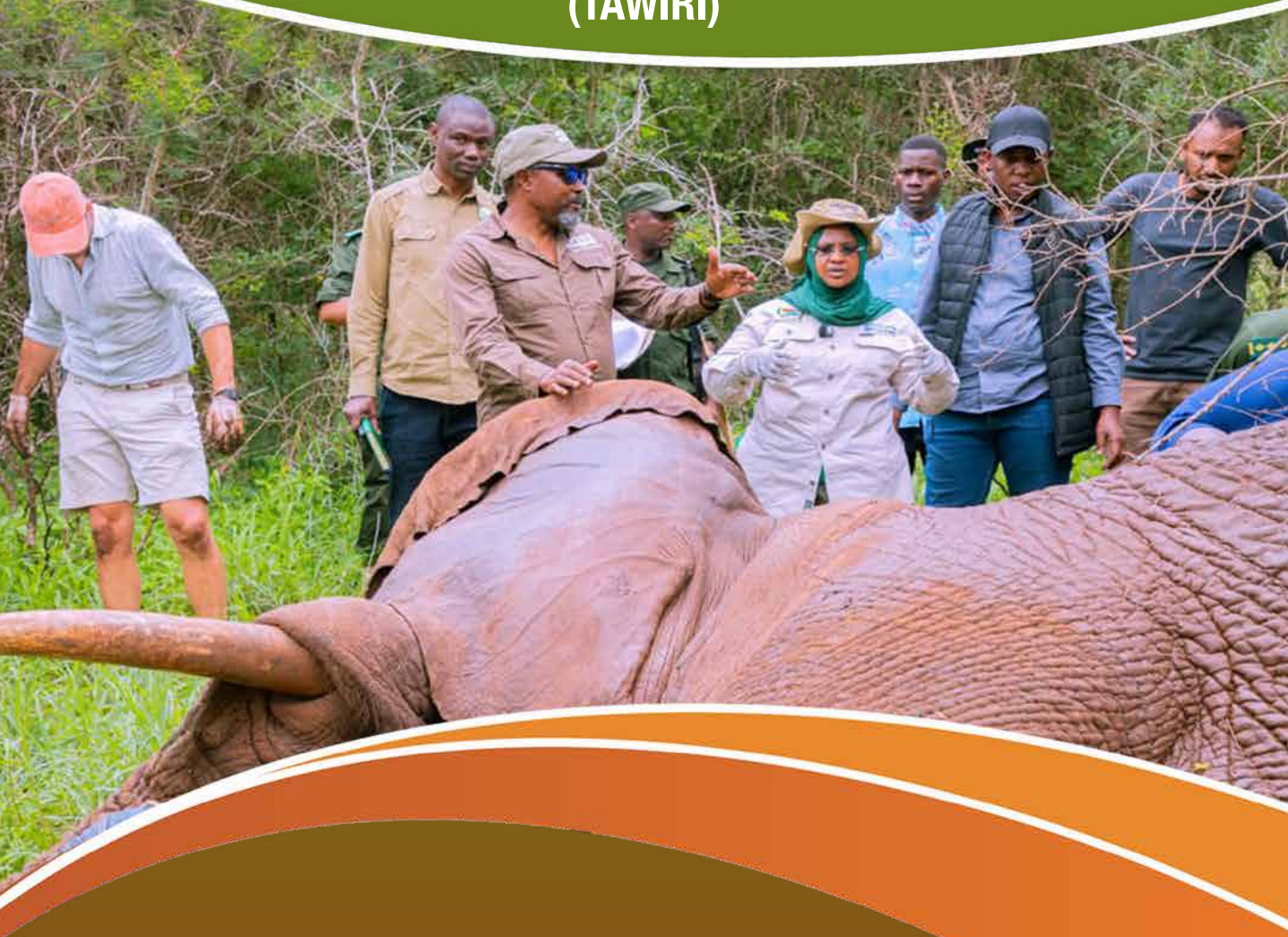




TANZANIA WILDLIFE RESEARCH INSTITUTE (TAWIRI)



PROCEEDINGS OF THE FIFTEENTH TAWIRI INTERNATIONAL SCIENTIFIC CONFERENCE

3rd – 5th DECEMBER, 2025

AT THE ARUSHA INTERNATIONAL CONFERENCE CENTRE, TANZANIA

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CONFERENCE THEME

**“Innovations in Wildlife Conservation and Sustainable Tourism:
Navigating the Future in a Changing World”**



TRIBUTE TO DR. JANE GOODALL



This tribute honors Dr. Jane Goodall for the extraordinary and groundbreaking research on Chimpanzees done in Gombe National Park, Western Tanzania. TAWIRI join the world to celebrate her as a pioneer in Primatology, as well as outstanding conservation scientific efforts, which revolutionized our understanding of primates, specifically Chimpanzee and our place in nature. TAWIRI acknowledges that Goodall's Scientific work transformed Wildlife Conservation and Tourism development, at the same time placing Tanzania-through Gombe National Park at the forefront of global efforts in protecting Chimpanzee and their habitat. Over Sixty years ago - Older than TAWIRI, Jane Goodall arrived on the shores of Lake Tanganyika, guided by curiosity, compassion and unshakable beliefs in the worth of every living being, but focused on Chimpanzee. She makes a Scientific Impact by demonstrating that Chimpanzees, our **"closest relatives"** are not only capable of using tools, but also, they can modify the tools to suit their needs, and exhibit complex social and emotional lives, including joy, grief, compassion and

even caring of each other to name a few. Further to that, Dr. Jane Goodall reveals that human is not separate from nature, rather an integral part and a key factor to conservation and tourism development. As we celebrate the 15th International Scientific Conference, TAWIRI honors Dr. Jane Goodall Scientific achievements, courage and love for Tanzania's wilderness and pledge to continue collaboration with JGI in research. Dr. Jane Goodall's life reminds us that, compassion is the heart of science, understanding is the soul of conservation and our planet's future depends on our choice today.

Rest In Peace Dr. Jane Goodall



Dr. Eblate Mjingo
DIRECTOR GENERAL
Tanzania Wildlife Research Institute

MESSAGE FROM THE ORGANIZING COMMITTEE

The 15th TAWIRI International Scientific Conference took place at the Arusha International Conference Centre (AICC) from 3rd to 5th December, 2025 guided by the theme titled ***“Innovations in Wildlife Conservation and Sustainable Tourism: Navigating the future in a changing world”***. This theme was divided into 12 sub-themes to enhance clarity and provide a better picture of the topics that were expected from the conference as follows:-

- i. Sustainable Tourism
- ii. Habitat Conservation, vegetation, ecology and invasive species
- iii. Financing wildlife conservation (biodiversity credits, wildlife credits and carbon credits)
- iv. Climate change, adaptation and resilience
- v. Human Wildlife Conflict
- vi. Infrastructures development and wildlife conservation
- vii. Innovation and technology in wildlife conservation
- viii. Natural resources governance and conservation
- ix. Water resources and wetland conservation
- x. Wildlife ecology, connectivity ecological interactions
- xi. Bee ecology and pollinators
- xii. Wildlife diseases and ecosystem health

The 15th TAWIRI International Scientific Conference was highly successful, recording the highest level of participation compared to all previous editions. This increase is largely attributed to the adoption of innovative technologies that enabled effective hybrid participation and enhanced engagement among remote attendees. A total of 1,659 participants attended through multiple modalities, representing an increase of 1,004 participants compared to the 655 recorded at the 14th TAWIRI International Scientific Conference in 2023. Participation comprised 624 (37.6%) in-person attendees, 357 (21.5%) exhibition visitors (non-presenters, including members of the general public), 545 (32.8%) participants joining via Zoom, and 133 (8.0%) via Google Meet. In total, participants were drawn from 31 countries (Table 1). The majority originated from Tanzania (1,035; 62%), followed by Kenya, Finland, and the Netherlands (each 4%), and Zimbabwe, the United Kingdom, and the United States (each 3%), while all remaining countries individually contributed 2% or less.

Delegates included representatives from Government Ministries and Departments, Local Government Authorities, management authorities and institutions, research and academic institutions, the private sector, conservation non-governmental organisations (NGOs), development partners and agencies, and media houses. The conference further attracted international participants from East Africa, Europe, the Americas, and Asia, underscoring its expanding global relevance in wildlife conservation and research.

Table 1: Number of Participants and their respective countries

S/N	Country	Number	S/N	Country	Number
1	Afghanistan	1	17	India	12
2	Angola	1	18	Italy	7
3	Austria	1	19	Japan	15
4	Algeria	3	20	Kenya	73
5	Argentina	3	21	Macau	12
6	Australia	16	22	Netherlands	60
7	Bahrain	4	23	South Sudan	2
8	Hungary	2	24	Sudan	3
9	Bangladesh	10	25	Sweden	3
10	Belgium	15	26	Switzerland	25
11	Canada	25	27	Tanzania	1035
12	Denmark	50	28	United Kingdom	56
13	Finland	68	29	United States	54
14	France	15	30	Zambia	12
15	Gabon	5	31	Zimbabwe	57
16	Germany	14			

Paper presented

The conference recorded an increased number of papers presented, with a total of 256 contributions. These comprised 183 oral presentations, 6 symposia, 3 workshops, 7 seminars, and 56 poster presentations, representing an increase of 24 papers compared to the 232 presented at the 14th Conference in 2023. Across the 12 sub-themes, *Wildlife Ecology and Ecological Interactions* accounted for the highest number of presentations, followed by *Human–Wildlife Conflict*. Other prominent sub-themes included *Natural Resources Governance and Conservation*, *Bee Ecology and Pollinators*, and *Innovation and Technology in Wildlife Conservation and Sustainable Tourism*, as illustrated in the figure below. The presentations addressed current challenges in wildlife management and proposed practical solutions and mitigation measures, particularly in relation to increasing human–wildlife conflict (HWC), biodiversity conservation, and the application of technology in conservation planning and management.

