeiro, M.C., Tobias, J.A., 2018. Habitat fragmentation narrows the distribution of avian functional traits associated with seed dispersal in tropical forest. Perspect. Ecol. Conserv. 16(2), 90-96.

Bregman, T.P., Şekercioğlu, C.H., Tobias, J.A., 2014. Global patterns and predictors of bird species responses to forest fragmentation: implications for ecosystem function and conservation. Biol. Conserv. 169, 372-83.

Bregman, T.P., Lees, A.C., MacGregor, H.E., Darski, B., de Moura, N.G., Aleixo, A., Barlow, J., Tobias, J.A., 2016. Using avian functional traits to assess the impact of land-cover change on ecosystem processes linked to resilience in tropical forests. Proc. R. Soc. B283(1844), 20161289.

Brereton, T.M., Roy, D.B., Middlebrook, I., Botham, M.S., Warren, M.S., 2011. The development of butterfly indicators in the United Kingdom and assessments in 2010. J. Insect Conserv. 15, 139-151.

Bubova, T., Vrabec, V., Kulma, M., Nowicki, P., 2015. Land management impacts on European butterflies of conservation concern: a review. J. Insect. Conserv. 19, 805-821.

Buechley, E.R., Şekercioğlu, C.H., Atickem, A., Gebremichael, G., Ndungu, J.K., Mahamued, B.A., Beyene, T., Mekonnen, T., Lens, L., 2015. Importance of Ethiopian shade coffee farms for forest bird conservation. Biol. Conserv.188, 50-60.

Bulger, G.F., 1869. List of birds obtained in Sikkim, Eastern Himalayas between March and July 1867. Ibis 2, 154-170.

Burnham, K.P., Anderson, D.R., 2002. Model Selection and Multimodel Inference: A Practical Information-Theoretic Approach. 2nd edition, Springer-Verlag, New York.

Cadotte, M.W., Carscadden, K., Mirotchnick, N., 2011. Beyond species: functional diversity and the maintenance of ecological processes and services. J. Appl. Ecol. 48(5), 1079-1087.

Cannon, P.G., Gilroy, J.J., Tobias, J.A., Anderson, A., Haugaasen, T., Edwards, D.P., 2019. Land-sparing agriculture sustains higher levels of avian functional diversity than land sharing. Glob.Change Biol. 25(5), 1576-1590.

Cardinale, B.J., Duffy, J.E., Gonzalez, A., Hooper, D.U., Perrings, C., Venail, P., Narwani, A., Mace, G.M., Tilman, D., Wardle, D.A., Kinzig, A.P., 2012. Biodiversity loss and its impact on humanity. Nature 486, 59-67.

Carrara, E., Arroyo-Rodríguez, V., Vega-Rivera, J.H., Schondube, J.E., de Freitas, S.M., Fahrig, L., 2015. Impact of landscape composition and configuration on forest specialist and generalist bird species in the fragmented Lacandona rainforest, Mexico. Biol. Conserv. 184, 117-126.

Carré, G., Roche, P., Chifflet, R., Morison, N., Bommarco, R., Harrison-Cripps, J., Krewenka, K., Potts, S.G., Roberts, S.P.M., Rodet, G., Settele, J., Steffan-Dewenter, I., Szentgyörgyi, H., Tscheulin, T., Westphal, C., Woyciechowski, M., Vaissière, B.E., 2009. Landscape context and habitat type as drivers of bee diversity in European annual crops. Agric. Ecosyst. Environ. 133, 40-47.

Casas-Crivillé, A., Valera, F., 2005. The European bee-eater (*Merops apiaster*) as an ecosystem engineer in arid environments. J. Arid Environ. 60(2), 227-238.

Chandra, K., Gupta, D., Gopi, K.C., Tripathy, B., Kumar, V., (Eds.), 2018. Faunal Diversity of Indian Himalaya. Zoological Survey of India, Kolkata, pp. 1-872.

Chandran, K., Vishnudas, C.K., 2018. A comparative study of mixed-species bird flocks in shaded coffee plantation and natural forest in Wayanad, Kerala. Indian BIRDS 14(4), 97-102.

Chang, C.H., Karanth, K.K., Robbins, P., 2018. Birds and beans: Comparing avian richness and endemism in arabica and robusta agroforests in India’s Western Ghats. Sci. Rep. 8(1), 1-9.

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

Chapin, F.S., Carpenter, S.R., Kofinas, G.P., Folke, C., Abel, N., Clark, W.C., Olsson, P., Stafford Smith, D.M, Walker, B., Young, O.R., Berkes, F., Biggs, R., Morgan Grove, J., Naylor, R.L. Pinkerton, E., Steffen, W., Swanson F.J., 2010. Ecosystem stewardship: sustainability strategies for a rapidly changing planet. Trends Ecol. Evol. 25, 241-249.

Chappell, M.J., LaValle, L.A., 2011. Food security and biodiversity: can I have both? An agroecological analysis. Agric. Human Values28(1), 3-26.

Chazdon, R.L., Harvey, C.A., Komar, O., Griffith, D.M., Ferguson, B.G., Martínez-Ramos, M., Morales, H., Nigh, R., Soto-Pinto, L., Van Breugel, M., Philpott, S.M., 2009. Beyond reserves: A research agenda for conserving biodiversity in human-modified tropical landscapes. Biotropica 41(2), 142-153.

Chen, C., Holyoak, M., Wang, Y., Si, X., Ding, P., 2019. Spatiotemporal distribution of seasonal bird assemblages on land-bridge islands: linking dynamic and static views of metacommunities. Avian Res. 10(1), 25. DOI: https://doi.org/10.1186/s40657-019-0164-7

Chettri, A., Sharma, K., Dewan, S., Acharya, B.K., 2018a. Bird diversity of tea plantations in Darjeeling Hills, Eastern Himalaya, India. Biodiversitas 19(3), 1066-1073.

Chettri, B., Bhupathy, S., Acharya, B.K., 2010a. Distribution pattern of reptiles along an Eastern Himalayan elevation gradient, India. Acta Oecol. 36, 16-22.

Chettri, B., Acharya, B.K., 2020. Distribution of amphibians along an elevation gradient in the Eastern Himalaya, India. Basic Appl. Ecol. 47, 57-70.

Chettri, N., 2010. Cross-taxon congruence in a tracking corridor of Sikkim Himalaya: Surrogate analysis for conservation planning. J. Nat. Conserv. 18, 75-88.

Chettri, N., 2015. Distribution of butterflies along a trekking corridor in the Khangchendzonga Biosphere Reserve, Sikkim, Eastern Himalayas. Conserv. Sci. 3, 1-10.

Chettri, N., Sharma, E., Deb, D.C., 2001. Bird community structure along a trekking corridor of Sikkim Himalaya: a conservation perspective. Biol. Conserv. 102, 1-16.

Chettri, N., Deb, D.C., Sharma, E., Jackson, R., 2005. The relationship between bird communities and habitat: A study along a trekking corridor of the Sikkim Himalaya. Mt. Res. Dev.25, 235-244.

Chettri, N., Sharma, E., Shakya, B., Thapa, R., Bajracharya, B., Uddin, K., Choudhury, D., Oli, K.P., 2010b. Biodiversity in the Eastern Himalayas: Status, trends and vulnerability to climate change. Climate change impact and vulnerability in the Eastern Himalayas - Technical report 2. Kathmandu: ICIMOD.

Chettri, P.K., Ethenpa, K.T., 2020. A Terek Sandpiper *Xenus cinereus* from the Sikkim Himalaya. Indian BIRDS 16(2), 63-64.

Chettri, P.K., Sharma, K., Dewan, S., Acharya, B.K., 2018b. Butterfly diversity in human-modified ecosystems of southern Sikkim, the eastern Himalaya, India. J. Threat. Taxa 10(5), 11551-11565.

Chettri, P.K., Lepcha, T.N., Chhetri, B., Gurung, K., 2019. First record of Laughing Dove (*Spilopelia senegalensis* Linnaeus, 1766) (Columbiformes) in Sikkim, Eastern Himalaya. Bionotes 21 (4), 106-108.

Chiawo, D.O., Kombe, W.N., Craig, A.J., 2018. Bird responses to land use change: guild diversity in a Kenyan coastal forest and adjoining habitats. Emu, 118(3), 281-292.

Choudhury, A., 2006. Birds of Dibru-Saikhowa National Park and Biosphere Reserve, Assam, India. Indian BIRDS 2(4), 95-106.

Choudhury, A., 2016. Saker Falcon *Falco cherrug* in northern Sikkim, India. Indian BIRDS 11(1), 19.

Clavero, M., Brotons, L., 2010. Functional homogenization of bird communities along habitat gradients: accounting for niche multidimensionality. Glob. Ecol. Biogeogr. 19, 684-696.

Cleary, D.F., Genner, M.J., Koh, L.P., Boyle, T.J., Setyawati, T., de Jong, R., Menken, S.B., 2009. Butterfly species and traits associated with selectively logged forest in Borneo. Basic Appl. Ecol. 10(3), 237-245.

Clough, Y., Putra, D.D., Pitopang, R., Tscharntke, T., 2009. Local and landscape factors determine functional bird diversity in Indonesian cacao agroforestry. Biol. Conserv. 142, 1032-1041.

Coad, L., Watson, J.E.M., Geldmann, J., Burgess, N.D., Leverington, F., Hockings, M., Knights, K., Di Marco, M., 2019. Widespread shortfalls in protected area resourcing undermine efforts to conserve biodiversity. Front. Ecol. Environ. 17(5), 259-264.

Cockle, K. L., Martin, K., Wesołowski, T., 2011. Woodpeckers, decay, and the future of cavity-nesting vertebrate communities worldwide. Front Ecol Environ. 9(7), 377-382.

Correia, I., do Nascimento, E.R. and Gouveia, S.F., 2020. Effects of climate and land-use gradients on avian phylogenetic and functional diversity in a tropical dry forest. J. Arid Environ. 173, 104024.

Corlett, R.T., 1998. Frugivory and seed dispersal by vertebrates in the Oriental (Indomalayan) Region. Biol. Rev. 73(4), 413-448.

Corlett, R.T., 2004. Flower visitors and pollination in the Oriental (Indomalayan) Region. Biol. Rev. 79, 497-532.

Cottee-Jones, H.E.W., Matthews, T.J., Bregman, T.P., Barua, M., Tamuly, J., Whittaker, R.J., 2015. Are protected areas required to maintain functional diversity in human-modified landscapes? PLoS One 10(5), e0123952.

Crowder, D.W., Northfield, T.D., Strand, M.R., Snyder, W.E., 2010. Organic agriculture promotes evenness and natural pest control. Nature466(7302), 109-112.

Curtis, R.J., Brereton, T.M., Dennis, R.L., Carbone, C., Isaac, N.J., 2015. Butterfly abundance is determined by food availability and is mediated by species traits. J. Appl. Ecol. 52(6), 1676-1684.

Cussera, S., Neffb, J.L., Jhaa, S., 2016. Natural land cover drives pollinator abundance and richness, leading to reductions in pollen limitation in cotton agroecosystems. Agric. Ecosyst. Environ. 226, 33-42.

Cuthbert, R.J., Taggart, M.A., Saini, M., Sharma, A., Das, A., Kulkarni, M.D., Deori, P., Ranade, S., Shringarpure, R.N., Galligan, T.H., Green, R.E., 2016. Continuing mortality of vultures in India associated with illegal veterinary use of diclofenac and a potential threat from nimesulide. Oryx, 50(1), 104-112.

Dahal, B.R., McAlpine, C.A., Maron, M., 2014. Bird conservation values of off-reserve forests in lowland Nepal. For. Ecol. Manag. 323, 28-38.

Daily, G.C., Ehrlich, P.R., Sánchez-Azofeifa, G.A., 2001. Countryside biogeography: use of human-dominated habitats by the avifauna of southern Costa Rica. Ecol. Applic. 11(1), 1-13.

Dainese, M., Isaac, N.J.B., Powney, G.D., Bommarco, R., Öckinger, E., Kuussaari, M., Benton, T.G., Gabriel, T., Hodgson, J. A., Kunin, W.E., Lindborg, R., Sait, S.M., Marini, L., 2017. Landscape simplification weakens the association between terrestrial producer and consumer diversity in Europe. Glob. Change Biol. 23, 3040-3051.

Daniels, R.R., Joshi, N.V., Gadgil, M., 1990. Changes in the bird fauna of Uttara Kannada, India, in relation to changes in land use over the past century. Biol. Conserv. 52(1), 37-48.

Daniels, R.J.R., Joshi, N.V., Gadgil, M., 1992. On the relationship between bird and woody plant diversity in the Uttara Kannada district of south India. Proc. Natl. Acad. Sci. U.S.A. 89, 5311-5315.

Darras, K., Rahman, D., Waluyo Sugito, Y.M., Prawiradilaga, D., Rozali, A., Fitriawan, I., Tscharntke, T., 2018. Birds of primary and secondary forest and shrub habitats in the peat swamp of Berbak National Park, Sumatra. F1000Res. 7, 229. DOI: 10.12688/f1000research.13996.2

Davidar, P., 1987. Fruit structure in two neotropical mistletoes and its consequences for seed dispersal. Biotropica19(2), 137-139.

Davis, J.D., Debinski, D.M., Danielson, B.J., 2007. Local and landscape effects on the butterfly community in fragmented Midwest USA prairie habitats. Landsc. Ecol. 22, 1341–1354.

De Cáceres, M., Legendre, P., Moretti, M., 2010. Improving indicator species analysis by combining groups of sites. Oikos 119(10), 1674-1684.

De Cáceres, M., Legendre, P., Wiser, S.K. and Brotons, L., 2012. Using species combinations in indicator analyses. Methods Ecol. Evol. 3(6), 973-982.

Dehling, D. M., Töpfer, T., Schaefer, H. M., Jordano, P., Böhning-Gaese, K., Schleuning, M., 2014. Functional relationships beyond species richness patterns: trait matching in plant-bird mutualisms across scales. Glob. Ecol. Biogeogr. 23(10), 1085-1093.

del Hoyo, J., Elliott, A., Sargatal, J., Christie, D.A., de Juana, E., (Eds.), 2020. Handbook of the Birds of the World Alive. Lynx Editions, Barcelona. https://birdsoftheworld.org/hbw/species/whnyuh1/1.0

Dennis, J.A., 2010. A resource-based habitat view for conservation: Butterflies in the British landscape. Wiley-Blackwell.

DeVault, T.L., Beasley, J.C., Olson, Z.H., Moleón, M., Carrete, M., Margalida, A., Sánchez-Zapata, J.A., 2016. Ecosystem services provided by avian scavengers. In: Şekercioğlu, Ç.H. Wenny, D.G., Whelan, C.J., (Eds.), Why Birds Matter: Avian Ecological Function and Ecosystem Services. University of Chicago Press, pp. 235-270.

Devictor, V., Julliard, R., Clavel, J., Jiguet, F., Lee, A., Couvet, D., 2008. Functional biotic homogenization of bird communities in disturbed landscapes. Glob. Ecol. Biogeogr. 17(2), 252-261.

Devictor, V., Mouillot, D., Meynard, C., Jiguet, F., Thuiller, W. Mouquet, N., 2010. Spatial mismatch and congruence between taxonomic, phylogenetic and functional diversity: the need for integrative conservation strategies in a changing world. Ecol. Lett. 13, 1030-1040.

Devy, M.S., Davidar, P., 2001. Response of wet forest butterflies to selective logging in Kalakad–Mundanthurai Tiger Reserve: Implications for conservation. Curr. Sci. 80(3), 400-405.

Dewan, S., Acharya, B.K. Ghatani, S., 2018. A new record of the lesser-known butterfly Small Woodbrown *Lethe nicetella*de Nicéville, 1887 (Lepidoptera: Nymphalidae: Satyrinae) from Kangchendzonga National Park, Sikkim, India. J. Threat. Taxa 10(6), 11775-11779.

Dewan, S., Chettri, I.K., Sharma, K., Acharya, B.K., 2019. Kitam Bird Sanctuary, the only low elevation protected area of Sikkim: A conservation hotspot for butterflies in the Eastern Himalaya. J. Asia-Pac. Entomol. 22(2), 575-583.

Dewan, S., Acharya, B. K., Vetaas, O. R., Ghatani, S. 2021. Do sub-groups of butterflies display different elevational distribution patterns in the Eastern Himalaya, India?. Front. Biogeogr. 13 (3), e49643.

Dey, P., Chatterjee, P., Mondal, K., Ghosh, D., Tripathy, B., Chandra, K., 2017. An annotated checklist of avifauna from Neora Valley National Park, Central Himalayas, India. J. Zoo. Stud. 4(3), 15-23.

Dhindsa, M.S., Saini, H.K., 1994. Agricultural ornithology: an Indian perspective. J. Biosci. 19(4), 391-402.

Díaz, I.A., Armesto, J.J., Reid, S., Sieving, K.E., Willson, M.F., 2005. Linking forest structure and composition: avian diversity in successional forests of Chiloé Island, Chile. Biol. Conserv. 123(1), 91-101.

Diaz, S., Cabido, M., 2001. Vive la difference: plant functional diversity matters to ecosystem processes. Trends Ecol. Evol. 16, 646-655.

Diaz, S., Lavorel, S., de Bello, F., Quetier, F., Grigulis, K., Robson, T.M., 2007. Incorporating plant functional diversity effects in ecosystem service assessments. Proc. Natl. Acad. Sci. U.S.A. 104(52), 20684-20689.

Diffendorfer, J.E., Loomis, J.B., Ries, L., Oberhauser, K., Lopez-Hoffman, L., Semmens, D., Semmens, B., Butterfield, B., Bagstad, K., Goldstein, J., Wiederholt, R., 2014. National Valuation of Monarch Butterflies Indicates an Untapped Potential for Incentive-Based Conservation. Conserv. Lett. 7(3), 253-262.

Ding, Z., Feeley, K.J., Wang, Y., Pakeman, R.J., Ding, P., 2013. Patterns of bird functional diversity on land-bridge island fragments. J. Anim. Ecol. 82(4), 781-790.