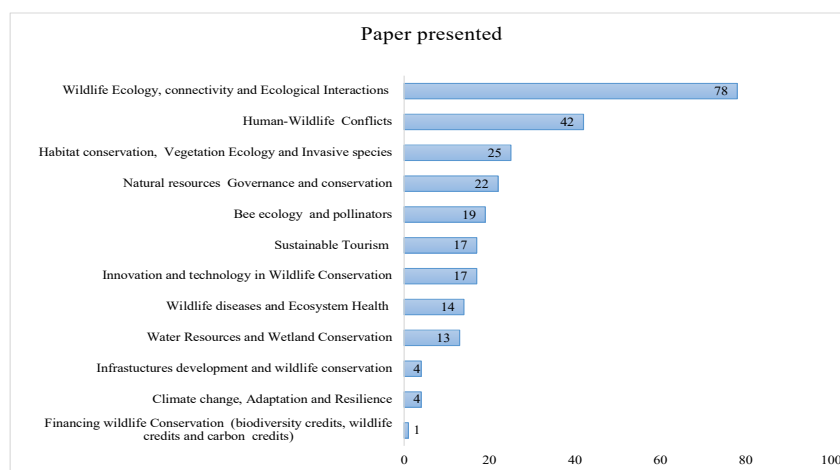


Figure 1: Papers presented in relation to subthemes

Sponsored students

TAWIRI supported a total of 12 postgraduate students (Master's and PhD levels) who qualified to present their research findings at the conference (Table 2). Of these, five were from the African College of Wildlife Management (Mweka), three from the University of Dodoma, two from the Nelson Mandela African Institution of Science and Technology, one from the University of Dar es Salaam, and one from Sokoine University of Agriculture. Additional students were supported by their respective institutions and private sponsors. Among the participating students, two were recognised as **Best Young Scientist Presenters**: Phoebe Mpanzo (Mweka) for the best oral presentation, and Mariam Michael (Sokoine University of Agriculture) for the best poster presentation.

Table 2: Sponsored students during the 15th TAWIRI International Scientific conference

S/N	Name	Title	University	Degree
1	Naushard Yunus	Contribution of Community-Based Conservation in Enhancing Wildlife Conservation at Makuyuni Wildlife Park, Northern Tanzania	Mweka	BSC -Year 3
2	Emanuel Charles	The land use and land cover changes at Minja forest reserve using remote sensing techniques	Mweka	BSC - year 3
3	Samwel Rashid	Assessing the Dynamics of Agricultural and Settlement Development and their Impact on Mountain Hanang Nature Forest Reserve Using Remote Sensing	Mweka	BSC - year 3
4	Awadhi Mndeme	Evaluating Land Use Land Cover in Mramba Forest Reserve Using GIS and Remote Sensing	Mweka	BSC - year 3
5	Godfrey Harry	Assessment of land use and land cover change A case study of Kitendeni wildlife corridors, Tanzania	Mweka	BSC - year 3
6	Mecklina M. Mbundi	Long-term changes in dung beetle community structure in the savanna grassland of the Serengeti ecosystem	Nelson Mandela	PhD - One health
7	Francis Chebby	Perceptions and Awareness of Chicken Farmers on Newcastle Disease and Indian House Crow (<i>Corvus splendens</i>)–Chicken Interactions: A Case of Dodoma, Tanzania	Nelson Mandela	PhD- Biodiversity
8	Faraja Kakulwa	Genetic barcoding uncovers hidden diversity of haplochromines cichlids in Shirati Bay, Lake Victoria	SUA	BSC - year 3
9	Paulo C. Athuman	Spatial Variation of the Indian House Crow (<i>Corvus splendens splendens</i>) Breeding Biology	UDOM	PhD. Environmental Science
10	Hefsiba Mawazo	Factors Influencing Community Awareness of the Endangered Rungwecebus kipunji	UDOM	BSC - year 3
11	Paul-Pastroy	Impact of Audio-Visual Content on Domestic Tourist Visit Intention: Case of Tanzania National Parks	UDSM	PhD -Management
12	Selela Mwakalila	Factors influencing tourists' satisfaction: A case study of homestays in Tanzania	UDSM	PhD - Management

On behalf of the Management and staff of TAWIRI, the Organising Committee of the 15th TAWIRI International Scientific Conference is deeply honoured to present these proceedings and to extend a cordial invitation to scientists, conservationists, management authorities, government officials, representatives of non-governmental and civil society organisations, institutions of higher learning, and the wider public to engage with the knowledge contained herein. These proceedings embody a rich collection of scientific insights, innovative approaches, and practical experiences generated through rigorous wildlife research and thoughtful deliberations during the conference. It is our sincere hope that the findings presented will serve as a valuable resource to guide policy formulation, inform strategic planning, and inspire evidence-based decision-making in support of sustainable wildlife conservation and natural resource management in the country.

We trust that this body of work will not only contribute to addressing current and emerging conservation challenges, but also foster stronger collaboration among stakeholders, promote innovation, and reinforce our collective commitment to safeguarding biodiversity for present and future generations.

REVIEWED AND PUBLISHED PAPERS

A total of 74 manuscripts were submitted and underwent peer review. Following the review process, 5 manuscripts were rejected, and 68 were accepted for publication in this volume of proceedings.

Organizing Committee Members of the 15th TAWIRI International Conference of 2025

i. Dr. Julius Keyyu	-	Chairperson
ii. Ms. Angela Hezekiah	-	vice Chaiperson
iii. Dr. Janemary Ntalwila	-	Secretary
iv. Ms. Alphoncina Mponzi	-	Depurty Secreatry
v. CPA Daniel wanna	-	Member
vi. Mr. Machoke Mwita	-	Member
vii. Ms. Elizabeth Mwakosya	-	Member
viii. Mr. Brian Maganga	-	Member
ix. Mr. Petro Luvanda	-	Member



TABLE OF CONTENTS

TRIBUTE TO DR. JANE GOODALL	iii
MESSAGE FROM THE ORGANIZING COMMITTEE	iv
RESOLUTIONS FOR THE 15TH TAWIRI INTERNATIONAL SCIENTIFIC CONFERENCE, HELD ON 3RD- 5TH DEC. 2025, AT THE ARUSHA INTERNATIONAL CONFERENCE CENTRE	xiv
ACKNOWLEDGEMENT	xvii
KEYNOTE SPEAKERS	
Innovation and technology in wildlife conservation: Harnessing technology and innovation to advance wildlife conservation in Tanzania <i>Dr. Ernest Eblate Mjingo</i>	<i>xix</i>
Trophy hunting as a conservation and local community empowerment tool in the context of Southern Africa? Lessons and experiences from Zimbabwe <i>Prof. Muboko N & Gandiwa E.</i>	<i>xx</i>
Sustainability in changing world: Cross-road or highway for wildlife tourism? <i>Prof. Dev Jani</i>	<i>xxi</i>
REVIEWED AND PUBLISHED PAPERS OF THE 15TH TAWIRI INTERNATIONAL SCIENTIFIC CONFERENCE OF 2025	
SUBTHEME: WILDLIFE ECOLOGY CONNECTIVITY ECOLOGICAL INTERACTION	
Transboundary movement and management of elephants at the Northern Tanzania and Southern Kenya borderland <i>Julius Keyyu, Bukombe John, Ally Nkwabi, Mwita Machoke, Kezia Oola', Ramadhan Juma, Edward Justin and Alex Lobora.....</i>	<i>1</i>
Monitoring of wildlife composition, abundance and diversity in relation to habitats in msolwa sector Nyerere National Park, Tanzania..... <i>Emmanuel H. Masenga, Matana Levi, Revocatus Meney, Lucas Ndakize, Asanterabi Lowassa, Ellen Ponsian, Hamenya Mpemba, Elisafi Manase, Brandon Kemp and Eblate Mjingo.....</i>	<i>15</i>
Preliminary rapid assessment on population and spatial distribution of ashy red colobus monkey (<i>Piliocolobus Tephrosceles</i>) in community forests in Biharamulo district, Northwestern Tanzania <i>Kaiza R. Kaganzi, Olivia W. Cox, Griffin Beckley, Oscar P. Nyakunga, Jackson D. Kabeho, Mwima A. Maugo, Agriphina C. Machaniga, Aida Cuni-Sanchez, Robert A. Marchant and Oliver C. Nyakunga.....</i>	<i>33</i>
Genetic barcoding uncovers hidden diversity of haplochromine cichlids in Shirati bay, Lake Victoria <i>Faraja S. Kakulwa, Magoti E. Ndaro, Jackson L. Saipereki, Anthony D. Funga and Cyrus Rumisha.....</i>	<i>45</i>
How prey threatens survival of predator: A field experience on African lion in Nyerere-Selous complex <i>Mikidadi Mtalika and Haruna Ramadhani</i>	<i>54</i>

Assessing the influence of wildlife herbivory, fire and climate on grass growth in Mikumi National Park, Tanzania <i>Rekichius M. Kadogo, Yohana W. Msangi, Deusdedit B. Fidelis, Shaban F. Shaban, Samuel N. Mtoka, Abduliazizi M. Hatib, Amani S. Kitegile and Guy W. Norton</i>	60
Environmental and anthropogenic drivers of spatial distribution of common eland in Burigi-Chato National Park, Tanzania <i>Loyce Majige, John Sanare, Cecilia Leweri, Rahabu Makongoro, Machoke Mwita, Hamza Kija, Wickson Kibasa, Goodluck Massawe, Joel Ephraim and Devolet Mtui</i>	69
Large-Medium species composition, diversity, and abundance in the Arusha National Park, Northern Tanzania <i>Elizabeth Mwakosya, Richard D. Lyamuya, Baraka Naftal, John Erasto, Magreth Gabriel, Felix Shayo, Damari Nassary, Machoke Mwita, Hamza Kija and Eblate Mjinga</i>	79
A survey of birds of Kitwe woodland in Kigoma-Ujiji Municipality, Western Tanzania <i>Chacha Werema and Ally K. Nkwabi</i>	91
Seasonal variation in ranging patterns and activity budget of yellow baboons in the Issa Valley, Tanzania <i>Mali. T. Lepukoti, Fiona. A. Stewart and Alex K. Piel</i>	101
Spatial variation of the indian house crow (<i>Corvus Splendens Splendens</i>) breeding biology, effectiveness and sustainability of control measures a review <i>Paulo Chiza Athumani, Franco Peniel Mbise and Ignas Safari</i>	110
Mammal diversity and distribution for enhancing visitors experience and conservation in Ruaha National Park <i>Cecilia Leweri, Lazaro Mangewa, Wilfred Marealle, John Bukombe, Jerome Kimaro, Hamza Kija, Pius Kavana, Richard Lyamuya, Machoke Mwita, Asanterabi Lowassa, Joel Ephraem, John Sanare, Janemary Ntalwila, Rahabu Makongoro, Alex Lobora, Edward Kohi, Julius Keyyu and Eblate Mjinga</i>	126
Long-term changes in dung beetle community structure in the savanna grassland of the Serengeti Ecosystem <i>Mecklina Michael Mbundi, J. Grant C. Hopcraft, Gareth P. Hempson, Robert F. Foster and Thomas A. Morrison</i>	142
Impact of habitat disturbance on the abundance of black-and-white colobus monkeys in Rau Forest, Moshi, Kilimanjaro, Tanzania <i>Nassoro Salimin Mfanga, Shabani Hamisi Mfanga, Janemary Ntalwila and Kwaslema Malle Hariohay</i>	155
Factors influencing community awareness of the endangered (<i>Rungwecebus kipunji</i>) <i>Hefsiba Mawazo, Janemary Ntalwila and Franco P. Mbise</i>	168
Analysis of changes in elephant population in tarangire National Park, Northern Tanzania <i>Agriphina C. Machaninga, Kaiza R. Kaganzi, Madeline Krug, Ngumbang A. Juat, Jessica P. R. Thorn and Oliver C Nyakunga</i>	176
Terrestriality in arboreal red-tailed monkeys (<i>Cercopithecus Ascanius</i>) in Issa Valley, Western Tanzania <i>Joram. L. Navaya, Fiona. A. Stewart, Alex K. Piel</i>	190
Preliminary faunal biodiversity assessment of community-managed coastal forests in Mkinga District, North-Eastern Tanzania. <i>Steven Temu, Pius Mollel, Samwel Slaa, Richard Katondo, Thomas Sawe, Kolady Kayanda and Almasi Kashindye</i>	200