Ding, Z., Liang, J., Hu, Y., Zhou, Z., Sun, H., Liu, L., Liu, H., Hu, H., Si, X., 2019. Different responses of avian feeding guilds to spatial and environmental factors across an elevation gradient in the central Himalaya. Ecol. Evol. 9(7), 4116-4128.

Dobrovolski, R., Melo, A.S., Cassemiro, F.A.S., Diniz-Filho, J.A.F., 2012. Climatic history and dispersal ability explain the relative importance of turnover and nestedness components of beta diversity. Glob. Ecol. Biogeogr. 21, 191-197.

Doledec, S., Chessel, D., Braak, C.J.F., Champely, S., 1996. Matching species traits to environmental variables: a new three-table ordination method. Environ. Ecol. Stat. 3, 143-166.

Dolia, J., Devy, M.S., Aravind, N.A., Kumar, A., 2008. Adult butterfly communities in coffee plantations around a protected area in the Western Ghats, India. Anim. Conserv. 11, 26-34.

Dong, K., Hao, G., Yang, N., Zhang, J.L., Ding, X.F., Ren, H.Q., Shen, J.F., Wang, J.L., Jiang, L., Zhao, N.X., Gao, Y.B., 2019. Community assembly mechanisms and succession processes significantly differ among treatments during the restoration of *Stipa grandis–Leymus chinensis* communities. Sci. Rep. 9(1), 1-9.

Doxa, A., Bas, Y., Paracchini, M.L., Pointereau, P., Terres, J.M., Jiguet, F., 2010. Low-intensity agriculture increases farmland bird abundances in France. J. Appl. Ecol. 47(6), 1348-1356.

Doxa, A., Paracchini, M.L., Pointereau, P., Devictor, V., Jiguet, F., 2012. Preventing biotic homogenization of farmland bird communities: the role of High Nature Value farmland. Agric. Ecosyst. Environ. 148, 83-88.

Dray, S., Dufour, A.B., 2007. The ade4 package: implementing the duality diagram for ecologists. J. Stat. Softw. 22, 1-20.

Dray, S., Chessel, D., Thioulouse, J., 2003. Co-inertia analysis and the linking of ecological data tables. Ecology 84, 3078-3089.

Dray, S., Choler, P., Dolédec, S., Peres-Neto, P.R., Thuiller, W., Pavoine, S., ter Braak, C.J.F., 2014. Combining the fourth-corner and the RLQ methods for assessing trait responses to environmental variation. Ecology 95, 14-21.

Dufrêne, M., Legendre, P., 1997. Species assemblages and indicator species: The need for a flexible asymmetrical approach. Ecol. Monogr. 67, 345-366.

Dunning Jr, J.B., 2008. CRC handbook of avian body masses. 2nd edition. CRC Press.

Eames, J.C., Steinheimer, F.D., Bansok, R., 2002. A collection of birds from the Cardamom Mountains, Cambodia, including a new subspecies of *Arborophila cambodiana*. Forktail18, 67-86.

Echeverri, A., Frishkoff, L.O., Gomez, J.P., Zook, J.R., Juárez, P., Naidoo, R., Chan, K.M., Karp, D.S., 2019. Precipitation and tree cover gradients structure avian alpha diversity in North-western Costa Rica. Divers. Distrib. 25(8), 1222-1233.

Edwards, F.A., Edwards, D.P., Hamer, K.C., Davies, R.G., 2013. Impacts of logging and conversion of rainforest to oil palm on the functional diversity of birds in Sundaland. Ibis 155, 313-326.

Ehlers Smith, D.A., Si, X., Ehlers Smith, Y.C., Kalle, R., Ramesh, T., Downs, C.T., 2018. Patterns of avian diversity across a decreasing patch-size gradient in a critically endangered sub-tropical forest system. J. Biogeogr. 45, 2118-2132.

Ehlers Smith, Y.C., Ehlers Smith, D.A., Seymour, C.L., Thebault, E., van Veen, F.F., 2015. Response of avian diversity to habitat modification can be predicted from life-history traits and ecological attributes. Landsc. Ecol. 30, 1225-1239.

Ekroos, J., Heliola, J., Kuussaari, M., 2010. Homogenization of lepidopteran communities in intensively cultivated agricultural landscapes. J. Appl. Ecol. 47, 459-467.

Ekroos, J., Kuussaari, M., Tiainen, J., Heliölä, J., Seimola, T., Helenius, J., 2013. Correlations in species richness between taxa depend on habitat, scale and landscape context. Ecol. Indic. 34, 528-535.

Ellis, J.C., 2005. Marine birds on land: a review of plant biomass, species richness, and community composition in seabird colonies. Plant Ecol. 181(2), 227-241.

Elsen, P.R., Kalyanaraman, R., Ramesh, K., Wilcove, D.S., 2016. The importance of agricultural lands for Himalayan birds in winter. Conserv. Biol. 31(2), 416-426.

Elsen, P.R., Ramesh, K., Wilcove, D.S., 2018. Conserving Himalayan birds in highly seasonal forested and agricultural landscapes. Conserv. Biol. 32(6), 1313-1324.

Fahrig, L., Baudry, J., Brotons, L., Burel, F.G., Crist, T.O., Fuller, R.J., Sirami, C., Siriwardena, G.M., Martin, J.L., 2011. Functional landscape heterogeneity and animal biodiversity in agricultural landscapes. Ecol. Lett. 14, 101-12.

Faria, D., Laps, R.R., Baumgarten, J., Cetra, M., 2006. Bat and bird assemblages from forests and shade cacao plantations in two contrasting landscapes in the Atlantic Forest of southern Bahia, Brazil. Biodivers. Conserv. 15(2), 587-612.

Feber, R.E., Firbank, L.G., Johnson, P.J., Macdonald, D.W., 1997. The effects of organic farming on pest and non-pest butterfly abundance. Agric. Ecosyst. Environ. 64(2), 133-139.

Fick, S. E., Hijmans, R.J., 2017. WorldClim 2: new 1-km spatial resolution climate surfaces for global land areas. Int. J. Climatol. 37(12), 4302-4315.

Filgueiras, B.K., Melo, D.H., Leal, I.R., Tabarelli, M., Freitas, A.V.L., Iannuzzi, L., 2016. Fruit-feeding butterflies in edge-dominated habitats: community structure, species persistence and cascade effect. J. Insect Conserv. 20(3), 539-548.

Filgueiras, B.K., Melo, D.H., Andersen, A.N., Tabarelli, M., Leal, I.R., 2019. Cross-taxon congruence in insect responses to fragmentation of Brazilian Atlantic forest. Ecol. Indic. 98, 523-530.

Fischer, C., Flohre, A., Clement, L.W., Batáry, P., Weisser, W.W., Tscharntke, T.,Thies, C., 2011. Mixed effects of landscape structure and farming practice on bird diversity. Agric. Ecosyst. Environ.141*(*1-2*)*, 119-125.

Fischer, J., Lindenmayer, D.B., Blomberg, S.P., Montague-Drake, R., Felton, A., Stein, J.A., 2007. Functional richness and relative resilience of bird communities in regions with different land use intensities. Ecosystems 10(6), 964-974.

Fischer, J., Brosi, B., Daily, G.C., Ehrlich, P.R., Goldman, R., Goldstein, J., Lindenmayer, D.B., Manning, A.D., Mooney, H.A., Pejchar, L., Ranganathan, J., 2008. Should agricultural policies encourage land sparing or wildlife-friendly farming?. Front. Ecol. Environ. 6(7), 380-385.

Fischer, J., Abson, D.J., Butsic, V., Chappell, M.J., Ekroos, J., Hanspach, J., Kuemmerle, T., Smith, H.G., von Wehrden, H., 2014. Land sparing versus land sharing: moving forward. Conserv. Lett. 7(3), 149-157.

Fleishman, E., Betrus, C.J., Blair, R.B., 2003. Effects of spatial scale and taxonomic group on partitioning of butterfly and bird diversity in the Great Basin, USA. Landscape Ecol. 18, 675–685.

Flohre, A., Fischer, C., Aavik, T., Bengtsson, J., Berendse, F., Bommarco, R., Ceryngier, P., Clement, L.W., Dennis, C., Eggers, S., Emmerson, M., Geiger, F., Guerrero, I., Hawro, V., Inchausti, P., Liira, J., Morales, M.B., Onate, J.J., Part, T., Weisser, W.W., Winqvist, C.,Thies, C., Tscharntke, T., 2011. Agricultural intensification and biodiversity partitioning in European landscapes comparing plants, carabids, and birds. Ecol. Appl. 21, 1772-1781.

Flynn, D.F., Gogol-Prokurat, M., Nogeire, T., Molinari, N., Richers, B.T., Lin, B.B., Simpson, N., Mayfield, M.M. DeClerck, F., 2009. Loss of functional diversity under land use intensification across multiple taxa. Ecol. Lett. 12(1), 22-33.

Flynn, D. F., Mirotchnick, N., Jain, M., Palmer, M. I., Naeem, S., 2011. Functional and phylogenetic diversity as predictors of biodiversity-ecosystem-function relationships. Ecology 92(8), 1573-1581.

Foley, J. A., DeFries, R., Asner, G. P., Barford, C., Bonan, G., Carpenter, S. R., Chapin, F.S., Coe, M.T., Daily, G.C., Gibbs, H.K., Helkowski, J.H., Holloway, T., Howard, E.A., Kucharik, C.J., Monfreda, C., Patz, J.A., Prentice, I.C., Ramankutty, N., Snyder, P.K., 2005. Global consequences of land use. Science 309, 570-574.

Foley, J.A., Ramankutty, N., Brauman, K.A., Cassidy, E.S., Gerber, J.S., Johnston, M., Mueller, N.D., O'Connell. C., Ray, D.K., West, P.C., Balzer, C., Bennett, E.M, Carpenter, S.R., Hill, J., Monfreda, C., Polasky, S., Rockström, J., Sheehan, J., Siebert, S., Tilman, D., Zaks, D.P.M., 2011. Solutions for a cultivated planet. Nature 478, 337-342

Foody, G.M., Cutler, M.E.J., 2003. Tree biodiversity in protected and logged Bornean tropical rain forests and its measurements by satellite remote sensing. J. Biogeogr. 30, 1053-1066.

Fordyce, J.A., DeVries, P.J., 2016. A tale of two communities: Neotropical butterfly assemblages show higher beta diversity in the canopy compared to the understory. Oecologia 181(1), 235-243.

Fourcade, Y., Ranius, T., Ockinger, E., 2017. Temperature drives abundance fluctuations, but spatial dynamics is constrained by landscape configuration: implications for climate-driven range shift in a butterfly. J. Anim. Ecol. 86, 1339-1351.

Frahnert, S., Eckhoff, P., Fiebig, J., 2012. Ernst Schäfer's comprehensive historical bird collection from Sikkim and Tibet at the Museum für naturkunde Berlin. Indian BIRDS 7(6), 160.

Francesconi, W., Nair, P.K.R., Levey, D.J., Daniels, J., Cullen Jr, L., 2013. Butterfly distribution in fragmented landscapes containing agroforestry practices in Southeastern Brazil. Agroforest. Syst. 87, 1321-1338.

Fujita, M., Koike, F., 2009. Landscape effects on ecosystems: birds as active vectors of nutrient transport to fragmented urban forests versus forest-dominated landscapes. Ecosystems 12(3), 391-400.

Fujita, M., Kameda, K.O., 2016. Nutrient dynamics and nutrient cycling by birds. In: Şekercioğlu, Ç.H., Wenny, D.G., Whelan, C.J., (Eds.), Why Birds Matter: Avian Ecological Function and Ecosystem Services. University of Chicago Press. pp. 271-291.

Gabriel, D., Sait, S.M., Hodgson, J.A., Schmutz, U., Kunin, W.E., Benton, T.G., 2010 Scale matters: the impact of organic farming on biodiversity at different spatial scales. Ecol. Lett. 13, 858-869.

Gámez-Virués, S., Perovic, D.J., Gossner, M.M., Börschig, C., Blüthgen, N., de Jong, H., Simons, N. K., Klein, A.M., Krauss, J., Maier, G., Scherber, C., Steckel, J., Rothenwöhrer, C., Steffan-Dewenter, I., Weiner, C. N., Weisser, W., Werner, M., Tscharntke, T.,Westphal, C., 2015. Landscape simplification filters species traits and drives biotic homogenization. Nat. Commun.6, 8568.

Ganguli-Lachungpa, U., Islam, M.Z., Rahamani, A.R., 2007. Important bird areas of Sikkim: priority sites for conservation. Gangtok: Department of Forest Environment and Wildlife Management.

García-Navas, V., Sattler, T., Schmid, H., Ozgul, A., 2020. Temporal homogenization of functional and beta diversity in bird communities of the Swiss Alps. Divers. Distr. 26(8), 900-911.

Garibaldi, L.A., Steffan-Dewenter, I., Winfree, R., Aizen, M.A., Bommarco, R., Cunningham, S.A., Kremen, C., Carvalheiro, L.G., Harder, L.D., Afik, O., Bartomeus, I., Benjamin, F., Boreux, V., Cariveau, D., Chacoff, N.P., Dudenhöffer, J.H., Freitas, B.M., Ghazoul, J., Greenleaf, S., Hipolito, J., Holzschuh, A., Howlett, B., Isaacs, R., Javorek, S.K., Kennedy, C.M., Krewenka, K.M., Krishnan, S., Mandelik, Y., Mayfield, M.M., Motzke, I., Munyuli, T., Nault, B.A., Otieno, M., Petersen, J., Pisanty, G., Potts, S.G., Rader, R., Ricketts, T.H., Rundlöf, M., Seymour, C.L., Schuepp, C., Szentgyorgyi, H., Taki, H., Tscharntke, T., Vergara, C.H., Viana, B.F., Wanger, T.C., Westphal, C., Williams, N., Klein, A.M., 2013. Wild pollinators enhance fruit set of crops regardless of honey bee abundance. Science 339, 1608-1611.

Genghini, M., Gellini, S., Gustin, M., 2006. Organic and integrated agriculture: the effects on bird communities in orchard farms in northern Italy. Biodivers. Conserv. 15(9), 3077-3094.

Ghosh-Harihar, M., 2013. Distribution and abundance of foliage-arthropods across elevational gradients in the east and west Himalayas. Ecol. Res. 28, 125-130.

Ghosh-Harihar, M., An, R., Athreya, R., Borthakur, U., Chanchani, P., Chetry, D., Datta, A., Harihar, A., Karanth, K.K., Mariyam, D., Mohan, D., 2019. Protected areas and biodiversity conservation in India. Biol. Conserv. 237, 114-124.

GIAHS, 2007. Sikkim Himalayas-Agriculture: Improving and scaling up of the traditionally managed agricultural systems of global significance. [Online]https://www.fao.org/nr/giahs/other-systems/other/asia-pacific/en/.

Gibbs, H.K., Ruesch, A.S., Achard, F., Clayton, M.K., Holmgren, P., Ramankutty, N., Foley, J.A., 2010. Tropical forests were the primary sources of new agricultural land in the 1980s and 1990s. Proc. Natl. Acad. Sci. U.S.A. 107(38), 16732-16737.

Gibson, L., Lee, T.M., Koh, L.P., Brook, B.W., Gardner, T.A., Barlow, J., Peres, C.A., Bradshaw, C.J., Laurance, W.F., Lovejoy, T.E., Sodhi, N.S., 2011. Primary forests are irreplaceable for sustaining tropical biodiversity. Nature 478 (7369), 378-381.

Gleditsch, J.M., Hruska, A.M., Foster, J.T., 2017. Connecting resource tracking by frugivores to temporal variation in seed dispersal networks. Front. Ecol. Evol. 5, 98.DOI:10.3389/fevo.2017.00098

Goded, S., Ekroos, J., Domínguez, J., Guitián, J.A., Smith, H.G., 2018. Effects of organic farming on bird diversity in North-West Spain. Agric. Ecosyst. Environ. 257, 60-67.

Goded, S., Ekroos, J., Azcárate, J.G., Guitián, J.A., Smith, H.G., 2019. Effects of organic farming on plant and butterfly functional diversity in mosaic landscapes. Agric. Ecosyst. Environ. 284, 106600.

Godfray, H.C.J., Beddington, J.R., Crute, I.R., Haddad, L., Lawrence, D., Muir, J.F., Pretty, J., Robinson, S., Thomas, S.M., Toulmin, C., 2010. Food security: the challenge of feeding 9 billion people. Science 327, 812-818.

Goodale, E., Kotagama, S.W., Raman, T.R.S., Sidhu, S., Goodale, U.M., Parker, S., Chen, J., 2014. The response of birds and mixed-species bird flocks to human-modified landscapes in Sri Lanka and Southern India. For. Ecol. Manag. 329, 384-392.

Gove, A.D., Hylander, K., Nemomisa, S., Shimelis, A., 2008. Ethiopian coffee cultivation-Implications for bird conservation and environmental certification. Conserv. Lett. 1(5), 208-216.

Grace, J.B., Anderson, T.M., Seabloom, E.W., 2016. Integrative modelling reveals mechanisms linking productivity and plant species richness. Nature 529, 390-393.

Grass, I., Loos, J., Baensch, S., Batáry, P., Librán-Embid, F., Ficiciyan, A., Klaus, F., Riechers, M., Rosa, J., Tiede, J. Udy, K., 2019. Land-sharing/-sparing connectivity landscapes for ecosystem services and biodiversity conservation. People and Nature, 1(2), 262-272.

Greenberg, R., Bichier, P., Sterling, J., 1997a. Bird populations in rustic and planted shade coffee plantations of eastern Chiapas, Mexico. Biotropica 29(4), 501-514.

Greenberg, R., Bichier, P., Angon, A.C., Reitsma, R., 1997b. Bird Populations in Shade and Sun Coffee Plantations in Central Guatemala. Conserv. Biol. 11(2), 448-459.

Greenler, S.M., Ebersole, J.J., 2015. Bird communities in tropical agroforestry ecosystems: An underappreciated conservation resource. Agrofor. Syst. 89(4), 691-704.

Grimmett, R., Inskipp, C., Inskipp, T., 2011. Birds of the Indian subcontinent. Christopher Helm, London.

Grimmett, R., Inskipp, C., Inskipp, T., Sherub, 2019. Birds of Bhutan and the Eastern Himalayas. Christopher Helm, London.

Gupta, I.J., Mondal, D.K., 2005. Red Data Book, Part II: Butterflies of India. Zoological Society of India, Kolkata.

Hadley, A.S., Betts, M.G., 2012. The effects of landscape fragmentation on pollination dynamics: absence of evidence not evidence of absence. Biol. Rev. 87, 526-544.

Hammond, P.C., Miller, J.C., 1998. Comparison of the biodiversity of Lepidoptera within three forested ecosystems. Ann. Entomol. Soc. Am. 91(3), 323-328.

Hanspach, J., Loos, J., Dorresteijn, I., von Wehrden, H., Moga, C. I., David, A., 2015. Functional diversity and trait composition of butterfly and bird communities in farmlands of Central Romania. Ecosyst. Health and Sustain. 1(10), 1-8.

Haribal, M., 1992. The butterflies of Sikkim Himalaya and their natural history. Sikkim Nature Conservation Foundation, Gangtok, India.

Harvey, C.A., Villalobos, J.A.G., 2007. Agroforestry systems conserve species-rich but modified assemblages of tropical birds and bats. Biodivers. Conserv. 16(8), 2257-2292.

Harvey, C.A., Medina, A., Sánchez, D.M., Vílchez, S., Hernández, B., Saenz, J.C., Maes, J.M., Casanoves, F., Sinclair, F.L., 2006. Patterns of animal diversity in different forms of tree cover in agricultural landscapes. Ecol. Appl. 16(5), 1986-1999.

Haslem, A., Bennett, A.F., 2008. Birds in agricultural mosaics: the influence of landscape pattern and countryside heterogeneity. Ecol. Appl. 18(1), 185-196.

Hausner, V.H., Yoccoz, N.G., Ims, R.A., 2003. Selecting indicator traits for monitoring land use impacts: birds in northern coastal birch forests. Ecol. Appl. 13(4), 999-1012.

Hawkins, B.A., Porter, E.E., 2002. Does herbivore diversity depend on plant diversity? The case of California butterflies. Am. Nat. 161(1), 40-49.

Hernandez, S.M., Mattsson, B.J., Peters, V.E., Cooper, R.J., Carroll, C.R., 2013. Coffee agroforests remain beneficial for neotropical bird community conservation across seasons. PLoS One 8(9), e65101.

Herrando, S., Brotons, L., Anton, M., Paramo, F., Villero, D., Titeux, N., Quesada, J., Stefanescu, C., 2016. Assessing impacts of land abandonment on Mediterranean biodiversity using indicators based on bird and butterfly monitoring data. Environ. Conserv. 43, 69-78.

Hiley, J.H., Bradbury, R.B., Thomas, C.D., 2016. Impacts of habitat change and protected areas on alpha and beta diversity of Mexican birds. Divers. Distrib. 22, 1245-1254.

Hobbs, N. T., Hilborn, R., 2006. Alternatives to statistical hypothesis testing in ecology: a guide to self teaching. Ecol. Appl. 16(1), 5-19.

Hobbs, R., Arico, S., Aronson, J., Baron, J. S., Bridgewater, P., Cramer, V., Epstein, P.R., Ewel, J.J., Klink, C.A., Lugo, A.E., Norton, D., Ojima, D., Richardson, D.M., Sanderson, E.W., Valladares, F., Vila, M., Zamora, R., Zobel, M., 2006. Novel ecosystems: theoretical and management aspects of the new ecological world order. Glob. Ecol. Biogeogr.15(1), 1-7.

Hole, D.G., Perkins, A.J., Wilson, J.D., Alexander, I.H., Grice, P.V., Evans, A.D., 2005. Does organic farming benefit biodiversity? Biol. Conserv. 122, 113-130.

Hooker, J.D., 1855. Himalayan Journals Notes of a Naturalist in Bengal the Sikkim and Nepal Himalayas the Khasi Mountains, etc. (Vol. I-II).John Murray, London.

Horner-Devine, M.C., Daily, G.C., Ehrlich, P.R., Broggs, C.L., 2003. Countryside biogeography of tropical butterflies. Conserv. Biol. 17, 168-177.

Hortal, J., Borges, P.A.V. Gaspar, S., 2006. Evaluating the performance of species richness estimators: sensitivity to sample grain size. J. Anim. Ecol. 75, 274-287.

Hu, Y., Ding, Z., Jiang, Z., Quan, Q., Guo, K., Tian, L., Hu, H. Gibson, L., 2018. Birds in the Himalayas: What drives beta diversity patterns along an elevational gradient?. Ecol. Evol. 8(23), 11704-11716.

Huston, M.A., 1997. Hidden treatments in ecological experiments: re-evaluating the ecosystem function of biodiversity. Oecologia, 110(4), 449-460.

Ibarra, J.T., Martin, K., 2015. Biotic homogenization: Loss of avian functional richness and habitat specialists in disturbed Andean temperate forests. Biol. Conserv. 192, 418-427.

Ibarra, J.T., Martin, M., Cockle, K.L. and Martin, K., 2017. Maintaining ecosystem resilience: functional responses of tree cavity nesters to logging in temperate forests of the Americas. Sci. Rep.7(1), 1-9.

ILTEO, 2015. Indian Long Term Ecological Observatories, Ministry of Environment, Forest & Climate Change, Government of India, pp. 1-52.

Isbell, F., Craven, D., Connolly, J., Loreau, M., Schmid, B., Beierkuhnlein, C., Bezemer, T.M., Bonin, C., Bruelheide, H., de Luca, E., Ebeling, A., Griffin, J., Qinfeng, G., Hautier, Y., Hector, A., Jentsch, A., Kreyling, J., Lanta, V., Manning, P., Meyer, S.T., Mori, A.S., Naeem, S., Niklaus, P.A., Polley, H.W., Reich, P.B., Roscher, C., Sea-bloom, E.W., Smith, M.D., Thakur, M.P., Tilman, D., Tracy, B.F., van der Putten, W.H., van Ruijven, J., Weigelt, A., Weisser, W.W., Wilsey, B., Eisenhauer, N., 2015. Biodiversity increases the resistance of ecosystem productivity to climate extremes. Nature 526, 574-577.

ISFR, 2019. India state of forest report 2019. Forest Survey of India, Ministry of Environment, Forests & Climate Change, Government of India.

IUCN SSC Butterfly Specialist Group, 2020. Butterflies.https://www.iucn.org/commissions/ssc-groups/invertebrates/butterfly. (Accessed on 25/08/2020.)

Jain, A., Kunte, K., Webb, E.L., 2016. Flower specialization of butterflies and impacts of non-native flower use in a transformed tropical landscape. Biol. Conserv. 201, 184-191.

Jainulabdeen, S., Prasad, S.K., 2004. Severe infestation of cabbage butterfly, *Pieris brassicae* (Linnaeus) on six species of Brassica and effect of abiotic factors on its population dynamics. J. Entomol. Res. 28(3), 193-197.

Jankowski, J.E., Merkord, C.L., Rios, W.F., Cabrera, K.G., Revilla, N.S., Silman, M.R., 2013. The relationship of tropical bird communities to tree species composition and vegetation structure along an Andean elevational gradient. J. Biogeogr. 40(5), 950-962.

Jew, E.K., Loos, J., Dougill, A.J., Sallu, S.M., Benton, T.G., 2015. Butterfly communities in miombo woodland: Biodiversity declines with increasing woodland utilisation. Biol.Conserv. 192, 436-444.

Johnson, M.D., Kellermann, J.L., Stercho, A.M., 2010. Pest reduction services by birds in shade and sun coffee in Jamaica. Anim. conserv. 13(2), 140-147.

Jolli, V., 2017. Hydro power development and its impacts on the habitats and diversity of montane birds of Western Himalayas. Vestnik zoologii 51(4), 311-324.

Jonason, D., Ekroos, J., Öckinger, E., Helenius, J., Kuussaari, M., Tiainen, J., Smith, H.G. Lindborg, R., 2017. Weak functional response to agricultural landscape homogenisation among plants, butterflies and birds. Ecography 40(10), 1221-1230.

Jordani, M.X., Hasui, E., da Silva, V.X., 2015. Natural enemies depend on remnant habitat size in agricultural landscapes. J. For. Res. 25 (4). DOI: 10.1007/s11676-015-0043-y

Joshi, K., Bhatt, D. 2011. Birds of three different forest habitats in Nainital district (western Himalaya), Uttarakhand, India. Indian BIRDS 7(2), 33-37.

Kale, M. A., Dudhe, N., Kasambe, R., Bhattacharya, P., 2014. Crop depredation by birds in Deccan Plateau, India. Int. J. Biodivers. 2014, 8.DOI: https://dx.doi.org/10.1155/2014/947683

Kalda., O., Kalda., R., Liira, J., 2015. Multi-scale ecology of insectivorous bats in agricultural landscapes. Agr. Ecosyst. Environ. 199, 105-13.

Kaltsas, D., Dede, K., Giannaka, J., Nasopoulou, T., Kechagioglou, S., Grigoriadou, E., Raptis, D., Damos, P., Vasiliadis, I., Christopoulos, V., Loukaki, E., 2018. Taxonomic and functional diversity of butterflies along an altitudinal gradient in two NATURA 2000 sites in Greece. Insect Conserv. Divers. 11(5), 464-478.

Kan, I., Motro, Y., Horvitz, N., Kimhi, A., Leshem, Y., Yom-Tov, Y., Nathan, R., 2014. Agricultural rodent control using barn owls: is it profitable?. Am. J. Agric. Econ. 96(3), 733-752.

Kandel, P., Thapa, I., Chettri, N., Pradhan, R., Sharma, E., 2018. Birds of the Kangchenjunga Landscape, the Eastern Himalaya: status, threats and implications for conservation. Avian Res. 9, 9. DOI: https://doi.org/10.1186/s40657-018-0100-2

Karanth, K.K., Sankararaman, V., Dalvi, S., Srivathsa, A., Parameshwaran, R., Sharma, S., Robbins, P., Chhatre, A., 2016. Producing Diversity: Agroforests Sustain Avian Richness and Abundance in India's Western Ghats. Front. Ecol. Evol.4, 111.DOI: 10.3389/fevo.2016.00111

Karmakar, T., Nitin, R., Sarkar, V., Baidya, S., Mazumder, S., Chandrasekharan, V. K., Das R., Girish Kumar, G.S., Lokhande, S., Veino J., Veino L., Veino R., Mirza Z., Sanap, R. V., Sarkar B., Kunte K., 2018. Early stages and larval host plants of some northeastern Indian butterflies. J. Threat. Taxa 10(6), 11780-11799.

Karmakar, T., Wangchuk, S., Basu, D.N., Kunte, K., (2021). A new species of *Zographetus* Watson, 1893 (Lepidoptera: Hesperiidae) from Sikkim, eastern Himalaya, India. Zootaxa 5072 (4), 373-379.

Karp, D.S., Rominger, A.J., Zook, J., Ranganathan, J., Ehrlich, P.R., Daily, G.C., 2012. Intensive agriculture erodes β-diversity at large scales. Ecol. Lett. 15, 963-970.

Karp, D.S., Mendenhall, C.D., Sandí, R.F., Chaumont, N., Ehrlich, P.R., Hadly, E. A., Daily, G.C., 2013. Forest bolsters bird abundance, pest control and coffee yield. Ecol. Lett.16(11), 1339-1347.

Katayama, N., 2016. Bird diversity and abundance in organic and conventional apple orchards in northern Japan. Sci. Rep. 6, 34210. DOI: 10.1038/srep34210

Katayama, N., Osada, Y., Mashiko, M., Baba, Y. G., Tanaka, K., Kusumoto, Y., Okubo, S., Ikeda, H., Natuhara, Y., 2019. Organic farming and associated management practices benefit multiple wildlife taxa: a large-scale field study in rice paddy landscapes. J. Appl. Ecol., 56(8), 1970-1981.

Katuwal, H.B., Basnet, K., Khanal, B., Devkota, S., Rai, S.K., Gajurel, J.P., Scheidegger, C., Nobis, M.P., 2016a. Seasonal changes in bird species and feeding guilds along elevational gradients of the Central Himalayas, Nepal. PLoS One 11(7), e0158362

Katuwal, H.B., Bhandari, J., Thapa, V., Gurung, R., Chaudhary, R., Magar, T.G., Chaudhary, H., 2016b. How many birds do the sacred forests hold?. J. Zoo. Stud. 3(4), 7-19.

Kehimkar, I., 2008. The Book of Indian Butterflies. Bombay Natural History Society and Oxford University Press, Mumbai.

Kehimkar, I., 2016. Butterflies of India. Bombay Natural History Society, Mumbai.

Kellermann, J.L., Johnson, M.D., Stercho, A.M., Hackett, S.C., 2008. Ecological and economic services provided by birds on Jamaican Blue Mountain coffee farms. Conserv. Biol. 22(5), 1177-1185.

Kemp, J.E., Linder, H.P., Ellis, A.G., 2017. Beta diversity of herbivorous insects is coupled to high species and phylogenetic turnover of plant communities across short spatial scales in the Cape Floristic Region. J. Biogeogr. 44, 1813-1823.

Kennedy, C.M., Lonsdorf, E., Neel, M.C., Williams, N.M., Ricketts, T.H., Winfree, R., Bommarco, R., Brittain, C., Burley, A.L., Cariveau, D., Carvalheiro, L.G., Chacoff, N.P., Cunningham, S.A., Danforth, B.N., Dudenhöffer, J.H., Elle, E., Gaines, H.R., Garibaldi, L.A., Gratton, C., Holzschuh, A., Isaacs, R., Javorek, S.K., Jha, S., Klein, A.M., Krewenka, K., Mandelik, Y., Mayfield, M.M., Morandin, L., Neame, L.A., Otieno, M., Park, M., Potts, S.G., Rundlöf, M., Saez, A., Steffan-Dewenter, I., Taki, H., Viana, B.F., Westphal, C., Wilson, J.K., Greenleaf, S.S., Kremen, C., 2013. A global quantitative synthesis of local and landscape effects on wild bee pollinators in agroecosystems. Ecol. Lett. 16, 584-599.

Kessler, M., Abrahamczyk, S., Bos, M., Buchori, D., Putra, D.D., Gradstein, S.R., Höhn, P., Kluge, J., Orend, F., Pitopang, R., Saleh, S., 2009. Alpha and beta diversity of plants and animals along a tropical land-use gradient. Ecol. Appl. 19(8), 2142-2156.

Khanal, B., 2013. Study on changes in butterfly fauna at different altitudinal levels in central Nepal. Ph.D. Thesis, Mizoram University, India.

Kitahara, M., Yumoto, M., Kobayashi, T., 2008. Relationship of butterfly diversity with nectar plant species richness in and around the Aokigahara primary woodland of Mount Fuji, central Japan. Biodivers. Conserv. 17, 2713-2734.

Kitchell, J.F., Schindler, D.E., Herwig, B.R., Post, D.M., Olson, M.H., Oldham, M., 1999. Nutrient cycling at the landscape scale: the role of diel foraging migrations by geese at the Bosque del Apache National Wildlife Refuge, New Mexico. Limnol. Oceanogr.44(3), 828-836.

Klein, A.M., Vaissièrè, B.E., Cane, J.H., Steffan-Dewenter, I., Cunningham, S.A., Kremen, C., Tscharntke, T., 2007. Importance of pollinators in changing landscapes for world crops. Proc. R. Soc. B 274, 303-313.

Knight, E.C., Mahony, N.A., Green, D.J., 2016. Effects of agricultural fragmentation on the bird community in sagebrush shrubsteppe. Agric. Ecosyst. Environ. 223, 278-288.

Koh, L.P., 2007. Impacts of land use change on South-east Asian forest butterflies: a review J. Appl. Ecol. 44, 703-713.

Koh, L.P., Sodhi, N.S., 2004. Co-extinctions of tropical butterflies and their host plants. Biotropica 36, 272-274.

Konvicka, M., Benes, J., Polakova, S., 2016. Smaller fields support more butterflies: comparing two neighbouring European countries with different socioeconomic heritage. J. Insect Conserv. 20(6), 1113-1118.

Kottawa-Arachchi, J.D., Gamage, R.N., 2015. Avifaunal diversity and bird community responses to man-made habitats in St. Coombs Tea Estate, Sri Lanka. J. Threat. Taxa 7(2), 6878-6890.

Kremen, C., 1992. Assessing the indicator properties of species assemblages for natural areas monitoring. Ecol. Appl. 2, 203-217.

Kremen, C., Williams, N.M., Aizen, M.A., Gemmill-Herren, B., LeBuhn, G., Minckley, R., Packer, L., Potts, S.G., Roulston, T., Steffan-Dewenter, I., Vàzquez, D.P., Winfree, R., Adams, L., Crone, E.E., Greenleaf, S.S., Keitt, T.H., Klein, A.-M., Regetz, J., Ricketts, T.H., 2007. Pollination and other ecosystem services produced by mobile organisms: a conceptual framework for the effects of land-use change. Ecol. Lett. 10, 299-314.

Kross, S.M., Tylianakis, J.M., Nelson, X.J., 2012. Effects of introducing threatened falcons into vineyards on abundance of passeriformes and bird damage to grapes. Conserv. Biol. 26(1), 142-149.

Kross, S.M., Bourbour, R.P., Martinico, B.L., 2016. Agricultural land use, barn owl diet, and vertebrate pest control implications. Agric. Ecosyst. Environ. 223, 167-174.

Kry, M., Lee, W.S., Park, S.J., Rhim, S.J., 2009. Bird communities in rice field and grasslands during the dry season in the Mekong Delta, Cambodia. J. Korean For. Soc. 98(2), 197-202.

Kühsel, S. Blüthgen, N., 2015. High diversity stabilizes the thermal resilience of pollinator communities in intensively managed grasslands. Nat. Commun. 6, 7989.

Kumar, S., Simonson, S.E., Stohlgren, T.J., 2009. Effects of spatial heterogeneity on butterfly species richness in Rocky Mountain National Park, CO, USA. Biodivers. Conserv. 18, 739-763.