Mammalian species composition and diversity in the Aga Khan University's Lands, Arusha Tanzania
Janemary Ntalwila Ally Nkwabi, John Sanare, Neema Kilimba, Richard D. Lyamuya, Murad Jivan, Loyce Majige, Pius Y. Kavana⁴ Cecilia Leweri, Deusdedith Bwenge, Asanterabi Lowassa Azmina Jivan, Happiness Moshi, Bashir Adelodun and Emmanuel Sulle..... 220

Bird species composition, diversity and abundance: The case of the Aga Khan university lands in Arusha, Tanzania
Ally K. Nkwabi, John Sanare, Murad Jivan, Neema Kilimba, Richard Lyamuya, Emmanuel Sulle, Azmina Jivan, Happiness Anold Moshi, Bashir Adelodun, Pius Y. Kavana and Janemary Ntalwila.... 232

Fifteen years of ex-situ recovery for the Kihansi spray toad (*nectophrynoides asperginis*) in tanzania: Demography, environmental controls, and reintroduction readiness at Kihansi captive breeding facility
John K. Bukombe, Hussein A. Elias, Lameck J. Magesa, Juma Kimera, Raymond Okick, Arnold D. Shao, Alphonse D. Echumba, Janemary Ntalwila¹, Felix A. Shayo², Samuel Mtoka, Devolent Mtui, Asanterabi Lowassa, Wilirk Ngalason, Cuthbert Nahonyo, Julius Keyyu and Ernest Mjinga..... 251

SUB-THEME: INNOVATION AND TECHNOLOGY IN WILDLIFE CONSERVATION

Real-time classification of serengeti wildebeest behaviour with edge machine learning and a long- range IOT network
Cyrus M. Kavwele, J. Grant C. Hopcraft, Juan M. Morales, Gerald Nyaffi, Nancy Kimuya and Colin J. Torney..... 260

Establishing environmental and social baselines for ai-driven climate adaptation and human-wildlife conflict mitigation in the Ruaha-Rungwa Ecosystem, Tanzania
Emmanuel Kazuva and Abel Mtui 277

SUB THEME: HUMAN-WILDLIFE CONFLICTS

Perceptions and awareness of chicken farmers on newcastle disease and indian house crow (*corvus splendens*) interactions in Dodoma, Tanzania
Francis Chebby, Naza Mmbaga, Paulo Athumani, Theresia Ndanu and Kelvin Ngongolo..... 286

A poisoning incidence involving a white-backed vulture (wbv) in Mikumi National Park, Tanzania: A case report and review of an emerging crisis
Haruna Ramadhani, Wolfgang Sagari and Mikidadi Mtalika 297

Crop preferences of african elephants: A comparative study of crop damages to farmers adjacent to Mikumi National Park, Tanzania
Deusdedith Bwenge Fidelis and Asanterabi Lowassa..... 305

Exploring local beliefs, attitudes and practices toward the white-backed vultures in Northern Tanzania: Implications for conservation and tourism
Nickson P. Mkiramweni..... 313

Local community coping and adaptation strategies on human-wildlife conflict at Kwakuchinja wildlife corridor, connecting Tarangire-Manyara National Parks
Emmanuel A. Ngomuo, Ellen K. Ponsian, Asanterabi Lowassa, Emmanuel H. Lyimo, Emmanuel Masenga, Kwaslema M. Hariohay, Rehema A. Shoo, Emanuel H. Martin, & Alex W. Kisingo..... 329

Socio-Economic and psychological impacts of elephant incursion in Rhotia-Karatu, Arusha Region, Northern Tanzania.
Glory Gerald Tarimo, Kwaslema Malle Hariohay, Asanterabi Lowassa and Janemary Ntalwila 342

Perceived patterns of hyena-induced livestock losses in communities adjacent to Mkomazi National Park <i>Franco Peniel Mbise</i>	3567
Avian-aircraft collision risk assessment at the Julius Nyerere International Airport in Dar Es Salaam, Tanzania <i>Jackline Nguma and Jasson John</i>	366
Human-Elephant interactions viewed as assets or liabilities in rural transformations: A case study from Tanzania with examples from other elephant range states <i>Donald Gregory Mpanduji and Paulo Chiza Athumani</i>	378
Factors influencing community perspectives on elephant crop damage and conservation support in Tanzania <i>Franco Peniel Mbise</i>	392
Assessment on impacts of crop raiding elephants to food availability and accessibility to Mikumi Farmers <i>Deusdedith Bwenge Fidelis and Asanterabi Lowassa</i>	403
Community engagement through outreach initiatives for villages living adjacent to Tarangire National Park <i>Franco P. Mbise, Timotheo Sosiya, Ntalwila and Eivin Røskraft</i>	411
Integrating AI technologies with indigenous practices to mitigate human-wildlife conflicts in communities adjacent to Ruaha National Park <i>Emmanuel Patroba Mhache* and Ruth John Wairimu</i>	419

SUB-THEME: BEE ECOLOGY AND POLLINATORS

Wild edible insects enhance food security and promote cultural heritage in the Lake Zone, Tanzania <i>Renatus Dioniz Bwiki, Aidath Ismail Ishalula and Filemon Elisante</i>	430
Preliminary survey of butterflies in Moyowosi Game Reserve, Western Tanzania <i>Selemani R. Moshi, Fredrick P. Mhalafu, Baraka N. Mbwambo, David A. Collins, Neema E. Kilimba, Richard D. Lyamuya, Emmanuel R. Pagiti, Shabani H. Matwili, Simula P. Maijo, Angela R. Mwakatobe, Sood A. Ndimuligo, Melack A. Iddi and Ally K. Nkwabi</i>	443
Butterfly species composition in various habitat types in the Aga Khan University's Arusha Climate and Environmental Research Centre <i>Neema Kilimba, Ally K. Nkwabi, Richard Lyamuya, John Sanare, Pius Kavana, Murad Jivan, Janemary Ntalwila, Happiness Anold Moshi, Asanterabi Lowassa, Azmina Jivan, Bashir Adelodun Emmanuel Sulle</i>	456
Assessment of the opportunities and challenges facing beekeeping activities at Rau Forest Reserve, Moshi <i>Ellen K. Ponsian, Emmanuel A. Ngomuo, Asanterabi Lowassa, Baraka M. Naftali, Dennis M. Kisika and Emmanuel H. Masenga</i>	471
Spatial modelling of beekeeping suitability under climatic seasonality and land-use accessibility in the greater Mahale Ecosystem, Tanzania <i>Baraka Naftal, Simula Maijo, Angela Mwakatobe, Sood Ndimuligo, Ally Nkwabi, Pius Kavana, Emanuel Pagiti, Alphonsina Machichanga, Paul Mjema, Wilfred Marealle, Ellen Ponsian, Revocatus Meney, Noris Mushi, Deus Mjungu and Eblate Mjingo</i>	482

SUB-THEME: CLIMATE CHANGE ADAPTATION AND RESILIENCE

- The implications of climate change on Wildlife Resources Management in Tanzania's Southern Protected Areas
Tutindaga George and Richard Y. M. Kangalawe..... 498
- Mapping stakeholders and assessing community perceptions for AI-Driven climate resilience in the Ruaha-Rungwa Ecosystem, Tanzania
Ladislau Batinoluho, Godfrey Nyangaresi and Daudi Lema..... 519

SUB-THEME: HABITAT CONSERVATION, VEGETATION ECOLOGY AND INVASIVE SPECIES

- Perception and willingness to conserve on-farm medicinal plants: Insights from communities in West Usambara Mountains, Tanzania
Chelestino P. Balama, Upendo L. Msalilwa, Samson Hilonga, Zacharia S. Mmary, Naftal L. Mahewa, Greyson F. Mwombeki, Ramadhani K. Mazige and Josephine Kajembe..... 528
- Using normalized difference vegetation index and land surface temperature analysis support sustainable management of riverine forests in Sudan?
Elmugheira M. I. Mohammed and Elharith A. A. Hagr..... 544
- The land use and land cover changes at minja forest reserve using remote sensing techniques, 1995–2025
Emmanuel Charles Mlinga and Gabriel Mayengo 555
- Assessment of land use and land cover change: a case study of Kitendeni Wildlife Corridor, Tanzania
Godfrey G. Harry and Gabriel Mayengo 564
- Demographic dynamics of yellow baboons (*Papio Cynocephalus*) in Mikumi National Park, Tanzania
Rekichius M. Kadogo, Deusdedit B. Fidelis, Samuel N. Mtoka, Amani S. Kitegile and Guy W. Norton..... 576
- Assessment of abundance and distribution of invasive alien plant species in Pugu – Kazimzumbwi Nature Forest Reserve (PKNFR), Tanzania
Ezekiel Edward Mwakalukwa, Sami Dawood Madundo and Canisius John Kayombo 583
- Carbon storage potential of miombo woodland trees in wildlife protected areas: a case study of Ruaha National Park, Tanzania
Raymond Okick, John Bukombe Nelson Mungoya, Richard Lyamuya, Alphonse Echumba, Asanterabi Lowassa, Ally Nkwabi, Arnold Shao and Felix Shayo 601
- Integrated assessment of invasive alien plant species in tanzania: Distribution, habitat suitability, environmental drivers and strategic management responses
John K. Bukombe, Hamza Kija, Julius Keyyu, Asanterabi Lowassa, Lazaro Mangewa, Raymond Okick, Janemary Ntalwila, Deusdedit Bwenge, William Kindeketa and Ezekiel E. Mwakalukwa... 610
- Plant species composition and diversity of the Aga Khan University, Arusha Campus, Northern Tanzania
Richard D. Lyamuya, Pius Y. Kavana, Ally Nkwabi, Neema Kilimba, Emmanuel Sulle, Murad Jivan, John Sanare, Azmina Jivan Happiness Moshi, Bashir Adelodun and Janemary Ntalwila.... 626
- Biodiversity conservation in wetlands through integrated water resources management: Key challenges, opportunities, and lessons from the Katuma Catchment, Tanzania
Philemon Mneney, Baraka Naftal, Simula Maijo, Juma Maziku, Scholastica Mbinile, Fadhili