Kumari, S., Behera, M.D., Tewari, H.R., 2010. Identification of potential ecotourism sites in West District, Sikkim using geospatial tools. Trop. Ecol. 51(1), 75-85.

Kunte, K., 1997. Seasonal patterns in butterfly abundance and species diversity in four tropical habitats in northern Western Ghats. J. Biosci. 22, 593-603.

Kunte, K., 2008. The Wildlife (Protection) Act and conservation prioritization of butterflies of the Western Ghats, southwestern India. Curr. Sci.94(6), 729-735.

Kunte, K., 2010.Rediscovery of the federally protected Scarce Jester Butterfly *Symbrenthia silana* de Nicéville, 1885 (Nymphalidae: Nymphalinae) from the Eastern Himalaya and Garo Hills, northeastern India. J. Threat. Taxa 2(5), 858-866.

Kunte, K., Joglekar, A., Ghate, U., Pramod, P., 1999. Patterns of butterfly, birds and tree diversity in the Western Ghats. Curr. Sci. 77, 577-586.

Kunte, K., Roy, P., Kalesh, S., Kodandaramaiah, U., (Eds.), 2018. Butterflies of India, v. 2.50. Indian Foundation for Butterflies. https://www.ifoundbutterflies.org

Kuussaari, M., Saarinen, M., Korpela, E. L., Pöyry, J., Hyvönen, T., 2014. Higher mobility of butterflies than moths connected to habitat suitability and body size in a release experiment. Ecol. Evol., 4(19), 3800-3811.

Laiolo, P., 2004. Diversity and structure of the bird community overwintering in the Himalayan subalpine zone: is conservation compatible with tourism?. Biol. Conserv. 115(2), 251-262.

Laiolo, P., 2005. Spatial and seasonal patterns of bird communities in Italian agroecosystems. Conserv. Biol. 19(5), 1547-1556.

Laliberté, E., Legendre, P., 2010. A distance-based framework for measuring functional diversity from multiple traits. Ecology, 91, 299-305.

Laliberté, E., Wells, J.A., DeClerck, F., Metcalfe, D.J., Catterall, C.P., Queiroz, C., Aubin, I., Bonser, S.P., Ding, Y., Fraterrigo, J.M., McNamara, S., Morgan, J.W., Merlos, D.S., Vesk, P.A., Mayfield, M.M., 2010. Land-use intensification reduces functional redundancy and response diversity in plant communities. Ecol. Lett.13, 76-86.

Laurance, W., Sayer, J., Cassman, K.G., 2014. Agricultural expansion and its impacts on tropical nature. Trends Ecol. Evol. 29, 107-116.

Lee, C.H., 1997. Barn owl for field rat control in cocoa. J. Trop. Agric. and Fd. Sc. 25(1), 43-54.

Lee, M.-K., Carrol, J.P., 2018. Effects of patch size and basal area on avian taxonomic and functional diversity in pine forests: Implication for the influence of habitat quality on the species–area relationship. Ecol. Evol. 8(14), 6909-6920.

Lee, M-B., Martin, J.A., 2017. Avian species and functional diversity in agricultural landscapes: Does landscape heterogeneity matter? PLoS One 12(1), e0170540.

Lees, A.C., Peres, C.A., 2008. Avian life-history determinants of local extinction risk in a hyper-fragmented neotropical forest landscape. Anim. Conser. 11(2), 128-137.

Lepcha, T.N., Chhetri, B., Chettri, P.K., 2020. Slender-billed Oriole *Oriolus tenuirostris* in the Sikkim Himalayas. Indian BIRDS 15(5), 155.

Letourneau, D.K., Jedlicka, J.A., Bothwell, S.G., Moreno, C.R., 2009. Effects of natural enemy biodiversity on the suppression of arthropod herbivores in terrestrial ecosystems. Annu. Rev. Ecol. Evol. Syst. 40, 573-592.

Levey, D.J., 1988. Spatial and temporal variation in Costa Rican fruit and fruit-eating bird abundance. Ecol. Monogr. 58(4), 251-269.

Levey, D.J., Bolker, B.M., Tewksbury, J.J., Sargent, S., Haddad, N.M., 2005. Effects of landscape corridors on seed dispersal by birds. Science, 309(5731), 146-148.

Li, N., Chu, H., Qi, Y., Li, C., Ping, X., Sun, Y. Jiang, Z., 2019. Alpha and beta diversity of birds along elevational vegetation zones on the southern slope of Altai Mountains: Implication for conservation. Glob. Ecol. Conserv. 19, e00643.

Li, S., Zou, F., Zhang, Q., Sheldon, F.H., 2013. Species richness and guild composition in rubber plantations compared to secondary forest on Hainan Island, China. Agrofor. Syst.87(5), 1117-1128.

Lichstein, J.W., 2007. Multiple regression on distance matrices: a multivariate spatial analysis tool. Plant Ecol. 188(2), 117-131.

Lien, V.V. Yuan, D., 2003. The differences of butterfly (Lepidoptera, Papilionoidea) communities in habitats with various degrees of disturbance and altitudes in tropical forests of Vietnam. Biodivers. Conserv. 12, 1099-1111.

Ligeza, S., Smal, H., 2003. Accumulation of nutrients in soils affected by perennial colonies of piscivorous birds with reference to biogeochemical cycles of elements. Chemosphere, 52(3), 595-602.

Lin, N., Nam, T.T., Perera, J., 2012. Response of birds to different management types of tea cultivation in a forest-agriculture landscape. In: Harrison, R., Shi, L.L., Liu, J.X. (Eds.) Proceedings of the advanced field course in ecology and conservation -XTBG 2012, pp. 11-20.

Lindell, C.A., Steensma, K.M., Curtis, P.D., Boulanger, J.R., Carroll, J.E., Burrows, C., Lusch, D.P., Rothwell, N.L., Wieferich, S.L., Henrichs, H.M., Leigh, D.K., 2016. Proportions of bird damage in tree fruits are higher in low-fruit-abundance contexts. Crop Prot. 90, 40-48.

Lindell, C., Eaton, R.A., Howard, P.H., Roels, S.M., Shave, M.E., 2018. Enhancing agricultural landscapes to increase crop pest reduction by vertebrates. Agric. Ecosyst. Environ. 257, 1-11.

Lindenmayer, D.B., Cunningham, R.B., Donnelly, C.F., Nix, H., Lindenmayer, B.D., 2002. Effects of Forest Fragmentation on Bird Assemblages in a Novel Landscape Context. Ecol. Monogr. 72(1), 1-18.

Lindenmayer, D.B., Blanchard, W., Tennant, P., Barton, P., Ikin, K., Mortelliti, A., Okada, S., Crane, M., Michael, D., 2015. Richness is not all: how changes in avian functional diversity reflect major landscape modification caused by pine plantations. Divers. Distrib. 21, 836-847.

Lindström, S., Klatt, B., Smith, H., Bommarco, R., 2018. Crop management affects pollinator attractiveness and visitation in oil seed rape. Basic Appl. Ecol. 26, 82-88.

Lodh, R., Agarwala, B.K., 2016. Rapid assessment of diversity and conservation of butterflies in Rowa Wildlife Sanctuary: An Indo-Burmese hotspot-Tripura, NE India. Trop. Ecol. 57(2), 231-242.

Loos, J., Dorresteijn, I., Hanspach, J., Fust, P., Rakosy, L., Fischer, J., 2014. Low-intensity agricultural landscapes in Transylvania support high butterfly diversity: Implications for conservation. PLoS One 9(7), e103256.

Loos, J., Turtureanu, P. D., von Wehrden, H., Hanspach, J., Dorresteijn, I., Frink, J. P., Fischer, J., 2015. Plant diversity in a changing agricultural landscape mosaic in Southern Transylvania (Romania). Agric. Ecosyst. Environ. 199, 350-357.

Losey, J.E., Vaughan, M., 2006. The economic value of ecological services provided by insects. BioScience 56, 311-323.

Lozada, T., De Koning, G.H.J., Marché, R., Klein, A.M., Tscharntke, T., 2007. Tree recovery and seed dispersal by birds: comparing forest, agroforestry and abandoned agroforestry in coastal Ecuador. Perspect. Plant Ecol. Evol. Syst. 8(3), 131-140.

Luck, G.W., Lavorel, S., McIntyre, S., Lumb, K., 2012. Improving the application of vertebrate trait-based frameworks to the study of ecosystem services. J. Anim. Ecol.81(5), 1065-1076.

Luck, G.W., Carter, A., Smallbone, L., 2013. Changes in bird functional diversity across multiple land uses: interpretations of functional redundancy depend on functional group identity. PloS One 8(5), e63671.

Luck, G. W., Hunt, K., Carter, A., 2015. The species and functional diversity of birds in almond orchards, apple orchards, vineyards and eucalypt woodlots. Emu 115(2), 99-109.

Maas, B., Clough, Y., Tscharntke, T., 2013. Bats and birds increase crop yield in tropical agroforestry landscapes. Ecol. Lett.16(12), 1480-1487.

Maas, B., Tscharntke, T., Saleh, S., Putra, D.D., Clough, Y., 2015. Avian species identity drives predation success in tropical cacao agroforestry. J. Appl. Ecol. 52(3), 735-743.

Mac Nally, R., Fleishman, E., Bulluck, L.P., Betrus, C.J., 2004. Comparative influence of spatial scale on beta diversity within regional assemblages of birds and butterflies. J. Biogeogr. 31(6), 917-929.

Majumder, J., Lodh, R., Agarwala, B.K., 2012. Variation in butterfly diversity and unique species richness along different habitats in Trishna Wildlife Sanctuary, Tripura, northeast India. Check List 8(3), 432-436.

Mammides, C., Schleuning, M.,Böhning-Gaese, K., Schaab, G., Farwig, N.,Kadis, C.,Coulson T., 2015. The indirect effects of habitat disturbance on the bird communities in a tropical African forest. Biodivers. Conserv. 24, 3083-3107.

Mandal, J. Raman, T.R.S., 2016. Shifting agriculture supports more tropical forest birds than oil palm or teak plantations in Mizoram, northeast India. Condor 18, 345-359.

Mandal, K., Bhattacharya, K., Chatterjee, P., Dey, P.K., Maheswaran, G.,Alam, I., 2018. Aves. In: Chandra, K., Gupta, D., Gopi, K.C., Tripathy, B., Kumar, V., (Eds.), Faunal Diversity of Indian Himalaya. Zoological Survey of India, Kolkata, pp. 831-854.

Mangan, A.M., Pejchar, L., Werner, S.J., 2017. Bird use of organic apple orchards: Frugivory, pest control and implications for production. PloS One, 12(9), e0183405.

Mani, M.S., 1974. Ecology and Biogeography in India. Dr. W. Junk, Publishers, The Hague: 1-773.

Manish, K., Pandit, M.K., 2018. Phylogenetic diversity, structure and diversification patterns of endemic plants along the elevational gradient in the Eastern Himalaya. Plant Ecol. Divers. 1-13.

Manish, K., Pandit, M.K., 2019. Identifying conservation priorities for plant species in the Himalaya in current and future climates: A case study from Sikkim Himalaya, India. Biol. Conserv. 233, 176-184.

Manish, K., Pandit, M.K., Telwala, Y., Nautiyal, D.C., Koh, L.P., Tiwari, S., 2017. Elevational plant species richness patterns and their drivers across non-endemics, endemics and growth forms in the Eastern Himalaya. J. Plant Res. 130 (5), 829-844.

Markandya, A., Taylor, T., Longo, A., Murty, M.N., Murty, S., Dhavala, K., 2008. Counting the cost of vulture decline-an appraisal of the human health and other benefits of vultures in India. Ecol. Econ. 67(2), 194-204.

Martin, E.A., Viano, M., Ratsimisetra, L., Laloë, F., Carrière, S.M., 2012. Maintenance of bird functional diversity in a traditional agroecosystem of Madagascar. Agric. Ecosyst. Environ.149, 1-9.

Martínez, D., García, D., 2017. Role of avian seed dispersers in tree recruitment in woodland pastures. Ecosystems 20(3), 616-629.

Martins, K.T., Gonzalez, A., Lechowicz, M.J., 2015. Pollination services are mediated by bee functional diversity and landscape context. Agric. Ecosyst. Environ. 200, 12-20.

Mas, A.H., Dietsch, T.V., 2003. An index of management intensity for coffee agroecosytems to evaluate butterfly species richness. Ecol. Appl. 13, 1491-1501.

Mas, A.H., Dietsch, T.V., 2004. Linking shade coffee certification to biodiversity conservation: butterflies and birds in Chiapas, Mexico. Ecol. Applic. 14(3), 642-654.

Maseko, M.S.T., Zungu, M.M., Ehlers Smith, D.A., Ehlers Smith, Y.C., Downs, C.T., 2019. High microhabitat heterogeneity drives high functional traits in forest birds in five protected forest areas in the urban mosaic of Durban, South Africa. Glob. Ecol. Conserv. 18, e00645.

Mason, N.W.H., Mouillot, D., Lee, W.G., Wilson, B., 2005. Functional richness, functional evenness and functional divergence: The primary components of functional diversity. Oikos 111, 112-118.

Matson, P.A., Parton, W.J., Power, A.G., Swift, M.J., 1997. Agricultural intensification and ecosystem properties.Science 277, 504-509.

Matuoka, M.A., Benchimol, M., Morante-Filho, J.C., 2020. Tropical forest loss drives divergent patterns in functional diversity of forest and non-forest birds. Biotropica 52(4), 738-748.

Mayfield, M.M., Boni, M.E., Daily, G.C. Ackerly, D., 2005. Species and functional diversity of native and human-dominated plant communities. Ecology 86, 2365-2372.

Mayfield, M.M., Bonser, S.P., Morgan, J.W., Aubin, I., McNamara, S. Vesk, P.A., 2010. What does species richness tell us about functional trait diversity? Predictions and evidence for responses of species and functional trait diversity to land-use change. Glob. Ecol. Biogeogr. 19, 423-431.

McCune, B., Grace, J.B., 2002. Analysis of Ecological Communities. MjM Software design, Gleneden beach, Oregon, USA.

Mcneely, J.A., Schroth, G., 2006. Agroforestry and biodiversity conservation - traditional practices, present dynamics, and lessons for the future. Biodivers. Conserv. 15, 549-554.

Menon, T., Sridhar, H., Shahabuddin, G., 2019. Effects of extractive use on forest birds in Western Himalayas: Role of local and landscape factors. For. Ecol. Manage. 448, 457-465.

Menéndez, R., González-Megías, A., Collingham, Y., Fox, R., Roy, D.B., Ohlemüller, R., Thomas, C.D., 2007. Direct and indirect effects of climate and habitat factors on butterfly diversity. Ecology 88(3), 605-611.

Meyrom, K., Motro, Y., Leshem, Y., Aviel, S., Izhaki, I., Argyle, F., Charter, M., 2009. Nest-box use by the Barn Owl *Tyto alba* in a biological pest control program in the Beit She'an valley, Israel. Ardea 97(4), 463-467.

Michael, D.R., Wood, J.T., O'Loughlin, T., Lindenmayer, D.B., 2016. Influence of land sharing and land sparing strategies on patterns of vegetation and terrestrial vertebrate richness and occurrence in Australian endangered eucalypt woodlands. Agric. Ecosyst. Environ. 227, 24-32.

Mihindukulasooriya, M.W.D.M., Ranawana, K. B., Majer J.D., 2014. Comparison of butterfly diversity in natural and regenerating forest in a biodiversity conservation site at maragamuwa, Sri Lanka. J. Bio. Env. Sci. (3), 387-391.

Milder, J.C., DeClerck, F.A.J., Sanfiorenzo, A., Sánchez, D.M., Tobar, D.E., Zuckerberg, B., 2010. Effects of farm and landscape management on bird and butterfly conservation in western Honduras. Ecosphere 1(1), 1-22.

Mitchell, M.G.E., Bennett, E.M., Gonzalez, A., 2014. Forest fragments modulate the provision of multiple ecosystem services. J. Appl. Ecol. 51, 909-18.

Mittermeier, R.A., Turner, W.R., Larsen, F.W., Brooks, T.M., Gascon, C., 2011. Global Biodiversity Conservation: The Critical Role of Hotspots. In: Zachos F., Habel J. (Eds.), Biodiversity Hotspots. Springer, Berlin, Heidelberg.

Monnet, A.C., Jiguet, F., Meynard, C.N., Mouillot, D., Mouquet, N., Thuiller, W., Devictor, V., 2014. Asynchrony of taxonomic, functional and phylogenetic diversity in birds. Glob. Ecol. Biogeogr. 23(7), 780-788.

Montaño-Centellas, F.A., McCain, C., Loiselle, B.A., 2020. Using functional and phylogenetic diversity to infer avian community assembly along elevational gradients. Glob. Ecol. Biogeogr. 29(2), 232-245.

Montoya, J.M., Raffaelli, D., 2010. Climate change, biotic interactions and ecosystem services. Philos. Trans. R. Soc. Lond., B, Biol. Sci. 365, 2013-2018.

Morante-Filho, J.C., Arroyo-Rodríguez, V., Faria, D., 2016. Patterns and predictors of β-diversity in the fragmented Brazilian Atlantic forest: a multiscale analysis of forest specialist and generalist birds. J. Anim. Ecol. 85(1), 240-250.

Mouchet, M.A., Villéger, S., Mason, N.W., Mouillot, D., 2010. Functional diversity measures: an overview of their redundancy and their ability to discriminate community assembly rules. Funct. Ecol. 24(4), 867-876.

Mouillot, D., Graham, N.A.J., Villéger, S., Mason, N.W.H., Bellwood, D.R., 2013. A functional approach reveals community responses to disturbance. Trends Ecol. Evol. 28, 167–177.

Muhamad, D., Okubo, S., Miyashita, T., Takeuchi, K., 2013. Effects of habitat type, vegetation structure, and proximity to forests on bird species richness in a forest–agricultural landscape of West Java, Indonesia. Agrofor. Syst. 87(6), 1247-1260.

Muñoz-Sáez, A., Perez-Quezada, J.F., Estades, C.F., 2017. Agricultural landscapes as habitat for birds in central Chile. Rev. Chil. de Hist. Nat. 90, 3. DOI: https://doi.org/10.1186/s40693-017-0067-0

Munyuli, T. 2012. Assessment of indicator species of butterfly assemblages in coffee–banana farming system in central Uganda. Afr. J. Ecol. 50, 77-89.

Myers, M.C., Mason, J.T., Hoksch, B.J., Cambardella, C.A., Pfrimmer, J.D., 2015. Birds and butterflies respond to soil-induced habitat heterogeneity in experimental plantings of tallgrass prairie species managed as agroenergy crops in Iowa, USA. J. Appl. Ecol. 52(5), 1176-1187.

Naidoo, R., 2004. Species richness and community composition of songbirds in a tropical forest-agriculture landscape. Anim. Conserv.7, 93-105.

Nakano, S., Murakami, M., 2001. Reciprocal subsidies: dynamic interdependence between terrestrial and aquatic food webs. Proc. Natl. Acad. Sci. U.S.A.98(1), 166-170.

Naniwadekar, R., Mishra, C., Datta, A., 2015. Fruit resource tracking by hornbill species at multiple scales in a tropical forest in India. J. Trop. Ecol. 31(6), 477-490.

National Agroforestry Policy, 2014. National Agroforestry Policy. Department of Agriculture and Cooperation, Ministry of Agriculture, Government of India, pp. 1-13.

Nava-Díaz, R., Pineda-López, R., Dorantes-Euan, A., 2020. Drivers of Functional Composition of Bird Assemblages in Green Spaces of a Neotropical City: A Case Study from Merida, Mexico. Trop.Conserv.Sci. 13, 1-13.

Negi, V.S., Pathak, R., Rawal, R.S., Bhatt, I.D., Sharma, S., 2019. Long-term ecological monitoring on forest ecosystems in Indian Himalayan Region: Criteria and indicator approach. Ecol. Indic. 102, 374-381.

Nelson, S.M., 2007. Butterflies (Papilionoidea and Hesperioidea) as potential ecological indicators of riparian quality in the semi-arid western United States. Ecol. Indic. 7(2), 469-480.

Newbold, T., Scharlemann, J.P.W., Butchart, S.H.M., Şekercioğlu, C.H., Alkemade, R., Booth, H., Purves, D.W., 2013. Ecological traits affect the response of tropical forest bird species to land-use intensity. Proc. R. Soc. B280, 20122131. DOI: 10.1098/rspb.2012.2131

Newbold, T., Hudson, L.N., Hill, S.L.L., Contu, S., Lysenko, I., Senior, R.A., Boerger, L., Bennett, D.J., Choimes, A., Collen, B., Day, J., De Palma, A., Diaz, S., Echeverria-Londono, S., Edgar, M.J., Feldman, A., Garon, M., Harrison, M.L.K., Alhusseini, T., Ingram, D.J., Itescu, Y., Kattge, J., Kemp, V., Kirkpatrick, L., Kleyer, M., Correia, D.L.P., Martin, C.D., Meiri, S., Novosolov, M., Pan, Y., Phillips, H.R.P., Purves, D.W., Robinson, A., Simpson, J., Tuck, S.L., Weiher, E., White, H.J., Ewers, R.M., Mace, G.M., Scharlemann, J.P.W., Purvis, A., 2015. Global effects of land use on local terrestrial biodiversity. Nature 520, 45-50.

Nidup, T., Dorji, T. Tshering, U., 2014. Taxon diversity of butterflies in different habitat types in Royal Manas National Park. J. Entomol. Zool. Stud. 2 (6), 292-298.

Nummi, P., Holopainen, S., 2014. Whole-community facilitation by beaver: ecosystem engineer increases waterbird diversity. Aquatic Conserv: Mar. Freshw. Ecosyst. 24(5), 623-633.

Öckinger, E., Schweiger, O., Crist, T.O., Debinski, D.M., Krauss, J., Kuussaari, M., Petersen, J.D., Pöyry, J., Settele, J., Summerville, K.S., Bommarco, R., 2010. Life-history traits predict species responses to habitat area and isolation: a cross-continental synthesis. Ecol. Lett. 13(8), 969-979.

Ohwaki, A., Tanabe, S.I., Nakamura, K., 2007. Butterfly assemblages in a traditional agricultural landscape: importance of secondary forests for conserving diversity, life history specialists and endemics. Biodivers. Conserv. 16(5), 1521-1539.

Oliveira Hagen, E., Hagen, O., Ibáñez-Álamo, J.D., Petchey, O.L., Evans, K.L., 2017. Impacts of Urban Areas and Their Characteristics on Avian Functional Diversity. Front. Ecol. Evol. 5, 84. DOI: 10.3389/fevo.2017.00084

Ollerton, J., Winfree, R., Tarrant, S., 2011. How many flowering plants are pollinated by animals? Oikos 120, 321-326.

Otero, X.L., De La Peña-Lastra, S., Pérez-Alberti, A., Ferreira, T.O., Huerta-Diaz, M.A., 2018. Seabird colonies as important global drivers in the nitrogen and phosphorus cycles. Nat. Commun. 9, 246. DOI: https://doi.org/10.1038/s41467-017-02446-8

Padhye, A.D., Dahanukar, N., Paingankar, M., Deshpande, M., Deshpande, D., 2006. Zoo's Print Journal 21(3), 2175-2181.

Pakkala, T., Lindén, A., Tiainen, J., Tomppo, E., Kouki, J., 2014. Indicators of forest biodiversity: which bird species predict high breeding bird assemblage diversity in boreal forests at multiple spatial scales?. Ann. Zool. Fennici 51, 457-476.

Pakeman, R.J., 2011. Functional diversity indices reveal the impacts of land use intensification on plant community assembly. J. Ecol. 99(5), 1143-1151.

Pande, S., Dahanukar, N., 2011. The diet of Indian Eagle Owl *Bubo bengalensis* and its agronomic significance. J. Threat. Taxa. 3(8), 2011-2017.

Pandey, R., Khadka K. K., Ghimire, A., Jha, P., Pathak, U., 2017. Elevational distribution of butterflies in the Himalayas: a case study from Langtang National Park, Nepal. J. Mt. Sci. 14(7), 1384-1390.

Pandit, M.K. 2017. Life in the Himalaya: an ecosystem at risk. Harvard University Press, Cambridge and Massachusetts.

Pandit, M.K., Sodhi, N.S., Koh, L.P., Bhaskar, A., Brook, B.W., 2007. Unreported yet massive deforestation driving loss of endemic biodiversity in Indian Himalaya. Biodivers. Conserv. 16, 153-163.

Paradis, A., 2020. Landscape Filters of Functional Trait Diversity and Composition, Doctoral dissertation, University of Ottawa, Canada.

Pavoine, S., Baguette, M., Stevens, V.M., Leibold, M.A., Turlure, C., Bonsall, M.B., 2014. Life history traits, but not phylogeny, drive compositional patterns in a butterfly metacommunity. Ecology 95(12), 3304-3313.

Pearson, D.L., 1994. Selecting indicator taxa for the quantitative assessment of biodiversity. Philos. Trans. R. Soc. Lond., B, Biol. Sci. 345(1311), 75-79.

Pedley, S.M., Barbaro, L., Guilherme, J.L., Irwin, S., O'Halloran, J., Proença, V., Sullivan, M.J., 2019. Functional shifts in bird communities from semi-natural oak forests to conifer plantations are not consistent across Europe. PloS One 14(7), e0220155.

Pe'er, G., Zinngrebe, Y., Hauck, J., Schindler, S., Dittrich, A., Zingg, S., Tscharntke, T., Oppermann, R., Sutcliffe, L.M.E., Sirami, C., Schmidt, J., Hoyer, C., Schleyer, C., Lakner, S., 2016. Adding some green to the greening: improving the EU's ecological focus areas for biodiversity and farmers. Conserv. Lett. 10, 517-530.

Peh, K.S.H., de Jong, J., Sodhi, N.S., Lim, S.L.H., Yap, C.A.M., 2005. Lowland rainforest avifauna and human disturbance: persistence of primary forest birds in selectively logged forests and mixed-rural habitats of southern Peninsular Malaysia. Biol. Conserv. 123(4), 489-505.

Peisley, R.K., Saunders, M.E., Luck, G.W., 2015. A systematic review of the benefits and costs of bird and insect activity in agroecosystems. Springer Science Reviews, 3(2), 113-125.

Pellissier, V., Barnagaud, J.Y., Kissling, W.D., Şekercioğlu, Ç., Svenning, J.C., 2018. Niche packing and expansion account for species richness–productivity relationships in global bird assemblages. Glob. Ecol. Biogeogr. 27(5), 604-615.

Pereira, G.C.N., Coelho, M.S., do Vale Beirão, M., Braga, R.F., Fernandes, G.W. 2017. Diversity of fruit-feeding butterflies in a mountaintop archipelago of rainforest. PloS One 12(6), e0180007.

Perfecto, I., Mas, A., Dietsch, T., Vandermeer, J., 2003. Conservation of biodiversity in coffee agroecosystems: a tri-taxa comparison in southern Mexico. Biodivers. Conserv. 12, 1239-1252.

Perfecto, I., Vandermeer, J., 2010. The agroecological matrix as alternative to the land-sparing/agriculture intensification model. Proc. Natl. Acad. Sci. U.S.A.107(13), 5786-5791.

Perović, D., Gámez-Virués, S., Börschig, C., Klein, A.M., Krauss, J., Steckel, J., Rothenwöhrer, C., Erasmi, S., Tscharntke, T., Westphal, C., 2015. Configurational landscape heterogeneity shapes functional community composition of grassland butterflies. J. Appl. Ecol. 52, 505-513.

Petchey, O.L., Evans, K.L., Fishburn, I.S., Gaston, K.J., 2007. Low functional diversity and no redundancy in British avian assemblages. J. Anim. Ecol.76(5), 977-985.

Peters, M. K., Hemp, A., Appelhans, T., Becker, J. N., Behler, C., Classen, A., Detsch, F., Ensslin, A., Ferger, S.W., Frederiksen, S.B., Gebert, F., Gerschlauer, F., Gütlein, A., Helbig-Bonitz, M., Hemp, C., Kindeketa, W.J., Kühnel, A., Mayr, A.V., Mwangomo, E., Ngereza, C., Njovu,H.K., Otte, I., Pabst, H., Renner, M., Röder, J., Rutten, G., Costa, D.S., Sierra-Cornejo, N., Vollstädt, M.G.R., Dulle, H.I., Eardley, C.D., Howell, K.M., Keller, A., Peters, R.S., Ssymank, Kakengi, V., Zhang, J., Bogner, C., Böhning-Gaese, K., Brandl, R., Hertel, D., Huwe, B., Kiese, R., Kleyer, M., Kuzyakov, Y., Nauss, T., Schleuning, M., Tschapka, M., Fischer, M., Steffan-Dewenter, I. 2019. Climate–land-use interactions shape tropical mountain biodiversity and ecosystem functions. Nature, 568(7750), 88-92.

Peterson, C.A., Lee, S.L., Elliott, J.E., 2001. Scavenging of Waterfowl Carcasses by Birds in Agricultural Fields of British Columbia. J. Field Ornithol. 150-159.

Phalan, B., Onial, M., Balmford, A., Green, R.E., 2011. Reconciling Food Production and Biodiversity Conservation: Land Sharing and Land Sparing Compared. Science 333, 1289-1291.

Phalan, B., Green, R.E., Dicks, L.V., Dotta, G., Feniuk, C., Lamb, A., Strassburg, B.B.N., Williams, B.R., zu Ermgassen, E.K.H.J., Balmford, A., 2016. How can higher-yield farming help to spare nature? Science 351 (6272), 450-451.

Philpott, S.M., Bichier, P., Rice, R., Greenberg, R., 2007. Field-Testing Ecological and Economic Benefits of Coffee Certification Programs. Conserv. Biol.21(4), 975-998.

Philpott, S.M., Arendt, W.J., Armbrecht, I., Bichier, P., Diestch, T.V., Gordon, C., Greenberg, R., Perfecto, I., Reynoso-Santos, R., Soto-Pinto, L, Tejeda-Cruz, C., Williams-Linera, G., Valenzuela, J., Zolotoff, J.M., 2008. Biodiversity loss in Latin American coffee landscapes: review of the evidence on ants, birds, and trees. Conserv. Biol. 22, 1093-1105.

Philpott, S.M., Soong, O., Lowenstein, J. H., Pulido, A.L., Lopez, D.T., Flynn, D.F., DeClerck, F., 2009. Functional richness and ecosystem services: bird predation on arthropods in tropical agroecosystems. Ecol Appl. 19(7), 1858-1867.

Plexida, S.G., Sfougaris, A.I., Papadopoulos, N.T., 2012. Quantifying beetle and bird diversity in a Mediterranean mountain agro-ecosystem. Isr. J. Ecol. Evol. 58(1), 1-25.

Pollard, E., Yates, T.J., 1993. Monitoring butterflies for ecology and conservation. London: Chapman and Hall. p. 292.

Pollard, E., 1977. A method for assessing changes in the abundance of butterflies. Biol. Conserv. 12, 116-134.

Potts, S.G., Biesmeijer, J.C., Kremen, C., Neumann, P., Schweiger, O., Kunin, W.E., 2010. Global pollinator declines: trends, impacts and drivers. Trends Ecol. Evol. 25, 345-353.

Poyry, J., Paukkunen, J., Helioelae, J., Kuussaari, M., 2009. Relative contributions of local and regional factors to species richness and total density of butterflies and moths in semi-natural grasslands. Oecologia 160, 577-587.

Prabowo, W.E., Darras, K., Clough, Y., Toledo-Hernandez, M., Arlettaz, R., Mulyani, Y.A., Tscharntke, T., 2016. Bird responses to lowland rainforest conversion in Sumatran smallholder landscapes, Indonesia. PLoS One 11(5), e0154876.

Praveen J., Jayapal, R., Pittie, A., 2016. A checklist of the birds of India. Indian BIRDS 11, 113-170.

Praveen J., Jayapal, R., Pittie, A., 2020a. Checklist of the birds of India (v4.1). Website: https://www.indianbirds.in/india/ [Date of publication: 25 July, 2020].

Praveen, J. Jayapal, R., Pittie, A., 2020b. Threatened birds of India (v3.1). Website: https://www.indianbirds.in/india/ [Date of publication: 30 July, 2020].

Prescott, G.W., Gilroy, J.J., Haugaasen, T., Medina Uribe, C.A., Foster, W.A., Edwards, D.P., 2016. Reducing the impacts of neotropical oil palm development on functional diversity. Biol. Conserv. 197, 139-145.

Puig-Montserrat, X., Stefanescu, C., Torre, I., Palet, J., Fàbregas, E., Dantart, J., Arrizabalaga, A., Flaquer, C., 2017. Effects of organic and conventional crop management on vineyard biodiversity. Agric. Ecosyst. Environ. 243, 19-26.

R Core Team, 2017. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/

Radford, J.Q., Bennett, A.F., 2007. The relative importance of landscape properties for woodland birds in agricultural environments. J. Appl. Ecol. 44(4), 737-747.

Rahbek, C., 1997. The Relationship among Area, Elevation, and Regional Species Richness in Neotropical Birds. Am. Naturalist 149(5), 875-902.

Rahut, B., Dutta, M., Bonpo, C.R., 2012. First photographic record of Blyth’s Rosefinch *Carpodacus grandis* from Sikkim, India. IndianBIRDS, 7(5), 148-149.

Rai, S., Bhutia, K.D., Kunte, K., 2012. Recent sightings of two very rare butterflies, *Lethe margaritae* Elwes, 1882 and *Neptis nycteus* de Nicéville, 1890, from Sikkim, eastern Himalaya, India. J. Threat. Taxa, 4(14), 3319-3326.

Rákosy, L., Schmitt, T., 2011. Are butterflies and moths suitable ecological indicator systems for restoration measures of semi-natural calcareous grassland habitats?. Ecol. Indic. 11(5), 1040-1045.

Raman, T.R.S., 2001. Effect of slash-and-burn shifting cultivation on rainforest birds in Mizoram, northeast India. Conserv. Biol. 15(3), 685-698.

Raman, T.R.S., 2003. Assessment of census techniques for interspecific comparisons of tropical rainforest bird densities: a field evaluation in the Western Ghats, India. Ibis 145, 9-21.

Raman, T.R.S., 2006. Effects of habitat structure and adjacent habitats on birds in tropical rainforest fragments and shaded plantations in the Western Ghats, India. Biodivers. Conserv. 15, 1577-1607.

Raman, T.R.S., Rawat, G.S., Johnsingh, A.J.T., 1998. Recovery of tropical rainforest avifauna in relation to vegetation succession following shifting cultivation in Mizoram, north-east India. J. Appl. Ecol. 35(2), 214-231.

Raman, T.R.S., Joshi, N.V., Sukumar, R., 2005. Tropical rainforest bird community structure in relation to altitude, tree species composition, and null models in the Western Ghats, India. J. Bombay Nat. His. Soc. 102, 145-157.

Raman, T. S., Sukumar, R., 2002. Responses of tropical rainforest birds to abandoned plantations, edges and logged forest in the Western Ghats, India. Anim. Conserv. 5(3), 201-216.

Ranganathan, J., Daniels, R.J.R., Chandran, M.D.S., Ehrlich, P.R. Daily, G.C., 2008. Sustaining biodiversity in ancient tropical countryside. Proc. Natl. Acad. Sci. U.S.A.105, 17852-17854.

Reitsma, R., Parrish, J.D., McLarney, W., 2001. The role of cacao plantations in maintaining forest avian diversity in southeastern Costa Rica. Agrofor. Syst.53(2), 185-193.

Rey, P.J., 1995. Spatio-temporal variation in fruit and frugivorous bird abundance in olive orchards. Ecology, 76(5), 1625-1635.

Ribera, I., Dolédec, S., Downie, I.S., Foster, G.N., 2001. Effect of land disturbance and stress on species traits of ground beetle assemblages. Ecology 82(4), 1112-1129.

Riva, F., Pinzon, J., Acorn, J.H., Nielsen, S.E., 2020. Composite effects of cutlines and wildfire result in fire refuges for plants and butterflies in boreal treed peatlands. Ecosystems 23, 485-497.