<i>Njilima, Thadeus Ndeseiyo, Siya Mamiro, Germanus J. Hape, Doyi Mazenzele and Angela Mwakatobe.</i>	650

SUB-THEME: INFRASTRUCTURE DEVELOPMENT AND WILDLIFE CONSERVATION

Status of wildlife roadkill in the Ngorongoro Conservation Area, Northern, Tanzania. <i>Richard D. Lyamuya, Gregory A. Mtega, Victoria Shayo, Gabriel Mayengo, Francis Makari and Pius Mollel</i>	664

SUB-THEME: INNOVATION AND TECHNOLOGY IN WILDLIFE CONSERVATION

Dynamics of agricultural and settlement development and their impact on Mountain Hanang Nature Forest Reserve using remote sensing <i>Samwel Ramadhan Rashid and Gabriel Mayengo</i>	675
Spatial patterns of mammal diversity and richness from year-long camera trapping in the Kitendeni Wildlife Corridor, Northern Tanzania <i>Baraka Naftal, Frank Damson, Emmanuel Kivuyo, Igno Isaack Laitayok, Mariamagdalena Paul, and Noah Sitati</i>	693

SUB-THEME: WATER RESOURCES AND WETLAND CONSERVATION

Threats, perceived wetland health, and drivers of community participation in conservation of the Mara River Wetland, Tanzania <i>Emmanuel R. Pagiti, Nsajigwa E. Mbije, Makarius C.S. Lalika, Angela Mwakatobe</i>	709
Water quality and reintroduction outcomes: A case study of the Kihansi spray toad <i>Felix A. Shayo^{1*}, Devolet T. Mtui, Clamsen Mmassy, Edward M. Kohi, Juma Kimera, Mikidadi Mtalika and Eblate E. Mjinga</i>	729
Genetic barcoding uncovers hidden diversity of haplochromine cichlids in shirati bay, lake Victoria <i>Faraja S. Kakulwa, Magoti E. Ndaro, Jackson L. Saipereki, Anthony D. Funga and Cyrus Rumisha</i>	739

SUB-THEME: WILDLIFE DISEASES AND ECOSYSTEM HEALTH

Preliminary survey of ectoparasites on rodents and shrews on Mount Kilimanjaro <i>Alphonse D. Echumba, Abdul S. Katakweba, Vedasto G. Ndibalema, Suzana M. Thomas, Richard Lyamuya, Genet B. Gebrezgiher, John K. Bukombe, Ally nkwabi, Asanterabi Lowassa, Felix Shayo, Arnold D. Shayo and Raymond E. Okick</i>	748
What does the researcher need to consider when planning for capture and collaring of carnivores in Nyerere National Park and Selous Game Reserve? <i>Mikidadi Mtalika and Haruna Ramadhani</i>	759
Surveillance of zoonotic diseases in wild chimpanzee populations: Lessons from the greater Gombe Ecosystem, Tanzania. <i>Priscilla S. Shao B.Sc., Jessica R. Deere Ph.D., Dismas E. Mwacha, D.V.M., Kibaliza Kimata B.Sc., Tedy R. Dasher., Gretchen C. Anchor, M.S., Deus Mjungu Ph.D., ⁴Lilian Pintea Ph.D., Karen A. Terio, D.V.M., Ph.D., Dipl. A.C.V.P., Thomas R. Gillespie, Ph.D., Elizabeth V. Lonsdorf, Ph.D.</i>	765
Climate and infrastructure drive the spread of <i>anopheles stephensi</i> in Kenya <i>Vincent Obanda, Benard Agwanda, Tim.J. L Wango and Olivia W. Lwande</i>	773

Prevalence of haemoparasites in rodents and shrews trapped from domestic and peridomestic houses in Morogoro Municipality, Tanzania <i>Alfred Thadeus Shayo and Abdul A. Katakweba</i>	783

SUB-THEME: NATURAL RESOURCES GOVERNANCE AND CONSERVATION

Can tanzania sustain inclusive green growth and shared prosperity? A theoretical review <i>Odass Bilame, Asanterabi Lowassa and Janemary Ntalwila</i>	790

Community-driven regeneration of miombo woodlands and chimpanzee corridors in western Tanzania <i>Annolbert Alexander Mutalemwa & Leticia Apolinary Mlawa</i>	799

SUB-THEME: SUSTAINABLE TOURISM

Categorising domestic tourists in Tanzania's National Parks: Types of tourists' theory approach <i>Kezia H. Mkwizu</i>	806

Assessing tourist vehicles in accommodating tourists with disabilities at Ngorongoro Conservation Area; tour guides' perception <i>Juliana Maganga and Gabriel Mayengo</i>	813

Sustainable tourism and women in higher education: perspectives on visits to national parks <i>Kezia H. Mkwizu and Zakia M. Ituja</i>	823

The Long-Term Monitoring of livestock predation by large carnivores in the Mkomazi Sub-Landscape, Tanzania <i>Emanuel Kivuyo, Elia Sabula and Noah Sitati</i>	830

Confronting Human-Wildlife Conflict in the Mkomazi Ecosystem, Tanzania <i>Emanuel Kivuyo, Kayanda Kolady, Elia Sabula and Noah Sitati</i>	840

RESOLUTIONS FOR THE 15TH TAWIRI INTERNATIONAL SCIENTIFIC CONFERENCE, HELD ON 3RD 5TH DEC. 2025, AT THE ARUSHA INTERNATIONAL CONFERENCE CENTRE

The Conference deliberations yielded rich insights, thoughtful reflections, and forward-looking recommendations aimed at strengthening the conservation sector in Tanzania. These resolutions underscore a shared commitment to safeguarding biodiversity, enhancing sustainable livelihoods, and addressing emerging challenges most notably the escalating human–wildlife conflicts. The key resolutions are presented as follows:

1. Innovation, Technology, and Data Science in Conservation and Tourism

Recognizing the transformative role of innovation and technology, the Conference resolves that:

- i. Advanced technologies including artificial intelligence, eco-acoustics, GIS, remote sensing, camera traps, IoT sensors, and DNA-based tools be systematically integrated into conservation and tourism for improved biodiversity monitoring and management.
- ii. Forensic and molecular techniques be strengthened to support anti-poaching initiatives, wildlife crime investigations, and prosecution processes.
- iii. Data-driven approaches be promoted to address human–wildlife conflicts through predictive modelling, early-warning systems, and disease surveillance at the wildlife–livestock interface.
- iv. Strategic partnerships with regional and international institutions be enhanced to advance research, monitoring, and evidence-based interventions.

2. Escalating Human–Wildlife Conflicts

Noting with concern the increasing incidence of human–wildlife conflicts driven by population growth, habitat loss, land-use change, and climate variability, the Conference resolves that:

- i. Integrated mitigation strategies encompassing land-use planning, early-warning systems, and rapid response mechanisms be strengthened and effectively implemented.
- ii. Community awareness and engagement be enhanced through sustained education on coexistence, wildlife behavior, reporting procedures, and compensation frameworks.
- iii. Applied research on conflict dynamics and mitigation effectiveness be intensified to inform policy and practice.
- iv. Investments in preventive infrastructure and technologies be prioritized in high-risk areas.
- v. Multi-sectoral collaboration be reinforced to ensure coordinated and adequately resourced responses.

3. Emerging and Zoonotic Diseases

Recognizing the growing threat of zoonotic and emerging diseases at the wildlife–livestock–human interface, the Conference resolves that:

- i. National and institutional diagnostic capacities be strengthened through modern laboratory infrastructure and rapid detection tools.
- ii. Cross-sectoral collaboration be enhanced under the One Health framework.
- iii. Genomic and molecular research be supported to track pathogens, antimicrobial resistance, and disease transmission pathways.
- iv. Community awareness and participation in surveillance and reporting systems be promoted.

4. Invasive Species Management

Concerned by the increasing spread of invasive plant species, the Conference resolves that:

- i. Early detection, control, and eradication measures be strengthened across ecosystems.
- ii. Sustainable and innovative utilization of invasive species be promoted.
- iii. Integrated management approaches be adopted.
- iv. Restoration of degraded ecosystems be prioritized through science-based interventions.

5. Land-Use Change and Habitat Degradation

Acknowledging the impacts of land-use change and habitat degradation, the Conference resolves that:

- i. Effective land-use planning and enforcement be strengthened at all levels.
- ii. Landscape-scale conservation partnerships be promoted.
- iii. Restoration of degraded rangelands be prioritized.
- iv. Ecological connectivity and protection of wildlife corridors be enhanced.
- v. Innovative financing mechanisms, including carbon and biodiversity credits, be advanced.
- vi. Collaborative land management approaches be encouraged.

6. Sustainable Tourism Development

Recognizing tourism as a cornerstone of the national economy, the Conference resolves that:

- i. Innovation in tourism policy, management, and marketing be strengthened.
- ii. Adaptive knowledge, attitudes, and practices in tourism be promoted.
- iii. The principles of environmental, social, and economic sustainability be upheld.
- iv. Tanzania's low-volume, high-quality tourism model be maintained.
- v. Community benefits and value chain development be enhanced.

7. Wildlife Crime and Poaching

Alarmed by the resurgence of wildlife crime, the Conference resolves that:

- i. Legal and institutional frameworks be strengthened to enhance deterrence and accountability.
- ii. Evidence-based and intelligence-led anti-poaching strategies be implemented.
- iii. Collaboration among enforcement agencies, communities, and partners be intensified.
- iv. Technological capacity and ranger support systems be enhanced.
- v. Demand reduction campaigns be strengthened at national and international levels.

8. Community Engagement in Conservation

Recognizing the central role of communities, the Conference resolves that:

- i. Inclusive and participatory approaches to conservation be strengthened.
- ii. Community empowerment, particularly for women, be prioritized.
- iii. Conflict-reducing agricultural practices be promoted.
- iv. Conservation education and awareness be expanded.
- v. Local stakeholder collaboration be enhanced.

9. Transboundary Wildlife Management

Acknowledging the transboundary nature of wildlife, the Conference resolves that:

- i. Monitoring of cross-border wildlife movements be strengthened.
- ii. Regional collaboration with neighboring countries be enhanced.
- iii. Harmonization of policies, legislation, and capacity-building efforts be promoted.

10. Beekeeping and Livelihoods

Recognizing the importance of beekeeping, the Conference resolves that:

- i. Climate-resilient beekeeping be promoted as a nature-based solution.
- ii. Policy and institutional support be strengthened.
- iii. Multi-stakeholder collaboration be enhanced to improve practices and ecosystem health.

11. Trophy Hunting

Recognizing its role in conservation and livelihoods, the Conference resolves that:

- i. Trophy hunting be sustainably managed and effectively regulated.
- ii. Relevant policy, legal, and institutional frameworks be strengthened and enforced.

12. Human Resource Capacity in Conservation and Tourism

Recognizing the importance of human capital, the Conference resolves that:

- i. Workforce capacity in conservation and tourism be strengthened.
- ii. Continuous professional development and skills enhancement be prioritized.
- iii. Staff welfare and motivation, particularly for frontline personnel, be improved.
- iv. Mental health awareness and support systems be institutionalized.

ACKNOWLEDGEMENT

Organizing a conference of this magnitude requires strong collaboration, dedicated support, and active participation from a wide range of stakeholders. Tanzania Wildlife Research Institute extends its sincere appreciation to all delegates particularly researchers, academicians, and conservationists whose participation and presentations greatly enriched the conference with valuable insights for conservation. Special recognition is accorded to the Conference Organizing Committee for their tireless commitment and exceptional efforts in ensuring the successful delivery of the event. We further express our heartfelt gratitude to all conservation authorities, institutions, organizations, and private sector partners for their generous financial and material contributions. Their continued support extends beyond this conference, significantly advancing conservation and research initiatives across the country. In the 15th TAWIRI International Scientific Conference, we sincerely acknowledge all supporting organizations (whose logos are presented below) and encourage them to continue their commendable partnership and unwavering support.

Sponsors of the 15th TAWIRI International Scientific Conference held form 3rd - 5th December 2025



KEYNOTE SPEAKERS

We are very grateful to the Key note speakers who prepared the plenary papers and delivered key messages during the conference. Three plenary papers were presented, each had special message that was linked to the main Conference Theme. Plenary presentations are, key papers that provides general overview of the chosen topic that need to be addressed and which conservation driven issues. Three key note papers were presented with a wider range covering important components related to conference theme. Summary of the papers and presenters are indicated below:-

INNOVATION AND TECHNOLOGY IN WILDLIFE CONSERVATION: HARNESSING TECHNOLOGY AND INNOVATION TO ADVANCE WILDLIFE CONSERVATION IN TANZANIA

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This paper underscores the critical role of innovation and technology in advancing wildlife conservation in Tanzania amid increasing ecological and socio-economic pressures. Conservation efforts in the 21st century must confront a range of complex challenges, including habitat loss and fragmentation, illegal wildlife trade and poaching, climate variability, invasive alien species, human–wildlife conflict, and emerging wildlife diseases all of which threaten biodiversity, livelihoods, and sustainable tourism. The presentation demonstrated how modern technologies are transforming conservation practice by enhancing wildlife monitoring, strengthening law enforcement, and enabling evidence-based management. Tools such as drones (UAVs), camera traps, satellite imagery, and high-resolution aerial surveys allow for more accurate and efficient monitoring of wildlife populations and habitats across extensive landscapes. In addition, Geographic Information Systems (GIS), remote sensing, artificial intelligence, and geospatial modelling offer powerful capabilities for analysing ecological patterns, informing land-use planning, and identifying priority conservation areas. Furthermore, molecular techniques, including DNA barcoding, play an important role in combating illegal wildlife trade and improving wildlife disease surveillance. The paper concludes that integrating innovation and technology into conservation strategies is essential for strengthening research, enhancing management effectiveness, and fostering resilient ecosystems and sustainable wildlife conservation in Tanzania.

TROPHY HUNTING AS A CONSERVATION AND LOCAL COMMUNITY EMPOWERMENT TOOL IN THE CONTEXT OF SOUTHERN AFRICA? LESSONS AND EXPERIENCES FROM ZIMBABWE

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Hunting in Africa is among the continent's oldest human activities, historically embedded in both subsistence and cultural practices. In contemporary conservation discourse, however, trophy hunting has become increasingly contested, often shaped more by emotionally driven animal welfare narratives than by empirical scientific evidence. This debate occurs within a broader context of escalating conservation costs across Africa, particularly in landscapes affected by intensifying human–wildlife conflict, habitat loss, and declining public funding for protected areas. The financial requirements for effective protected area management are substantial. Estimates indicate that managing protected areas costs between US\$460 and US\$2,048 per km² (Parker *et al.*, 2009), depending on location, ecological conditions, and enforcement needs. In contrast, revenues generated from trophy hunting are comparatively lower, ranging from approximately US\$138 to US\$1,091 per km² (Lindsey *et al.*, 2025). This discrepancy highlights a persistent funding gap in conservation finance, particularly in countries where tourism revenue remains limited. Despite Africa's rich biodiversity and global conservation significance, the continent captures only about 3–5% of global tourism revenue, underscoring the need for diversified and innovative financing mechanisms to sustain conservation efforts. Against this backdrop, this paper focuses on Zimbabwe as a case study to critically examine the role of trophy hunting within conservation and development frameworks. It reviews the existing legal and policy frameworks governing hunting, evaluates its ecological and financial contributions to wildlife conservation, and assesses its role in supporting community livelihoods and local development. The analysis further explores whether trophy hunting can function as a viable conservation tool or whether alternative models may be required to ensure long-term ecological sustainability and equitable benefit-sharing for communities living alongside wildlife.

SUSTAINABILITY IN CHANGING WORLD: CROSS-ROAD OR HIGHWAY FOR WILDLIFE TOURISM?

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Sustainability is now widely recognized as a fundamental requirement across all sectors; however, its multidimensional and continuously evolving nature makes its practical implementation both complex and context-dependent. In the tourism sector, this challenge is even more pronounced due to dynamic spatial and temporal interactions between supply and demand, as well as the sensitivity of tourism systems to environmental, economic, and social change. As a result, achieving sustainability in tourism requires adaptive and forward-looking management approaches that can respond effectively to uncertainty and transformation. Innovation in tourism management therefore emerges as a critical pathway for maintaining sustainability in the face of constant change. In the specific context of wildlife tourism, this need for innovation becomes even more urgent, as managers and stakeholders must carefully balance ecological conservation objectives with economic development goals. This balance raises important questions regarding the roles, responsibilities, and preparedness of different stakeholders, including governments, private sector actors, local communities, and conservation organizations, in ensuring that tourism development does not compromise biodiversity and ecosystem integrity. Operating within such a complex environment requires systems that are resilient, flexible, and capable of accommodation rather than rigid adherence to uniform “one-size-fits-all” solutions. Effective wildlife tourism management must therefore embrace adaptive strategies that reflect local ecological conditions, socio-economic realities, and governance structures. The papers collectively emphasize key principles of sustainable tourism while proposing innovative management strategies that integrate the Triple Ps—Profit, People, and Planet—with the dual Es—Efficiency and Effectiveness. This integrated framework is intended to strengthen decision-making, enhance resource use, and ensure that tourism development remains both economically viable and environmentally responsible while adapting to inevitable change.

**REVIEWED AND PUBLISHED PAPERS OF THE 15TH TAWIRI
INTERNATIONAL SCIENTIFIC CONFERENCE OF 2025**



SUBTHEME: WILDLIFE ECOLOGY CONNECTIVITY AND ECOLOGICAL INTERACTION

TRANSBOUNDARY MOVEMENT AND MANAGEMENT OF ELEPHANTS AT THE NORTHERN TANZANIA AND SOUTHERN KENYA BORDER LAND

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ABSTRACT

This study was conducted to determine cross border or transboundary movement and management of elephants between the northern Tanzania and southern Kenya borderland. A total of 13 elephants (10 males and 3 females) were fitted with Vertex Plus-13 (+2) Iridium elephant Global Positioning System (GPS) collars in 2020 in Longido Game Controlled Area and Lake Natron game Controlled Area in Longido District in northern Tanzania for monitoring of their movement pattern. Movements of collared elephants at the northern Tanzania-southern Kenya borderland were monitored for five years until 2025. Results showed that overall movement patterns of individual elephants varied greatly with some elephants having wide or large-scale movement patterns than others. One bull (ID 34240) showed large-scale movement in the landscape to the Tarangire-Manyara ecosystem and at the borderland. Movements in the borderland between northern Tanzania and southern Kenya were highly revealed. One elephant (ID 34237) spent most of the time in Kenya and was killed at Kajiado due to human-elephant conflicts conflict mitigation in 2021. Analysis of the twelve remaining elephants revealed that a total of eleven elephants (91%) crossed the border to Kenya at least once during the study period. A total of seven elephants (72%) spent most of their time in northern Tanzania. Overall, collared elephants spent 78% of their time in Tanzania and 22% of their time in Kenya. Movement of elephants was also recorded within the two popular wildlife corridors in northern namely the Tanganyeeiti/ Kikoti and Laasarak wildlife corridors. Ground truthing of movements to locate collared elephants also revealed several human activities in the landscape including human settlement, farming/ cultivation, charcoal making, and livestock grazing. Ground truthing also showed that elephants spent more time in places with more resources especially food and water. It is concluded that there is a high cross-border or transboundary movement of elephants in the northern Tanzania and southern Kenya borderland. Therefore, joint cross border monitoring and management of elephants is important for sustainable transboundary conservation of the species.

Key words: *Elephant, Northern Tanzania, Southern Kenya, Transboundary Movement*

INTRODUCTION

Transboundary areas are international political boundaries that define countries and territories, thus subjecting landforms like mountains, rivers

and lakes to differing sovereignties (Prescott & Triggs, 2008). Most of the transboundary areas support exceptional levels of species endemism and play important roles in ensuring ecological connectivity by enhancing movement of

wildlife populations. For example, at least 76% of African elephants (*Loxodonta Africana*) are transboundary populations (Lindsay *et al.*, 2017). Most of these transboundary areas are located in environmentally heterogeneous areas often suffering from weak governance, consequently biodiversity in these regions is in jeopardy.

Transboundary areas are increasingly important towards biodiversity conservation due to issues that will drive substantial biodiversity loss (Sutherland *et al.*, 2017). First, climate change is expected to continue to drive shifts in species distribution, whilst pushing species across international borders where differences in policies could pose new threats (Rüter *et al.*, 2014) birds, reptiles, amphibians and butterflies. Second, biodiversity is increasingly threatened by illegal wildlife trade and poaching in border areas (Ellison, 2014), hence this highlights the necessity to coordinate and sanction international collaboration and cooperation for the protection of nature through international treaties, such as Convention on International Trade in Endangered Species of wild flora and fauna (CITES), Convention on Biological Diversity (CBD) and Convention on the Conservation of Migratory Species of wild animals (CMS). Therefore, strengthening biodiversity conservation at transboundary conservation areas has thus emerged as a high priority for safeguarding ecological integrity between and among countries through transboundary conservation.