Robinson, N., Armstead, S., Deane Bowers, M., 2012. Butterfly community ecology: the influences of habitat type, weather patterns, and dominant species in a temperate ecosystem. Entomol. Exp. Appl. 145(1), 50-61.

Round, P.D., Gale, G.A., Brockelman, W.Y., 2006.A comparison of bird communities in mixed fruit orchards and natural forest at Khao Luang, southern Thailand. Biodivers. Conserv. 15, 2873-2891.

Rundio, D.E., Lindley, S.T., 2012. Reciprocal fluxes of stream and riparian invertebrates in a coastal California basin with Mediterranean climate. Ecol. Res. 27(3), 539-550.

Rundlof, M., Bengtsson, J., Smith, H.G., 2008. Local and landscape effects of organic farming on butterfly species richness and abundance. J. Appl. Ecol. 45, 813-820.

Rurangwa, M.L., Aguirre-Gutiérrez, J., Mathews, T.J., Niyigaba, P., Wayman, J.P., Tobias, J.A., Whittaker, R.J., 2021. Effects of land-use change on avian taxonomic, functional and phylogenetic diversity in a tropical montane rainforest.Divers. Distrib. 27(9), 1732-1746.

Šálek, M., Hula, V., Kipson, M., Daňková, R., Niedobová, J., Gamero, A., 2018. Bringing diversity back to agriculture: Smaller fields and non-crop elements enhance biodiversity in intensively managed arable farmlands. Ecol. Indic. 90, 65-73.

Santos, T., Tellería, J.L., Díaz, M., Carbonell, R., 2006. Evaluating the benefits of CAP reforms: can afforestations restore bird diversity in Mediterranean Spain?. Basic Appl. Ecol. 7(6), 483-495.

Sathyakumar, S., Poudyal, K., Bhattacharya, T., Bashir, T., 2011. Galliformes of Khangchendzonga Biosphere Reserve, Sikkim, India. In: Arawatia, M.L., Tambe, S. (Eds.), Biodiversity of Sikkim: Exploring and Conserving Global Hotspots. IPR Department, Government of Sikkim, Gangtok, India, pp. 301-315.

Sayer, C.A., Bullock, J.M. Martin, P.A., 2017. Dynamics of avian species and functional diversity in secondary tropical forests. Biol. Conserv. 211, 1-9.

Schnug, E., Jacobs, F., Stöven, K., 2018. Guano: The White Gold of the Seabirds. In: Mikkola, H. (Ed.), Seabirds.IntechOpen, London, pp. 79-100.

Schriever, T.A., Cadotte, M.W., Williams, D.D., 2014. How hydroperiod and species richness affect the balance of resource flows across aquatic-terrestrial habitats. Aquat. Sci. 76(1), 131-143.

Schroth, G., Harvey, C., Vincent, G., 2004. Complex agroforests: their structure, diversity, and potential role in landscape conservation. In: Schroth, G., da Fonseca, G.A.B., Harvey, C.A., Gascon, C., Vasconcelos, H.L., Izac, A.M.N. (Eds.), Agroforestry and biodiversity conservation in tropical landscapes. Island Press, Washington, DC, pp 227-260.

Schulze, C.H., Waltert, M., Kessler, P.J.A., Pitopang, R., Shadbuddin, D., Veddeler, M., Muhlenberg, S., Gradstein, R., Leuschner, C., Steffen-Dewenter, I., Tscharntke, T., 2004a. Biodiversity indicator groups of tropical land-use systems: comparing plants, birds and insects. Ecol. Appl. 14, 321-1333.

Schulze, C.H., Steffan-Dewenter, I., Tscharntke, T., 2004b. Effects of land use on butterfly communities at the rain forest margin: a case study from Central Sulawesi. In: Gerold, G., Fremerey, M., Guhardja, E. (Eds.), Land Use, Nature Conservation and the Stability of Rainforest Margins in Southeast Asia. Springer-Verlag, Berlin, pp. 281-297.

Schulze, C.H., Schneeweihs, S., Fiedler, K., 2010. The potential of land-use systems for maintaining tropical forest butterfly diversity. In: Tscharntke, T., Lauschener, C., Veldkamp, E., Faust, H., Guhardja E., Bidin A. (Eds.), Tropical Rainforests and Agroforests under Global Change: Ecological and Socio-economic Valuations. Springer-Verlag, Berlin, pp. 73-96.

Sehgal, J., Mandal, D.K., Mandal, C., Vadivelu, S., 1992. Agro-ecological zones of India. Second Edition. Nagpur, India. Technical Bulletin No. 24. NBSS & LUP (ICAR), p. 130.

Sekar, S., 2012. A meta-analysis of the traits affecting dispersal ability in butterflies: can wingspan be used as a proxy? J. Anim. Ecol. 81(1), 174-184.

Şekercioğlu, C.H., 2002. Effects of forestry practices on vegetation structure and bird community of Kibale National Park, Uganda. Biol. Conserv. 107, 229-240.

Şekercioğlu, C.H., 2006. Increasing awareness of avian ecological function. Trends Ecol. Evol. 21(8), 464-471.

Şekercioğlu, C.H., 2012. Bird functional diversity and ecosystem services in tropical forests, agroforests and agricultural areas. J. Ornithol. 153, S153-S161.

Şekercioğlu, Ç.H., Daily, G.C., Ehrlich, P.R., 2004. Ecosystem consequences of bird declines. Proc. Natl. Acad. Sci. U.S.A.101(52), 18042-18047.

Şekercioğlu, C.H., Loarie, S.R., Oviedo-Brenes, F., Daily, G.C., Ehrlich, P.R., 2007. Persistence of forest birds in the Costa Rican agricultural countryside. Conserv. Biol. 21, 482-494.

Şekercioğlu, Ç.H., Wenny, D.G., Whelan, C.J. (Eds.), 2016. Why birds matter: Avian ecological function and ecosystem services. University of Chicago Press.

Sengupta, P., Banerjee, K. K., Ghorai, N., 2014. Seasonal diversity of butterflies and their larval food plants in the surroundings of upper Neora Valley National Park, a sub-tropical broad leaved hill forest in the eastern Himalayan landscape, West Bengal, India. J. Threat. Taxa 6(1), 5327-5342.

Sethi, P.I.A., Howe, H.F., 2009. Recruitment of hornbill-dispersed trees in hunted and logged forests of the Indian Eastern Himalaya. Conserv. Biol. 23(3), 710-718.

Seymour, C.L., Simmons, R.E., Joseph, G.S., Slingsby, J.A., 2015. On bird functional diversity: species richness and functional differentiation show contrasting responses to rainfall and vegetation structure in an arid landscape. Ecosystems 18(6), 971-984.

Shahabuddin, G., 1997. Preliminary observations on the role of coffee plantations as avifaunal refuges in the Palni hills of the Western Ghats. J. Bombay Nat. Hist. Soc. 94, 10-21.

Shahabuddin, G., Ali, R., 2001. Impacts of land use change on forest butterfly communities in the Western Ghats of Southern India. J. Trop. For. Sci. 13, 434-449.

Shahabuddin, G., Goswami, R., Krishnadas, M., Menon, T., 2021. Decline in forest bird species and guilds due to land use change in the Western Himalaya. Glob. Ecol. Conserv.25, e01447.

Sharma, E., Kerkhoff, E., 2004. Farming Systems in the Hindu Kush-Himalayan Region. In: Adhikari, R., Adhikari, K. (Eds.) Evolving Sui Generis Options for the Hindu Kush-Himalayas. South Asian Watch on Trade, Economics and Environment, Modern Printing Press, Kathmandu, pp. 10-15.

Sharma, G., Acharya, B.K., 2013. Agriculture systems and management diversity. In: Gazetteer of Sikkim. Home Department, Government of Sikkim, Gangtok, India, pp. 225-259.

Sharma, G., Partap, U., Sharma, E., Rasul, G., Awasthe, R.K., 2016a. Agrobiodiversity in the Sikkim Himalaya: Socio-cultural significance, status, practices, and challenges. ICIMOD Working Paper 2016/5 Kathmandu: ICIMOD.

Sharma, G., Hunsdorfer, B., Singh, K.K., 2016b. Comparative analysis on the socio-ecological and economic potentials of traditional agroforestry systems in the Sikkim Himalaya. Trop. Ecol. 57(4), 751-764.

Sharma, K., Acharya, B.K., Sharma, B.K., Valente, D., Pasimeni, M.R., Petrosillo, I., Selvan, T., 2020. Land use effect on butterfly alpha and beta diversity in the Eastern Himalaya, India. Ecol. Indic. 110, 105605. DOI: 10.1016/j.ecolind.2019.105605

Sharma, L.N., Vetaas, O.R., 2015. Does agroforestry conserve trees? A comparison of tree species diversity between farmland and forest in mid-hills of central Himalaya. Biodivers. Conserv. 24, 2047-2061.

Sharma, S., Bhat, D., 2016. Migratory Pallas's Gull *Larus ichthyaetus* (Pallas, 1773): a new record from Sikkim, the eastern Himalaya, India. J. Threat. Taxa 8(12), 9449-9453.

Shooner, S., Davies, T.J., Saikia, P., Deka, J., Bharali, S., Tripathi, O.P., Singha, L., Khan, M.L., Dayanandan, S., 2018. Phylogenetic diversity patterns in Himalayan forests reveal evidence for environmental filtering of distinct lineages. Ecosphere 9(5), e02157.

Shrestha, B.R., Sharma, M., Magar, K.T., Gaudel, P., Gurung, M.B., Oli, B., 2018. Diversity and status of butterflies at different sacred forests of Kathmandu valley, Nepal. J. Entomol. Zool. Stud. 6(3), 1348-1356

Si, X., Baselga, A., Ding, P., 2015. Revealing Beta-Diversity Patterns of Breeding Bird and Lizard Communities on Inundated Land-Bridge Islands by Separating the Turnover and Nestedness Components. PLoS One 10(5), e0127692.

Si, X., Cadotte, M.W., Zeng, D., Baselga, A., Zhao, Y., Li, J., Wu, Y., Wang, S., Ding, P., 2017. Functional and phylogenetic structure of island bird communities. J. Anim. Ecol. 86(3), 532-542.

Sidhu, S., Shankar Raman, T.R., Goodale, E., 2010. Effects of plantations and home-gardens on tropical forest bird communities and mixed-species bird flocks in the southern Western Ghats. J. Bombay Nat. His. Soc. 107(2), 91-108.

Singh, A.P. 2009. Butterflies of Kedarnath Musk Deer Reserve, Garhwal Himalaya, India. J. Threat. Taxa 1(1), 37-48.

Singh, A.P., 2010. Woodpecker (Picidae) diversity in borer- *Hoplocerambyx spinicornis* infested sal *Shorea robusta* forests of Dehradun valley, lower western Himalaya. Indian BIRDS 6 (1), 2-11.

Singh, A.P., 2016. Moist temperate forest butterflies of western Bhutan. J. Threat. Taxa 8(3), 8596-8601.

Singh, A.P., 2017a. Records of Tibetan Blackbird *Turdus maximus*, and Himalayan Forest Thrush *Zoothera salimalii* from Pangolakha Wildlife Sanctuary, Sikkim, India. Indian BIRDS 12 (6), 168-169.

Singh, A.P., 2017b. Butterflies associated with major forest types in Arunachal Pradesh (Eastern Himalaya), India: Implications for ecotourism and conservation planning. J. Threat. Taxa 9(4), 10047-10075.

Singh, A.P., Pandey, R., 2004. A model for estimating butterflies species richness of areas across the Indian Sub-continent: species proportion of family Papilionidae as an indicator. J. Bom. Nat. Hist. Soc., 101(1), 79-89.

Singh, A.P., Gogoi, L., Sebastain, J., 2015. The seasonality of butterflies in a semi-evergreen forest: Gibbon Wildlife Sanctuary, Assam, northeastern India. J. Threat. Taxa 7(1), 6774-6787.

Singh, S., 1999. Assessing management effectiveness of wildlife protected areas in India. Parks 9(2), 34-49.

Sinu, P.A., 2011. Avian pest control in tea plantations of sub-Himalayan plains of Northeast India: Mixed-species foraging flock matters. Biol. Control 58, 362-366.

Sitters, H., York, A., Swan, M., Christie, F., Di Stefano, J., 2016a. Opposing responses of bird functional diversity to vegetation structural diversity in wet and dry forest. PloS ONE 11(10), e0164917.

Sitters, H., Di Stefano, J., Christie, F., Swan, M., York, A., 2016b. Bird functional diversity decreases with time since disturbance: Does patchy prescribed fire enhance ecosystem function?. Ecol. Appl. 26(1), 115-127.

Slancarova, J., Bartonova, A., Zapletal, M., Kotilinek, M., Faltynek Fric, Z., Micevski, N., Kati, V. Konvicka, M., 2016. Life history traits reflect changes in Mediterranean butterfly communities due to forest encroachment. PloS One, 11(3), e0152026.

Socolar, J.B., Gilroy, J.J., Kunin, W.E., Edwards, D.P. 2016. How should beta-diversity inform biodiversity conservation?. Trends Ecol. Evol. 31(1), 67-80.

Sodhi, N.S., Şekercioğlu, C.H., Barlow, J., Robinson, S.K., 2011. Conservation of Tropical Birds. John Wiley & Sons.

Soini, E., 2006. Bird diversity and land use on the slopes of Mt Kilimanjaro and the adjacent plains, Tanzania. Afr. Zool. 41(2), 193-209.

Soininen, J., Heino J., Wang, J., 2018. A meta-analysis of nestedness and turnover components of beta diversity across organisms and ecosystems. Glob. Ecol. Biogeogr. 27(1), 96-109.

Solomou, A., Sfougaris, A., 2011. Comparing conventional and organic olive groves in central Greece: plant and bird diversity and abundance.Renew. Agric. Food Syst. 26(04), 297-316.

Sreekar, R., Mohan, A., Das, S., Agarwal, P., Vivek, R., 2013. Natural windbreaks sustain bird diversity in a tea-dominated landscape. PLoS One 8, e70379.

Sreekar, R., Srinivasan, U., Mammides, C., Chen, J., Goodale, U.M., Kotagama, S.W., Sidhu, S., Goodale, E., 2015. The effect of land-use on the diversity and mass-abundance relationships of understory avian insectivores in Sri Lanka and southern India. Sci. Rep. 5, 11569. DOI: 10.1038/srep11569

Sreekar, R., Katabuchi, M., Nakamura, A., Corlett, R.T., Slik, J.F., Fletcher, C., He, F., Weiblen, G.D., Shen, G., Xu, H. Sun, I.F., 2018. Spatial scale changes the relationship between beta diversity, species richness and latitude. R. Soc. Open Sci.5(9), 181168. DOI: 10.1098/rsop.181168

Sreekar, R., Koh, L.P., Mammides, C., Corlett, R.T., Dayananda, S., Goodale, U.M., Kotagama, S.W. Goodale, E., 2020. Drivers of bird beta diversity in the Western Ghats–Sri Lanka biodiversity hotspot are scale dependent: roles of land use, climate, and distance. Oecologia 193, 801-809.

Srinivasan, U., Elsen, P.R., Wilcove, D.S., 2019. Annual temperature variation influences the vulnerability of montane bird communities to land-use change. Ecography 42, 1-11.

Stattersfield, A.J., Crosby, M., Long, M.J., Wegge, D.C., 1998. Endemic Bird Area of the World: Priorities for biodiversity conservation. Birdlife International, Cambridge, UK.

Steffan-Dewenter, I., Tscharntke, T., 2000. Butterfly community structure in fragmented habitats. Ecol. Lett. 3(5), 449-456.

Steffan-Dewenter, I., Munzenberg, U., Burger, C., Thies, C., Tscharntke, T., 2002. Scale dependent effects of landscape context on three pollinator guilds. Ecology 83, 1421-1432.

Steven, H., 1923. Notes on the Birds of the Sikkim, Himalayas. J. Bomb. Nat. Hist. Soc. 29(2), 503-518.

Steven, H., 1925. Notes on the Birds of the Sikkim, Himalayas. J. Bomb. Nat. Hist. Soc.30(4),872-893.

Stuhldrehera, G., Fartmann, T., 2018. Threatened grassland butterflies as indicators of microclimatic niches along an elevational gradient–Implications for conservation in times of climate change. Ecol. Indic. 94, 83-98.

Subasinghe, K., Sumanapala, A.P., Weerawardhena, S.R., 2014. The impact of forest conversion on bird communities in the northern flank of the Knuckles Mountain Forest Range, Sri Lanka. J. Asia-Pac. Biodivers. 7(4), 367-373.

Subramanya, S., Radhamani, T.R., 1993. Pollination by birds and bats. Curr. Sci. 65(3), 201-209.

Surya, G.S., Keitt, T.H., 2019, Altitudinal limits of Eastern Himalayan birds are created by competition past and present. PLoS One 14(7), e0217549.

Sweaney, N., Lindenmayer, D.B., Driscoll, D.A., 2014. Is the matrix important to butterflies in fragmented landscapes? J. Insect Conserv. 18, 283-294.

Syiem, B.L.N., Goswami, V.R., Vasudev, D., 2018. “In a tree by the brook, there’s a songbird who sings”: Woodlands in an agricultural matrix maintain functionality of a wintering bird community. PloS One13(8), e0201657.

Tabashnik, B.E., 1982. Responses of pest and non-pest *Colias* butterfly larvae to intraspecific variation in leaf nitrogen and water content. Oecologia, 55(3), 389-394.

Takeuchi, M., 2019. Winter bird communities in the heterogeneous farmlands of the Aso region in Japan. J. Asia-Pac. Biodivers. 12(2), 152-159.

Tamang, S.R., Joshi, A., Pandey, J., Raut, N., Shrestha, B.R., 2019. Diversity of butterflies in eastern lowlands of Nepal. The Himalayan Naturalist, 2(1), 3-10.

Tanalgo, K.C., Pineda, J.A.F., Agravante, M.E., Amerol, Z.M., 2015. Bird diversity and structure in different land-use types in lowland south-central Mindanao, Philippines. Trop. Life Sci. Res. 26(2), 85-103.