Transboundary conservation has been proposed as a means for peacemaking (Barquet, 2015), and efforts are growing because of recognition that intact and connected ecosystems requires large scale planning and management approaches that cross political boundaries (Rands *et al.*, 2010), due to recent multicountry conservation planning initiatives and programs attest to this trend (Erg *et al.*, 2015). Nevertheless, political boundaries impede conservation because of lack of coordinated action (Kark *et al.*, 2015). At the species level, transboundary migratory animals especially

African elephant have received an attention through agreements and need for international coordination in species management (Trouwborst *et al.*, 2017). Despite increasing interest in transboundary conservation and international agreements that promote cross border management of species, the international boundaries remain an impediment to wildlife conservation (Kark *et al.*, 2015). One of the key challenge in transboundary conservation is varied laws or priorities pertaining to wildlife between countries and unequal capacity to manage species, which hinder transboundary conservation actions (Arrondo *et al.*, 2018). Therefore, there is a strong need to strengthen transboundary collaboration, which has proven to be successful through trans-boundary data sharing and standardization monitoring of species.

The borderlands between northern Tanzania and southern Kenya form a dynamic transboundary landscape that support large, mobile elephants' populations. Elephants are known to occupy large home ranges which transverse borders on annual or seasonal basis through migratory routes, linking protected landscapes such as Amboseli, Tsavo, west Kilimanjaro -Lake Natron ecosystem and parts of the Serengeti-Mara ecosystem. This cross-border connectivity is critical for elephants' access to forage and water and for the long-term persistence of viable populations. However, collaboration between Kenya and Tanzania on trans boundary elephant conservation and management has been majorly informal (Omondi *et al.*, 2023), and mostly has been with Non-Governmental organisations with partial Government involvement. The informal collaboration has been a main cause of transboundary elephant conservation disputes, being exacerbated by differing wildlife conservation policies and laws between Kenya and Tanzania. The wildlife policy in Tanzania allows trophy or sport hunting of elephants, while Kenya banned trophy hunting in 1970s. In few years, there have been allegations from Kenyan conservation NGOs on killing of big tuskers belonging to Kenya by trophy hunters

in Tanzania in Longido district (Poole *et al.*, 2024), disputes that were rebutted by Tanzania (Keyyu *et al.*, 2025). These disputes and informal cross border conservation wrangles have jeopardized collaboration between the two countries at all levels. Hence, there is need of these countries to enter into formal collaboration as well as specific agreements which will culminate into development of transboundary ecosystem management plans.

Transboundary movement of elephant between Northern Tanzania and Southern Kenya is very important for wildlife conservation. This is because animals and humans live in large and rich environments and need to move/ navigate in order to survive, and that animal movement is mainly driven by need for resources (mainly fodder/ food, water, cover), and other biological needs. Moreover, no single habitat has all the resources and other requirements every time and throughout the year, therefore movement ensures resources availability whether it is local, national or cross-border (international/ transboundary). The borderland between northern Tanzania and southern Kenya form a dynamic transboundary landscape that support large, mobile elephants' populations; and the elephant population is increasing. This cross-border connectivity is critical for elephants' access to forage and water and for the long-

term persistence of viable populations. Therefore, the main objective of this study was to develop a shared cross-border vision on science-driven efforts in elephant and other species conservation in order to guide informed conservation decisions. The specific objectives of the study were to establish and verify the current status of transboundary movements of elephants between northern Tanzania and southern Kenya borderland; determine the nature, type and magnitude of elephant movement and lastly to create awareness on transboundary elephant and other species management for informed decision making among stakeholders.

MATERIALS AND METHODS

Study area

The study was conducted in the west Kilimanjaro - lake Natron landscape, covering an area of approximately 10,061 km², which is part of the Maasai Steppe region in Tanzania. The landscape is located between 1° 37' to 3° 13' South and 35° 49' to 38° 00' East (Fig. 1). The landscape spans the northern western regions of Kilimanjaro extending eastward from Arusha National Park to lake Natron (TAWIRI, 2025).

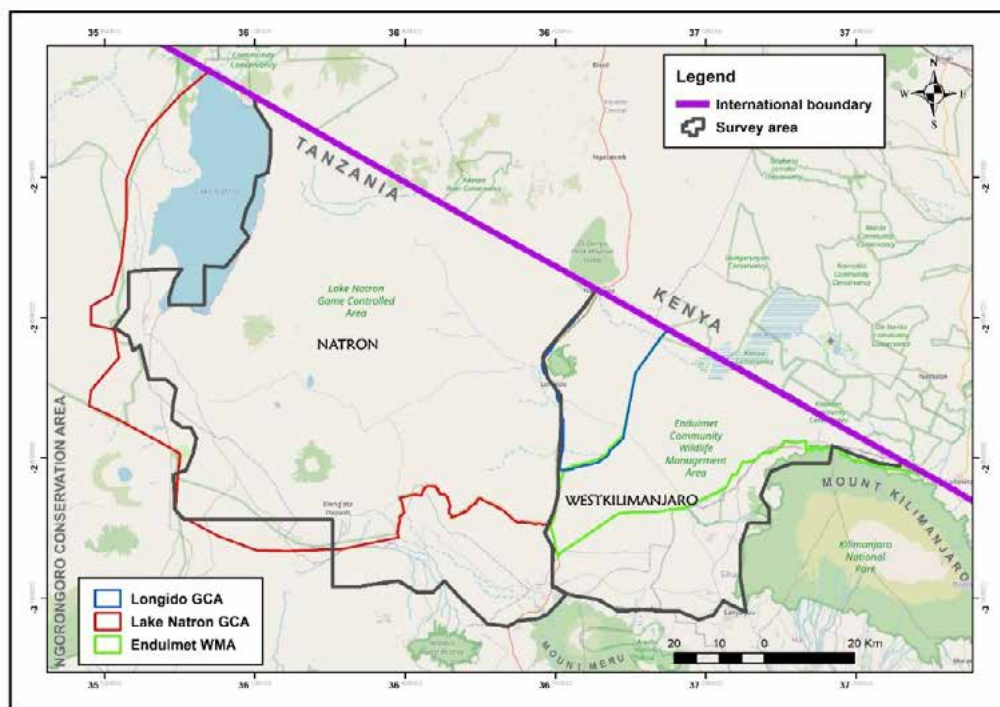


Figure1: Map of the study area, the west Kilimanjaro- Lake Natron landscape and Tanzania-Kenya borderland

The lake Natron area, covering 7,047 km², is defined to the north by the Tanzania-Kenya border, extending from the border town of Namanga in the east and continuing northwest to the northern tip of lake Natron. The western boundary runs along the eastern shore of lake Natron and continues south following the eastern border of the Ngorongoro Conservation Area. To the south, the boundary extends from the southeastern corner of the Ngorongoro Conservation Area eastward to the northwestern corner of Arusha National Park (TAWIRI, 2025).

The West-Kilimanjaro area, covering 3,014 km², is within the Longido and Siha districts of the Arusha and Kilimanjaro regions, respectively. Its northern boundary extends to the Tanzania-Kenya border, stretching southeast from Namanga to Irkaswa village. The eastern boundary is defined by the Kilimanjaro National Park, extending southward near the Sanya Juu community. To the south, the rea's boundary stretches from west of Sanya Juu to the northeastern corner of Arusha National

Park, then follows the park's northern border until it meets the Arusha-Nairobi Road, which forms the western boundary of the West-Kilimanjaro area (TAWIRI, 2025).

Climate, Soil, Vegetation and Socio-economic activities

The west Kilimanjaro - Lake Natron landscape experiences a semi-arid to sub-humid climate, characterized by distinct wet and dry seasons. Rainfall is highly variable, with annual precipitation ranging from 300 mm in the drier lowlands to over 1,200 mm in higher elevations near Kilimanjaro. The area has two main rainy seasons: the long rains from March to May and the short rains from October to December.

The West Kilimanjaro highlands are characterized by fertile volcanic soils, primarily Andosols, which are derived from volcanic ash deposits. These soils are loamy to silty in texture, well-drained, and rich in organic matter, making them highly suitable for agriculture, especially crops like coffee and

maize (Zinke *et al.*, 2023). Moving into the mid-elevation zones, including areas around Longido and the Ngorongoro foothills, the soils transition to Luvisols, Cambisols, and occasionally Vertisols in wetter depressions. These soils range from sandy loam to clay and offer moderate fertility, though they are more vulnerable to erosion and nutrient depletion if not managed well (FAO, 2001; IUSS Working Group WRB, 2022). In contrast, the Lake Natron basin presents a lowland arid environment dominated by Solonchaks and Calcisols—saline and alkaline soils that form under high evaporation and minimal leaching. These soils have a silty to clayey texture and poor drainage, often forming salt crusts and alkaline flats near the lake. With very low fertility and a pH often exceeding 8.5, the area supports only salt-tolerant vegetation and is largely unsuitable for conventional farming, posing significant land use constraints (Jones *et al.*, 2013).

The landscape features diverse vegetation types influenced by altitude, rainfall, and soil conditions. In the lower-lying, drier areas around lake Natron and west Kilimanjaro, vegetation consists mainly of open grasslands, acacia woodlands, and semi-arid scrublands. The middle elevations support savannahs dominated by open woodlands (*Vulcheria* spp and *Commiphora* spp), while higher elevations near Kilimanjaro transition into montane forests. Wetlands and seasonal riverine vegetation are found along water sources, providing critical habitats for wildlife and livestock. The main socio-economic activity in the area is pastoralism where the Maasai and other indigenous communities rely on livestock herding, particularly cattle, goats, and sheep, as their primary livelihood. Grazing areas extend westward from the Kiserian-Mriata Ridge (on the eastern side of the study area) extending westward encompassing the grasslands adjacent to Gelai (2,942m) and Kitumbeine (2,858m) mountains. Small scale agriculture is also practiced in areas with relatively higher rainfall, such as parts of Siha and Longido districts, where communities

practice subsistence farming, growing crops such as maize, beans, and vegetables (TAWIRI, 2025).

METHODS

Monitoring of elephant movement in northern Tanzania-southern Kenya borderland

Movement of elephants in the area was collected by fitting 13 elephants with GPS collars. The distribution of collars among elephants and study areas are shown on Table No. 1. Immobilization and collaring of elephants involved a team from a range of wildlife stakeholders in the study area. The first step involved searching for candidate elephants in the area using a spotter plane (Cessna 182 with registration number 5H-TWR) to help locate elephant herds and immobilization was subsequently conducted by darting the elephant using a ground crew that composed of a wildlife ecologist, driver and a certified wildlife veterinarian. The team followed closely the spotted elephant until a conducive environment prevailed to enable safe immobilization including but not limited to terrain, habitat type, etc. The veterinarian immobilized the selected elephants using Etorphine Hydrochloride (M99: C-vet UK) in darts fired from a modified .22-caliber dart gun. When an elephant was immobilized and became recumbent, it was fitted with a Vertex Plus-13 (+2) Iridium elephant GPS satellite collar unit using standard procedures. Whilst on the ground, the elephant was monitored for heart rate, respiration and pulse rate, measurements of body parameters and sampling of blood and tail hairs was also conducted for storage in the wildlife specimen bank. When all activities were completed, the effect of the immobilizing drug was reversed using Diprenorphine (M5050: C-vet UK) to revive the animal and monitored until full recovery and moving. Datasets from the collared animals was remotely downloaded (hourly), hosted and analyzed at the Information Technology (IT) facility that was established at TAWIRI Head office.

Monitoring of collared elephants was done remotely at the collar manufacturer's website at <https://www.vectronic-wildlife.com/> and GPS X Plus software was installed on the project computers. The website displayed a maximum of 2500 hourly fixes and hence datasets exceeding that amount was downloaded through GPS X Plus and mapped separately using Arc Map GIS software. From time to time, datasets from the collared animals were

remotely downloaded (hourly fixes) using GPS X Plus software and hosted at the Information Technology (IT). The established movement database was linked to Arc Map software which enabled further processing/mapping of individual elephant movement patterns. Collared elephants were monitored remotely and also by ground truthing on quarterly basis from 2020 to 2025.

Table 1: Collaring locations of 13 elephants in the landscape in northern Tanzania

SN	Collar ID	Sex	Latitude	Longitude	District	Year of deployment
1	34236	Male	-2.554960	36.717960	Longido	2020
2	34237	Female	-2.517990	36.671350	Longido	2020
3	34238	Male	-2.552860	36.782780	Longido	2020
4	34240	Female	-2.558520	36.696030	Longido	2020
5	34242	Male	-2.897510	36.744640	Longido	2020
6	34244	Male	-2.995290	36.917490	Longido	2020
7	34245	Male	-2.519910	36.671470	Longido	2020
8	34247	Male	-2.921390	36.915910	Longido	2020
9	34248	Male	-2.624790	36.555320	Longido	2020
10	35215	Male	-2.668998	36.583732	Longido	2020
11	35216	Male	-2.900350	36.921240	Longido	2020
12	35220	Female	-2.731175	36.694986	Longido	2020
13	35223	Male	-2.756400	37.031090	Longido	2020

RESULTS

Mapping movement of 13 collared elephants revealed active movement throughout northern Tanzania especially in Longido district, nearby districts as well as movements across the border to southern Kenya. Elephant movement patterns confirmed elephant active movements between Kilimanjaro National Park and Lake Natron GCA, Enduimet WMA and Lake Natron GCA, Arusha and Kilimanjaro National Parks, Longido-Lake Natron GCA and Tarangire-Manyara Ecosystem, Tarangire-Manyara Ecosystem and southern Kenya, and

also between Longido-Lake Natron GCA and Amboseli National Park in Kenya.

Combined or overall movement pattern of 13 collared elephants is shown in Fig. 2. Generally, movement pattern of individual elephants varied greatly with some elephants having wide or large-scale movement patterns than others. One bull (ID 34240) showed large-scale movement in the landscape to the Tarangire-Manyara ecosystem (Fig 3). Movements in the borderland between northern Tanzania and southern Kenya were highly revealed. One elephant (ID 34237) spent most of the time

in southern Kenya and was killed at Kajiado during human-elephant conflict management in 2021 (Fig 4). One elephant (ID 34238) spent all of the time in northern Tanzania and did not go to Kenya at all in five years (Fig. 5). Fig 6: One of the seven elephants (ID 35215) spent most of the time in northern Tanzania and very little time in southern Kenya (Fig. 6). Analysis of the twelve remaining elephants revealed that a total of eleven elephants (91%) crossed the border to southern Kenya at least once during the monitoring period. A total of seven elephants (72%) spent most of their time in northern Tanzania. Overall, collared elephants

spent 78% of their time in Tanzania and 22% of their time in Kenya.

Movement of elephants was also recorded within the two popular corridors in northern Tanzania in Longido district namely the Tanganyeeti/Kikoti and Laasarak wildlife corridors. Ground truthing of movements to locate collared elephants also revealed a number of human activities in the landscape especially movement corridors including human settlement, farming/cultivation, charcoal making, and livestock grazing.

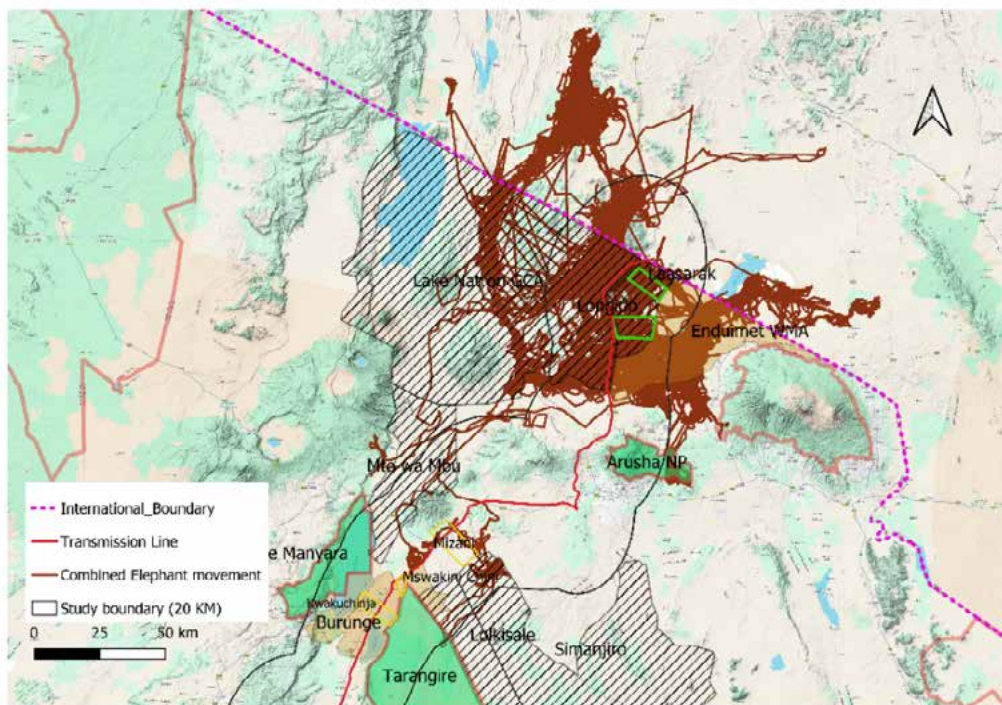


Figure 2: Combined movement pattern of all 13 collared elephants in the northern Tanzania and southern Kenya borderland

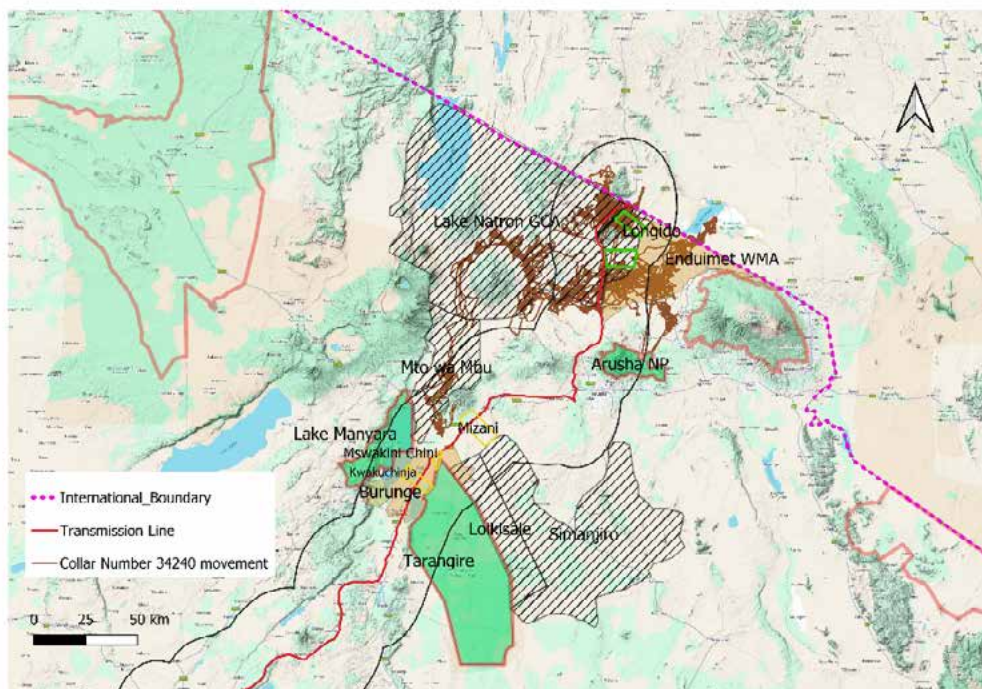


Figure 3: An elephant bull (ID 32240) that showed wide or large-scale movement pattern in the landscape to Tarangire-Manyara ecosystem

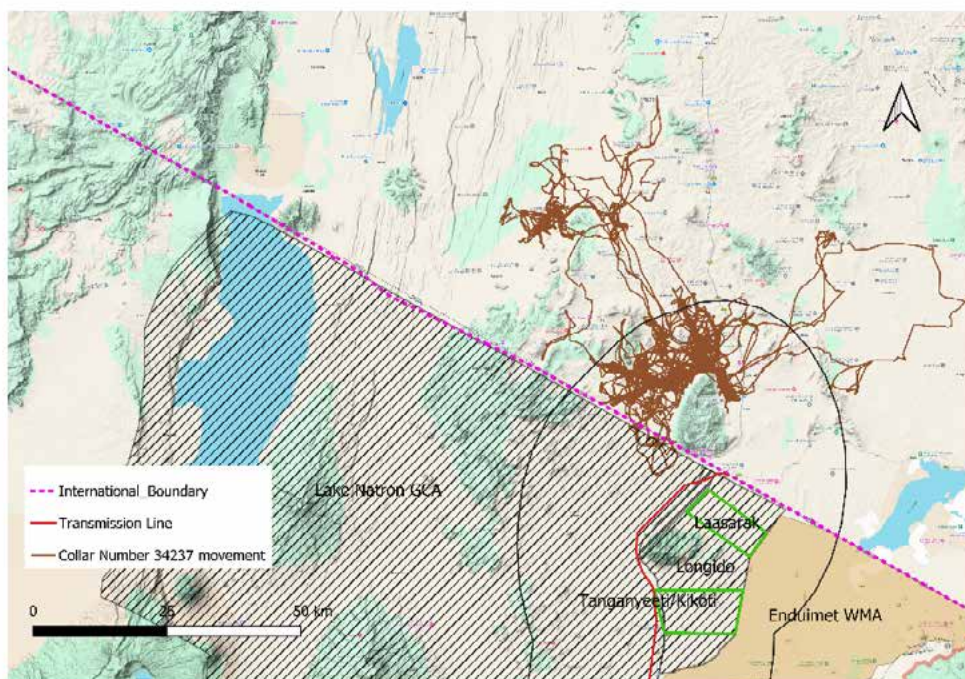


Figure 4: A female elephant (ID 34237) that spent most of the time in Kenya and was killed in Kajiado district in 2021 during human-wildlife conflict management