Tejeda-Cruz, C., Sutherland, W.J., 2004. Bird responses to shade coffee production. Anim. Conserv. 7(2), 169-179.

Thinh, V.T., 2006. Bird species richness and diversity in relation to vegetation in Bavi National Park, Vietnam. Ornithol. Sci. 5(1), 121-125.

Thiollay J.M., 1995. The role of traditional agroforests in the conservation of rain forest bird diversity in Sumatra. Conserv. Biol. 9, 335-353

Thiollay, J.M. 1997. Disturbance, selective logging and bird diversity: a Neotropical forest study. Biodivers. Conserv. 6(8), 1155-1173.

Thiollay, J.M., 1999. Responses of an avian community to rain forest degradation.Biodivers. Conserv. 8, 513-534.

Thomas, J.A., 2005. Monitoring change in the abundance and distribution of insects using butterflies and other indicator groups. Philos. Trans. R. Society B: Biol. Sci. 360, 339-357.

Tilman, D., 2001. Functional diversity. Encyclopedia of biodiversity, 3(1), 109-120.

Tinoco, B.A., Santillán, V.E., Graham, C.H., 2018. Land use change has stronger effects on functional diversity than taxonomic diversity in tropical Andean hummingbirds. Ecol. Evol. 8(6), 3478-3490.

Tsai, C.F., Lee, Y.F., Chen, Y.H., Chen, W.M., Kuo, Y.M., 2015. Species turnover in tropical montane forest avifauna links to climatic correlates. Glob. Ecol. Conserv. 3, 541-552.

Tscharntke T., Klein, A.M., Kruess, A., Steffan-Dewenter, I., Thies, C., 2005. Landscape perspectives on agricultural intensification and biodiversity-ecosystem service management. Ecol. Lett. 8, 857-874.

Tscharntke, T., Şekercioğlu, C.H., Dietsch, T.V., Sodhi, N.S., Hoehn, P., Tylianakis, J.M., 2008. Landscape constraints on functional diversity of birds and insects in tropical agro-ecosystems. Ecology 89, 944-951.

Tscharntke, T., Clough, Y., Wanger, T. C., Jackson, L., Motzke, I., Perfecto, I., Vandermeer,J., Whitbread, A., 2012. Global food security, biodiversity conservation and the future of agricultural intensification. Biol. Conserv. 151, 53-59.

Tuck, S.L., Winqvist, C., Mota, F., Ahnstrom, J., Turnbull, L.A., Bengtsson, J., 2014. Land-use intensity and the effects of organic farming on biodiversity: a hierarchical meta-analysis. J. Appl. Ecol. 51, 746-755.

Uchida, K., Ushimaru, A., 2015. Land abandonment and intensification diminish spatial and temporal β-diversity of grassland plants and herbivorous insects within paddy terraces. J. Appl. Ecol. 52(4), 1033-1043.

Uehara-Prado, M., Brown Jr, K.S., Freitas, A.V.L., 2007. Species richness, composition and abundance of fruit-feeding butterflies in the Brazilian Atlantic Forest: comparison between a fragmented and a continuous landscape. Glob. Ecol. Biogeogr. 16(1), 43-54.

UNEP-WCMC (Comps.), 2018. The Checklist of CITES Species Website. CITES Secretariat, Geneva, Switzerland. Compiled by UNEP-WCMC, Cambridge, UK. https://checklist.cites.org. (Accessed on 26 December 2018).

Uniyal, V.P., 2004. Butterflies of Nanda Devi National Park-a world heritage site. Ind. For. 130(7), 800-804.

Uniyal, V.P., 2007. Butterflies in the Great Himalayan conservation landscape, Himachal Pradesh, Western Himalaya. Entomon 32, 119-127.

Uniyal, V.P., Mathur, P.K., 1998. Diversity of butterflies in the great Himalayan national park, Western Himalaya. Ind. J. For. 21(2), 150-155.

Van Bael, S.A., Bichier, P., Ochoa, I., Greenberg, R., 2007. Bird diversity in cacao farms and forest fragments of western Panama. Biodivers. Conserv. 16(8), 2245-2256.

van der Poel, P., 2013. Results of Rapid Biodiversity Survey in Bumdeling Wildlife Sanctuary, Technical Report, p. 37.

Van Halder, I., Thierry, M., Villemey, A., Ouin, A., Archaux, F., Barbaro, L., Balent, G. and Benot, M.L., 2017. Trait-driven responses of grassland butterflies to habitat quality and matrix composition in mosaic agricultural landscapes. Insect Conserv. Divers. 10(1), 64-77.

Vandermeer, J., Perfecto, I., 2007. The agricultural matrix and a future paradigm for conservation. Conserv. Biol. 21, 274-277.

Vanitha, V., Kumar, C., Thiyagesan, K., 2014. Roost and diet selection by Southern Spotted Owlet *Athene brama brama* (Temminck, 1821) in the Cauvery Delta of Nagapattinam District, southern India. J. Threat. Taxa 6(6), 5845-5850.

Varshney, R.K., Smetacek, P. (Eds.), 2015. A synoptic Catalogue of the Butterflies of India, p. 255.

Velazco, S.J.E., Villalobos, F., Galvão, F. and De Marco Júnior, P., 2019. A dark scenario for Cerrado plant species: Effects of future climate, land use and protected areas ineffectiveness. Divers. Distrib. 25(4), 660-673.

Villéger, S., Mason, N.W.H. Mouillot, D., 2008. New multidimensional functional diversity indices for a multifaceted framework in functional ecology. Ecology 89, 2290-2301.

Villemey, A., Van Halder, I., Ouin, A., Barbaro, L., Chenot, J., Tessier, P., Calatayud, F., Hilaire Martin, H., Roche, P., Archaux, F., 2015. Mosaic of grasslands and woodlands is more effective than habitat connectivity to conserve butterflies in French farmland. Biol. Conserv. 191, 206-215.

Vu, L.V., 2009. Diversity and similarity of butterfly communities in five different habitat types at Tam Dao National Park, Vietnam. Journal of Zoology 277, 15-22.

Vu, L.V., 2015. Diversity of butterfly communities in different habitats in limestone tropical rain forest of Vietnam. Russian Entomol. J. 24(3), 1-10.

Vu, L.V., Bonebrake, T.C., Vu, M.Q., Nguyen, N.T., 2015. Butterfly diversity and habitat variation in a disturbed forest in northern Vietnam. Pan-Pac. Entomol. 91(1), 29-39.

Vu, V.L., 2007. Ecological indicator role of butterflies in Tam Dao National Park. Russ. Entomol. J. 16(4), 479-486.

Warren-Thomas, E., Nelson, L., Juthong, W., Bumrungsri, S., Brattström, O., Stroesser, L., Chambon, B., Penot, É., Tongkaemkaew, U., Edwards, D.P., Dolman, P.M., 2020. Rubber agroforestry in Thailand provides some biodiversity benefits without reducing yields. J. Appl. Ecol. 57(1), 17-30.

Waltert, M., Mardiastuti, A., Mühlenberg, M., 2004. Effects of land use on bird species richness in Sulawesi, Indonesia. Conserv. Biol. 18(5), 1339-1346.

Waltert, M., Bobo, K.S., Sainge, N.M., Fermon, H., Mühlenberg, M., 2005. From forest to farmland: habitat effects on Afrotropical forest bird diversity. Ecol. Appl. 15(4), 1351-1366.

Waltert, M., Serge, K.B., Kaupa, S., Montoya, M.L., Nsanyi, M.S., Fermon, H., 2011. Assessing conservation values: biodiversity and endemicity in tropical land use systems. PLoS One 6, e16238.

Watson, J.E.M., Dudley, N., Segan, D.B., Hockings, M., 2014. The performance and potential of protected areas. Nature 515, 67-73.

Wenny, D.G., Şekercioğlu, C.H., Cordeiro, N.J., Rogers, H.S., Kelly, D., 2016. Seed dispersal by fruit-eating birds. In: Şekercioğlu, Ç.H., Wenny, D.G., Whelan, C.J. (Eds.) Why Birds Matter: Avian Ecological Function and Ecosystem Services. University of Chicago Press, pp. 107-145.

Weyland, F., Baudry, J., Ghersa, C.M., 2014. Rolling Pampas agroecosystem: which landscape attributes are relevant for determining bird distributions?. Rev. Chil. de Hist. Nat. 87(1), 1-12.

Whelan, C.J., Şekercioğlu, Ç.H., Wenny, D.G., 2015. Why birds matter: from economic ornithology to ecosystem services. J. Ornithol. 156(1), 227-238.

White, P.J., Kerr, J.T., 2007. Human impacts on environment–diversity relationships: evidence for biotic homogenization from butterfly species richness patterns. Glob. Ecol. Biogeogr. 16(3), 290-299.

Winqvist, C., Bengtsson, J., Aavik, T., Berendse, F., Clement, L. W., Eggers, S., Fischer, C., Flohre, A., Geiger, F., Liira, J., Part, T., Thies, C., Tscharntke, T., Weisser, W.W., Bommarco, R., 2011. Mixed effects of organic farming and landscape complexity on farmland biodiversity and biological control potential across Europe. J. Appl. Ecol. 48, 570-579.

World Bank, 2019. Agricultural land (% of land area). https://data.worldbank.org/indicator/AG.LND.AGRI.ZS.

Wynter-Blyth, M.A., 1957. Butterflies of the Indian Region, Bombay Natural History Society, Mumbai, India.

Yabuhara, Y., Yamaura, Y., Akasaka, T., Yamanaka, S., Nakamura, F., 2019. Seasonal variation in patch and landscape effects on forest bird communities in a lowland fragmented landscape. For. Ecol. Manage. 454, 117-140.

Yashmita-Ulman, Sharma, M., Kumar, A., 2016. Agroforestry systems as habitat for avian species: assessing its role in conservation. Proc. Zool. Soc. 71, 127-145.

Zellweger, F., Roth, T., Bugmann, H., Bollmann, K., 2017. Beta diversity of plants, birds and butterflies is closely associated with climate and habitat structure. Glob. Ecol. Biogeogr. 26(8), 898-906.

Zhang, M., Chang, C., Quan, R., 2017a. Natural forest at landscape scale is most important for bird conservation in rubber plantation. Biol. Conserv. 210, 243-252.

Zhang, M.X., Kyaw, M., Li, G.G., Zhao, J.B., Zeng, X.L., Swa, K., Quan, R.C., 2017b. Bird diversity in northern Myanmar and conservation implications. Zoo. Res. 38(5), 264-280.

Zhang, Q., Holyoak, M., Chen, C., Liu, Z., Liu, J., Che, X., Dong, A., Yang, C., Zou, F., 2020a. Trait-mediated filtering drives contrasting patterns of species richness and functional diversity across montane bird assemblages. J. Biogeogr. 47(1), 301-312.

Zhang, Q., Holyoak, M., Goodale, E., Liu, Z., Shen, Y., Liu, J., Zhang, M., Dong, A., Zou, F., 2020b. Trait–environment relationships differ between mixed-species flocking and nonflocking bird assemblages. Ecology 101(10), e03124.

Zhijun, W., Young, S.S., 2003. Differences in bird diversity between two swidden agricultural sites in mountainous terrain, Xishuangbanna, Yunnan, China. Biol. Conserv. 110(2), 231-243.

Zingg, S., Grenz, J., Humbert, J.Y., 2018. Landscape-scale effects of land use intensity on birds and butterflies. Agric. Ecosyst. Environ. 267, 119-128.

Zomer, R.J., Ustin, S.L., Carpenter, C.C., 2001. Land cover change along tropical and subtropical riparian corridors within the Makalu Barun National Park and Conservation Area, Nepal. Mt. Res. Dev. 21, 175-183.

Zou, Y., Xiao, H., Bianchi, F.J.J.A., Jauker, F., Luo, S., van der Werf, W., 2017. Wild pollinators enhance oilseed rape yield in small-holder farming systems in China. BMC Ecol. 17(1), 1-7.

PUBLICATIONS AND PRESENTATIONS

Publications:

Sharma, K., Acharya, B.K., Sharma, B.K., Valente, D., Pasimeni, M.R., Petrosillo, I., Selvan, T. 2020. Land use effect on butterfly alpha and beta diversity in the Eastern Himalaya, India. *Ecological Indicators* 110, 105605. (ISSN: 1470 160X).

Dewan. S., Chettri, I.K., **Sharma, K.**, Acharya, B.K. 2019. Kitam Bird Sanctuary, the only low elevation protected area of Sikkim: A conservation hotspot for butterflies in the Eastern Himalaya. *Journal of Asia-Pacific Entomology* 22, 575-583. (ISSN: 12268615).

Chettri, P.K., **Sharma, K.**, Dewan. S., Acharya, B.K. 2018. Butterfly diversity in human modified ecosystems of southern Sikkim, the eastern Himalaya, India. *Journal of Threatened Taxa* 10(5), 11551-11565 (ISSN: 0974 7907)

Sharma, K., Acharya, B.K., Sharma, G. 2018. Psychophily and prey-predator interactions in Parijat *Nyctanthes arbor-tristis* – A cultural keystone species. PANDA ENVIS Sikkim Newsletter 10(4), 52-59.

Presentations at National and International Conferences:

Sharma, K., Acharya, B.K., Sharma G. High bird diversity in organic and traditionally managed agroecosystems of Eastern Himalaya. Oral presentation at International Conference on “Chemical Ecology, Environment and Human Health: Emerging Frontiers and Synthesis (ICCEEHH, 2019)” organized by Department of Zoology, School of Life Sciences, Sikkim University during August 9-10, 2019.

Sharma, K., Acharya, B.K., Sharma G. Butterfly communities shows contrasting patterns of alpha and beta diversity along agroecosystem-forest gradient of the Sikkim Himalaya, India. Poster presentation at National Conference on “Recent Trends in Biological Research and Future Prospects” organized by Department of Zoology, School of Life Sciences, Sikkim University during May 28-29, 2018.

Ecological Indicators 110 (2020) 105605

Contents lists available at ScienceDirect

Ecological Indicators

journal homepage: www.elsevier.com/locate/ecolind

Land use effect on butterfly alpha and beta diversity in the Eastern Himalaya, India

Kishor Sharma[a], Bhoj Kumar Acharya[a,*], Ghanashyam Sharma[b], Donatella Valente[c], Maria Rita Pasimeni[c], Irene Petrosillo[c], Thiru Selvan[d]

[a] Department of Zoology, School of Life Sciences, Sikkim University, Tadong, Gangtok, Sikkim 737102, India
[b] The Mountain Institute-India, Tadong, Gangtok, Sikkim 737102, India
[c] Laboratory of Landscape Ecology, Department of Biological and Environmental Sciences and Technologies, University of Salento, Lecce, Italy
[d] Department of Forestry and Biodiversity, Tripura University, Suryamaninagar, Agartala, Tripura, India

ARTICLE INFO

Keywords:
Agroecosystems
Ecosystem services
Forests
Human-modified landscapes
Indicator species
Traditional management practices

ABSTRACT

Conservation of biodiversity in agroecosystems is a global challenge as conversion of forest to agroecosystems has been one of the major causes for biodiversity loss through habitat transformation. The agroecosystems, especially those traditionally managed or organic, are reported to retain high biodiversity including endemic, specialists and conservation concern species. Among others, butterflies are the most vulnerable taxa reacting sensitively and rapidly to climate and habitat changes, and represent as bio-indicators to predict the health of an ecosystem. However, the assessment of land use effect on butterfly diversity has not yet been undertaken in the Eastern Himalayas. Therefore, this study was designed along agroecosystem-forest gradient to understand: the patterns of butterfly alpha diversity taking into account the variation across seasons, elevation, forest specialization and larval host specificity; the patterns of butterfly beta diversity; and plausible environmental determinants of butterfly alpha and beta diversity. We assessed the patterns of taxonomic alpha and beta diversity of butterflies and their determinants in the Indigenous Farming Systems (IFS) {large cardamom-based agroforestry systems (LCAS), mandarin orange-based agroforestry systems (MOAS) and farm-based agroforestry systems (FAS)} along with adjacent natural forests (Forests) in the study area during December 2012–August 2017. We recorded 268 species of butterflies from six families which included two-third forest specialists, one-third monophagous and one-fifth conservation concern species. Along the agroecosystem-forest gradient, alpha diversity of butterflies declined for total, forest specialists, monophagous and protected species. However, pairwise beta diversity increased and the multiple beta diversity was dominated by substitution components. Alpha diversity was determined by tree species richness, tree density, canopy cover, elevation, mean annual precipitation (MAP), season, whereas tree species richness, tree density, tree basal area, canopy cover, elevation, mean annual temperature, and MAP influenced beta diversity. We also identified 15 indicator species dominated by forest specialists suitable for long term ecological monitoring program in the Eastern Himalaya. The organic and traditionally managed agroecosystems of Sikkim play a complementary role to the protected areas (PAs) in fostering biodiversity conservation and ecosystem service provision, especially in the areas with high human pressure and low PA and forest coverage.

1. Introduction

Climate change, agricultural intensification, global environmental changes, such as the habitat loss and fragmentation, results in negative consequences for biodiversity and natural capital (Chapin et al., 2000; Foley et al., 2011). Conservation of biodiversity in agroecosystems is a global concern (Vandermeer and Perfecto, 2007; Flohre et al., 2011) as conversion of forest to agroecosystems has been one of the major causes for biodiversity loss through habitat transformation in terms of land-cover changes (Gibbs et al., 2010), landscape simplification and fragmentation (Ekroos et al., 2010; Börschig et al., 2013; Loos et al., 2014; Dainese et al., 2017) associated with rapid land-use intensification (Matson et al., 1997, Tscharntke et al., 2005; Billeter et al., 2008; Karp et al., 2012) and change (Foley et al., 2005, Newbold et al., 2015). The shift in research interest from assessment of biodiversity towards the concept of ecosystem services in the context of social-ecological systems

---

[*] Corresponding author.
E-mail address: bkacharya@cus.ac.in (B.K. Acharya).

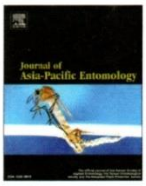
https://doi.org/10.1016/j.ecolind.2019.105605
Received 6 May 2019; Received in revised form 26 July 2019; Accepted 30 July 2019

Journal of Asia-Pacific Entomology 22 (2019) 575–583

Contents lists available at ScienceDirect

Journal of Asia-Pacific Entomology

journal homepage: www.elsevier.com/locate/jape



Full length article

Kitam Bird Sanctuary, the only low elevation protected area of Sikkim: A conservation hotspot for butterflies in the Eastern Himalaya

Sailendra Dewan, Iswar Kumar Chettri, Kishor Sharma, Bhoj Kumar Acharya*

Department of Zoology, School of Life Sciences, Sikkim University, Tadong, Gangtok 737102, East Sikkim, India

ARTICLE INFO

Keywords:
Butterflies
Conservation
Generalist
Habitat preference
Specialist
Tropical forests

ABSTRACT

The lowland areas of the Himalayan region are subjected to immense anthropogenic pressure because of least representation in the protected area network. Kitam Bird Sanctuary is the only representative protected area that occurs below 1000 m in Sikkim state of India (a part of globally significant biodiversity hotspot of Himalayas) and serves as the refuge for various species of flora and fauna. Here we studied butterfly diversity and community composition in Kitam Bird Sanctuary (a small protected area of 6 km^2 geographical area) following point count method spread across predetermined transects. Altogether 1674 butterflies belonging to 111 species and six families were recorded after completion of 240 point counts. Among these, 18 species are federally protected under the Wildlife (Protection) Act (1972) of India. Most of the butterflies were forest specialist in terms of habitat preference, whereas based on host plant specificity, the butterfly community was mostly dominated by generalist feeder (Oligophagous II and Polyphagous). Butterfly community parameters showed a strong correlation with habitat variables. While Kitam Bird Sanctuary is primarily designated for conservation of lowland birds, the high diversity of butterflies both in terms of taxonomic richness and trait composition suggests that the sanctuary harbors an ideal habitat for butterflies of the tropical region and invites conservation attention.

Introduction

Habitat loss remains the greatest threat to biodiversity, especially in tropical regions (Laurance, 1999; Brooks et al., 2002; Primack, 2014). The loss and degradation of habitat have been identified as the main threat to 85% of IUCN Red List species (IUCN, 2015). Butterflies are highly sensitive to alteration in habitat structure due to their exclusive dependency and specificity to plants, both as larval host as well as nectars (Kremen, 1992; Blair and Launer, 1997). Increasing land use intensities leading to degradation and fragmentation of habitats causes a decline in the diversity of butterflies (Schulze et al., 2004; Posa and Sodhi, 2006; Francesconi et al., 2013).

Biodiversity conservation measures around the globe have been focused mainly on the establishment of protected areas (Burner et al., 2001). Studies have shown that protected areas are rich in biodiversity, including butterflies, as compared to unprotected forests (Akwashiki et al., 2007; Chinaru and Joseph, 2011; Klorvuttimontara et al., 2011). The occurrence of complex vegetation structure which creates an array of micro-habitats has been attributed to the high diversity of butterflies in protected areas (Orimaye et al., 2016). Butterflies complementarily are a good indicator of forest health and serves as the surrogate taxa for biodiversity wealth (Bhardwaj et al., 2012; Hayes et al., 2009). Hence, assessment of butterfly diversity serves as a basis for evaluating the status of protected areas (Majumder et al., 2012).

Besides taxonomic composition based indices (e.g species richness and Shannon-Weiner diversity), life history traits are used to measure biodiversity in recent years (Slancarova et al., 2016; Lee and Martin, 2017). Traits strongly influence functioning of a species in an ecosystem and reflects how species respond to environmental conditions (Petchey and Gaston, 2002; Dreiss et al., 2015; Aguirre-Gutiérrez et al., 2017). In case of lepidopterans, difference in life history traits such as host plant association and habitat specialization are responsible for differential sensitivities to habitat loss (Summerville and Crist, 2004; Soga and Koike, 2012). Generalist species increases with the land use intensification, whereas specialist group displays the opposite trend (Börschig et al., 2013; Kitahara et al., 2000).

The Himalayan state of Sikkim, located in the Eastern Himalayan mountain in India, consists of around 689 species of butterflies representing approximately 50% species found in the Indian sub-continent (Haribal, 1992; Kehimkar, 2016). The continuous gradation of climate and changes in vegetation structure due to a steep rise in elevation offers a unique habitat for the existence of wide diversity of butterflies in the region. The lowland tropical valleys in Sikkim, especially below 1000 m, have been considered as hotspot for butterflies

---

* Corresponding author.
E-mail address: bkacharya@cus.ac.in (B.K. Acharya).

https://doi.org/10.1016/j.aspen.2019.04.002
Received 28 December 2018; Received in revised form 28 March 2019; Accepted 1 April 2019

Journal of Threatened Taxa | www.threatenedtaxa.org | 26 April 2018 | 10(5): 11551–11565

ARTICLE

BUTTERFLY DIVERSITY IN HUMAN-MODIFIED ECOSYSTEMS OF SOUTHERN SIKKIM, THE EASTERN HIMALAYA, INDIA

ISSN 0974-7907 (Online)
ISSN 0974-7893 (Print)

OPEN ACCESS

Prem Kumar Chettri [1], **Kishor Sharma** [2], **Sailendra Dewan** [3] & **Bhoj Kumar Acharya** [4]

[1,2,3,4] Department of Zoology, School of Life Sciences, Sikkim University, Tadong, Gangtok, Sikkim 737102, India
[1] Present address: Forest, Environment and Wildlife Management Department, Government of Sikkim, Deorali, Sikkim 737102, India
[1] chettriprem22@gmail.com, [2] kisarma@gmail.com, [3] dewansailendra1992@gmail.com, [4] bkacharya@cus.ac.in (corresponding author)

Abstract: Understanding wild biodiversity of agroecosystems and other human dominated landscapes are crucial for the management and conservation of biological resources. Here, we studied the diversity, abundance, similarity and functionality of butterflies in different human modified ecosystems in southern Sikkim, the Eastern Himalaya. The study was conducted from January 2015 to May 2015 by covering three habitat types namely, farm-based agroforestry, large cardamom-based agroforestry and adjacent natural forest ecosystem. We followed point count method along the transect to collect data on butterflies in the study area. A total of 911 individual butterflies representing six families and 44 species were recorded during the present study in southern Sikkim. Species richness and abundances of butterflies were significantly different among the systems. While diversity and abundance were higher in forest patches, each system harbored unique species assemblages with low similarity between habitats. The information on larval host plants were available for 41 butterfly species which depended on 128 plant species belonging to 27 families. The butterfly community was dominated by oligophagous II (19 species) followed by polyphagous (11 species), monophagous (8 species) and oligophagous I (3 species). Similarly, generalist feeders had higher species and abundance compared to specialist feeders. Specialist species were confined to forest habitat, whereas generalist species were mostly restricted to cultivated systems. The findings of the study highlighted the need for conservation of traditionally managed agroecosystems in order to conserve butterflies and other associated biodiversity.

Keywords: Agroecosystems, butterfly, conservation, eastern Himalaya; host plants.

DOI: http://doi.org/10.11609/jott.3641.10.5.11551-11565 | **ZooBank:** urn:lsid:zoobank.org:pub:3C98090F-8CB7-40D4-9592-A37C23BA1E37

Editor: Sanjay Sondhi, Titli Trust, Dehradun, India. **Date of publication:** 26 April 2018 (online & print)

Manuscript details: Ms # 3641 | Received 08 July 2017 | Final received 23 March 2018 | Finally accepted 04 April 2018

Citation: Chettri, P.K., K. Sharma, S. Dewan & B.K. Acharya (2018). Butterfly diversity in human-modified ecosystems of southern Sikkim, the eastern Himalaya, India. *Journal of Threatened Taxa* 10(5): 11551–11565; http://doi.org/10.11609/jott.3641.10.5.11551-11565

Funding: This manuscript is an outcome of the M.Sc. dissertation of the first author.

Competing interests: The authors declare no competing interests.

Author Details: Prem Kumar Chettri holds MSc (Zoology) from Sikkim University and currently works at the Forest, Environment and Wildlife Management Department, Government of Sikkim. He has a keen interest on biodiversity of Sikkim Himalaya with special focus on butterflies and plants. Kishor Sharma is a PhD scholar at the Department of Zoology, Sikkim University. His research interests are to understand the diversity and distribution patterns of birds and butterflies in the agroecosystem-forest gradient of Sikkim Himalaya. Sailendra Dewan is a PhD scholar at the Department of Zoology, Sikkim University. His research focuses on distribution pattern and phylogeny of butterflies along elevation gradients in the Sikkim Himalaya. Bhoj Kumar Acharya is a faculty in the Department of Zoology, Sikkim University, Gangtok. His research interests are to understand the large-scale ecological patterns with particular emphasis on species richness, abundance and distribution patterns along ecological gradients in the Himalaya.

Author Contribution: Study designed by BKA, PKC, KS; data collected by PKC, KS; analyzed the data by KS, PKC, SD, BKA and PKC, KS, SD, BKA wrote the manuscript.

Acknowledgements: We thank Vice Chancellor, Sikkim University, Dean, School of Life Sciences and Head, Department of Zoology, Sikkim University for facilities to undertake this research, and faculty members of Zoology Department, Sikkim University for cooperation and insightful discussion. We also thank PCCF cum Secretary, Chief Wildlife Warden and Principal Research Officer, Forests, Environment and Wildlife Management Department, Government of Sikkim for granting permission to carry out the study. Support received from local communities of Ralong Village is highly appreciated.

Psychophily and prey-predator interactions in Parijat *Nyctanthes arbor-tristis* - A cultural keystone species

KISHOR SHARMA[1], BHOJ KUMAR ACHARYA[2], GHANASHYAM SHARMA[3]

[1,2] Department of Zoology, School of Life Sciences, Sikkim University, Tadong, Gangtok, Sikkim 737102, India
[3] The Mountain Institute-India, Tadong, Gangtok, Sikkim 737102, India
[1] kisarma@gmail.com, [2]bkacharya@cus.ac.in, [3]banstolag@yahoo.co.in

Butterfly pollination

Almost 87.5% species of the world's estimated flowering plants and more than three-fourth of the global food crops depends on animal pollination (Zoophily) (Klein et al. 2007; Ollerton et al. 2011). Wild pollinators other than honey bees have recently been recognized for their role in increasing and stabilizing crop-pollination services against land use change (Rader et al. 2016). Pollination by butterflies (Psychophily) has been reported to be important in both natural forests and human-modified ecosystems, because of their strong flying capacity and ability to move pollens to long distances (Schmitt 1980; Andersson et al. 2002; Donoso et al. 2016; Jain et al. 2016). Butterflies usually depends on olfactory/chemical signal to pollinate plants having white/dull coloured, strongly scented flowers (Balasubramanian 1990; Andersson et al. 2002), whereas they depend on visual signal to pollinate plants with brightly coloured, faintly scented flowers (Borges et al. 2003). Most of the Lepidopterans depend on angiosperms for host plants, and for floral nectar (source of sugars and amino acids) at larval and adult stages, respectively (Corlett 2004).

Spider predation on butterflies

Spider predation on butterflies is a common phenomenon in nature. However, some butterflies have developed defense mechanism to successfully escape predation by spiders through both chemical defense as in Danainae and Ithomiinae butterflies (Nymphalidae family) by acquiring/sequestering Pyrrolizidine alkaloids (PAS) through pharmacophagy of plants of Apocynaceae and Solanaceae families (Sourakovet al. 1996) and also by mimicry through false heads as in hairstreak butterfly (Lycaenidae family) (Sourakov 2013).

Parijat (*Nyctanthes arbor-tristis*)

Parijat (*Nyctanthes arbor-tristis)* belonging to Oleaceae family is an important and sacred plant of India with great values in local tradition, culture, traditional medicine systems, and religious practices. It is believed that the plant relieves tiredness and the fairy (*apsara*) from heaven used to touch the plants to get relief from stress and tiredness. It is also called as night jasmine or flower of sorrow. The meaning of the scientific name itself is 'sad tree'. The sadness is mostly because the flowers develop during night and falls down before sunrise and go away from the plants or flowers lose their brightness during day time.

There are many legends associated with this tree and flowers. According to Hindu mythology parijat flower is associated with love story of Lord Krishna and his wife Rukmini, and the jealousy of another wife Satyabhama. The story goes like this- The Narada *Rishi* (sage) brought some flowers of parijat from *Indralok* (land of the king of gods) and went to meet Lord Krishna. Lord Krishna presented those flowers to Rukmini but Narada immediately went to meet Satyabhama and told her that all the beautiful flowers were given to Rukmini alone and no more left for her. Satyabhama was so jealous, hence, she asked Lord Krishna to bring entire tree with flowers for her. Since the Parijat was available only in heaven, lord Krishna along with Satyabhama went to *Indralok* and requested Indra (king of gods) for the same. While King Indra refused to give it because Narada had already alerted him, lord Krishna took entire tree with flowers to earth as he defeated Indra in confrontation. Conceding defeat Indra said- anyway take the tree but you will not be able to see the flowers during the day. In order to satisfy both of his wives, tree was cleverly planted by Lord Krishna in Satyabhama's garden in such a manner that all the flowers fell in Rukmini's garden, with whom he loved much.

Certificate of Participation

This is to certify that

Kishor Sharma

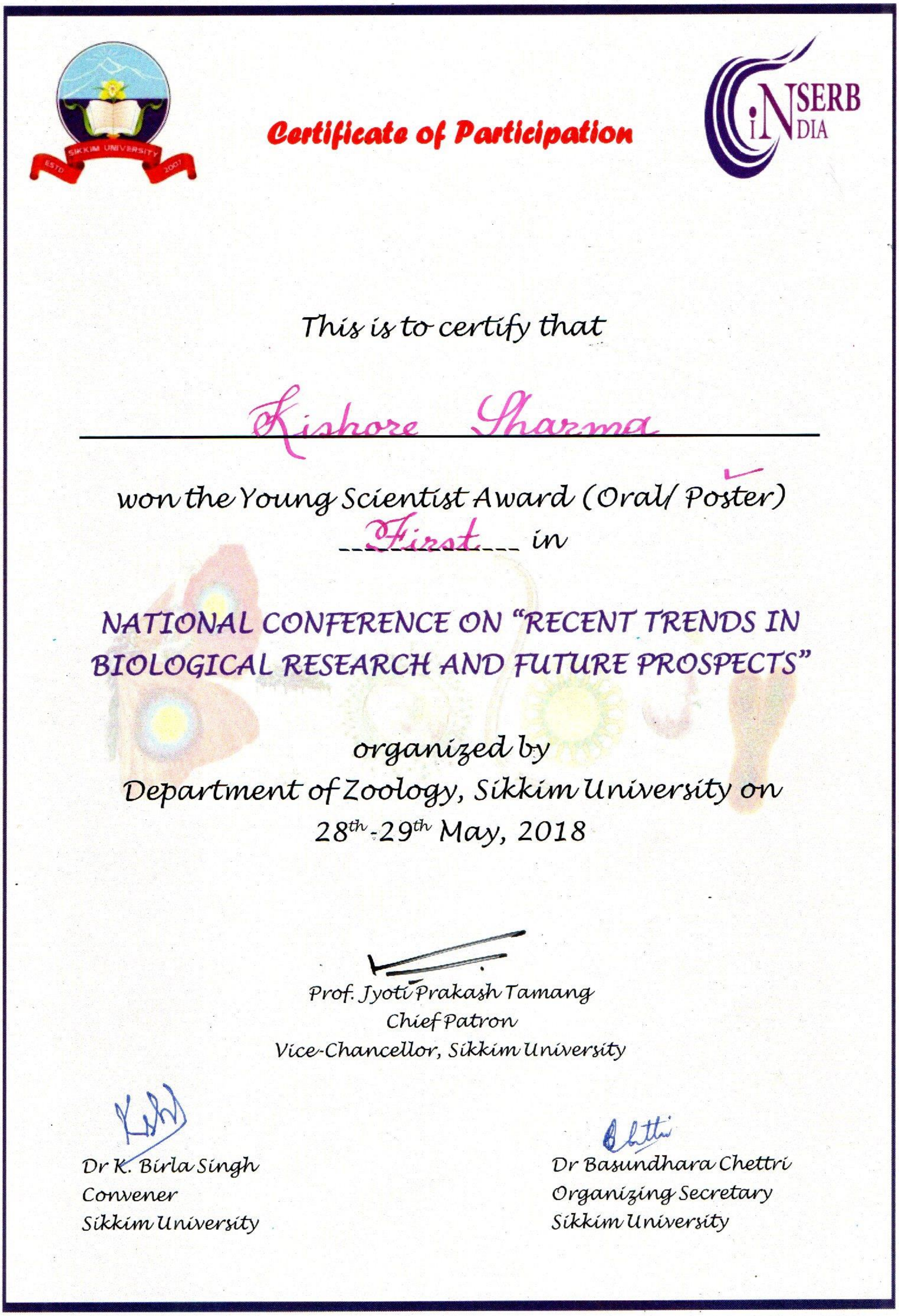
✓ delivered a talk/ presented a paper/ participated in

INTERNATIONAL CONFERENCE ON "CHEMICAL ECOLOGY, ENVIROMENT AND HUMAN HEALTH: EMERGING FRONTIERS AND SYNTHESIS (ICCEEHH 2019)"

Organized by
Department of Zoology, Sikkim University on
August 9-10, 2019

Professor Avinash Khare
Chief Patron
Vice-Chancellor, Sikkim University

Dr. K. Birla Singh
Convener
Sikkim University



SERB INDIA

Certificate of Participation

This is to certify that

Kishore Sharma

won the Young Scientist Award (Oral/ [✓ Poster])

First in

NATIONAL CONFERENCE ON "RECENT TRENDS IN BIOLOGICAL RESEARCH AND FUTURE PROSPECTS"

organized by
Department of Zoology, Sikkim University on
28th-29th May, 2018

Prof. Jyoti Prakash Tamang
Chief Patron
Vice-Chancellor, Sikkim University

Dr K. Birla Singh
Convener
Sikkim University

Dr Basundhara Chettri
Organizing Secretary
Sikkim University