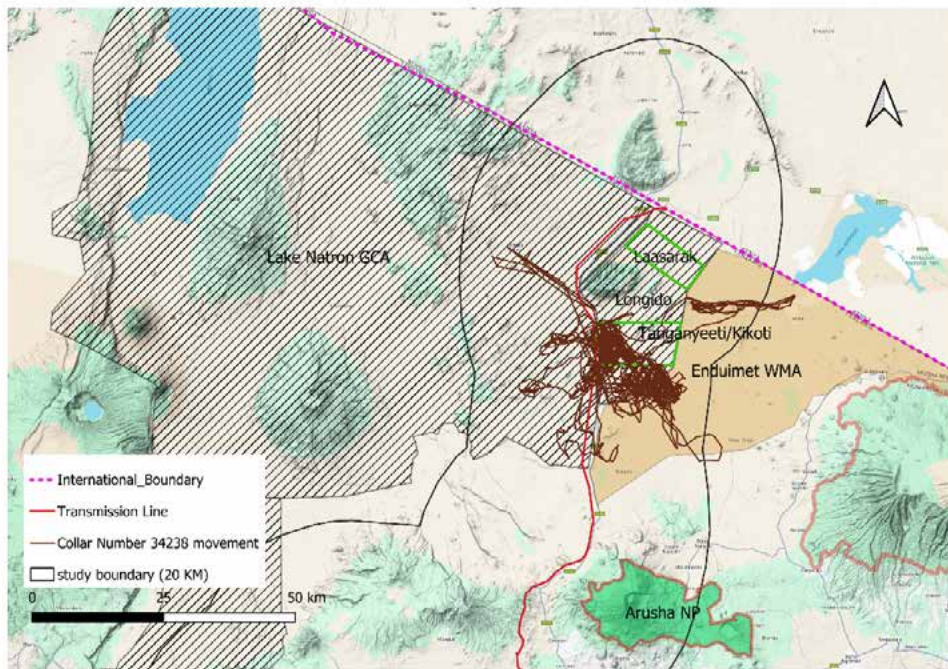


Figure 5: An elephant (ID 34238) that spent all of the time in northern Tanzania and did not go to Kenya at all in five years

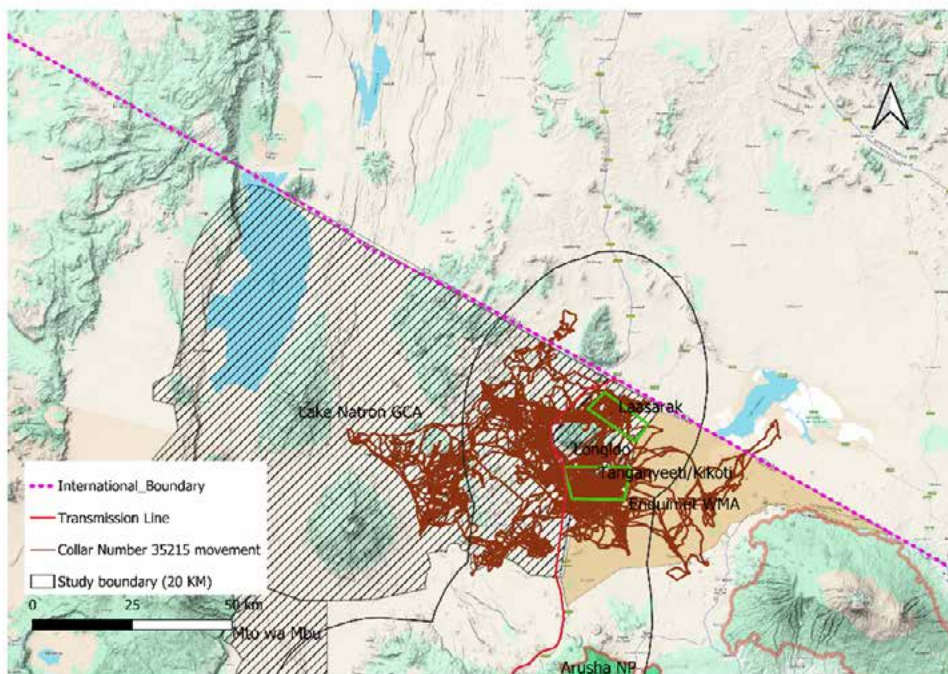


Figure 6: One of the seven elephants (ID 35215) that spent most of the time in northern Tanzania and very little time in southern Kenya

Home range of elephants

Home range of individual elephants is shown on Table 2. The home range of individual elephants varied greatly among collared elephants. Elephant with collar ID 35223 had

the largest home range of 16,570km² while elephant with collar ID 34238 had the smallest home range of 2,460km². Generally male elephants showed the largest average home range than female elephants.

Table 2: Home range and hourly fixes of collared elephants in the landscape

SN	Collar ID	Sex	Home range (km ²)	Hourly fixes
1	34236	Male	7,037	6,177
2	34237	Female	2,271	13,031
3	34238	Male	3,871	2,460
4	34240	Female	5,385	13,713
5	34242	Male	1,664	8,370
6	34244	Male	4,556	16,454
7	34245	Male	2,008	6,391
8	34247	Male	3,383	10,012
9	34248	Male	5,089	15,710
10	35215	Male	2,763	15,851
11	35216	Male	11,720	11,591
12	35220	Female	2,786	13,982
13	35223	Male	4,618	16,570

DISCUSSION AND CONCLUSION

The study has shown that there is high transboundary movement of elephants between the northern Tanzania and southern Kenya borderland, and that eleven out of thirteen elephants (91%) that were collared in northern Tanzania crossed the border to southern Kenya at least once during the monitoring period. The study has shown that most of the movements were especially between Lake Natron GCA, Longido GCA and Enduimet WMA in northern Tanzania and Amboseli National Park in southern Kenya. The study has shown great importance of safeguarding this transboundary ecosystem for biodiversity

conservation. However, it is worth to note that, although transboundary landscapes are critical tools for protecting ecosystems and ecological processes that transcend national jurisdictions, they are challenging to create or manage due to the differences in governance contexts and capacity and power dynamics among countries (Horigue *et al.*, 2024). This is even more complicated or challenging where bordering countries have different conservation policies and legislations. Therefore, there is a great need to strengthen and harmonize transboundary collaboration between northern Tanzania and southern Kenya in order to enhance species conservation.

The study has also shown that Tanganyeeiti/Kikoti wildlife corridor facilitate movement of elephants to several protected areas including Kilimanjaro National Park on the eastern boundary, Arusha National Park to the south, and Amboseli National Park in southern Kenya (which is about 20 km north of the Tanzania-Kenya border). Additionally, this important wildlife corridor also provides links to two private conservation areas in Tanzania namely West Kilimanjaro Ranch and Ndarakwai Ranch, Longido GCA, and Ngasurai Open Area. Therefore, this wildlife corridor should be given a priority in corridor conservation and restoration activities due to its importance in facilitating wildlife movement in the landscape, together with Laasarak wildlife corridor that is close to the Namanga border. Unfortunately, the Tanganyeeiti wildlife corridor is currently threatened by increasing human activities especially livestock grazing, human settlement and subsistence agriculture (TAWIRI, 2020). There is a great need to gazette this important wildlife corridor as it also facilitates wildlife movement to other ecosystems including the Serengeti ecosystem and Tarangire-Manyara ecosystem.

The study has shown that the home range of elephants varied greatly among individuals, with large movements of elephants during the dry season and small movements per day and home range during the wet season. Ground truthing and habitat suitability data has also shown that important ecological drivers for elephant movement were mainly fodder and water. Therefore, bordering countries should improve water availability as well as increase size of protected areas where possible in order to increase forage availability in the landscape. Increasing the size of protected areas will also reduce human-elephant conflicts as most elephants might spent more of their time inside protected areas. The killing of one collared elephant in a farm during human-wildlife elephant conflict management at Kajiado in Kenya was an indication that elephants sometimes enter village land in search of forage or water.

The study has also shown increasing human activities in some of the wildlife corridors especially the Tanganyeeiti corridor, and collared elephants moved fast at night when crossing human dominated landscapes. These anthropogenic activities, coupled with the increased human population are a major threat facing elephants and other species within corridors. In spite of the fact that communities in Longido district are pastoralists, there is a great need to embark on development of village land use plans so that areas used for wildlife movement in villages are set as wildlife corridors, especially at this stage where investors and immigrants have started securing land in those villages and processing land rights or title deeds, which will complicate future gazettelement of wildlife corridors.

This study has shown that most of the collared elephants spent most of their time in northern Tanzania than in southern Kenya. This indicates that landscapes in northern Tanzania have more resources (forage and water) or safe for elephants than other areas. However, these results should be interpreted with caution until results from elephants collared in southern Kenya are available, as collared elephants for this study might be resident to Tanzania. Results from this study have indicated the need for cross-border management of elephants and other species between northern Tanzania and southern Kenya. However, transboundary wildlife management has been associated with a number of obstacles including issues of fit, particularly differences in environmental research capacity, socioeconomic contexts, internal institutional arrangements, capacity for ecological research and conservation, ability to adjust and update or share data, conservation costs, jurisdictional differences, engagement of multiple levels of organization, and differences in legal bases and policy (Horigue *et al.*, 2024). Transboundary management also results into specific spatial and temporal scales (Cumming *et al.*, 2006; Soga & Gaston, 2020), where spatial scales and boundaries often relate to jurisdictions where governments can exercise their authority over their territory (Cumming

et al., 2006). Moreover, political boundaries expose contiguous ecosystems to a wide range of governance regimes, structures, and policies and different activities based on people's social, economic, and cultural interests (Arrow *et al.*, 1996, Cash *et al.*, 2006; Cumming *et al.*, 2006). This separation has led to conflicts across nations, peoples, and communities, including resource-use competition, boundary disputes, and less effective management of threats that originate from outside territories (Sandwith *et al.*, 2001; Schoon, 2011). In order to avoid transboundary wildlife movement disputes or obstacles, The UN resolution 1803 (XVII) of 14 December 1962 on Permanent Sovereignty over Natural Resources (UN-PSNR, 2026; Brinker, 2026) is key in resolving transboundary wildlife movement and management disputes. The resolution or declaration confers states and their people the right to own, control/ manage, use, and freely to dispose of their natural resources/ wealth within their borders. Therefore, when elephants or other wildlife species have crossed an international border into another country or state, their management will be governed by the Wildlife Policy, Legislations, Regulations and Guidelines of that particular country; unless a bilateral agreement has been established and agreed upon by partner states (Keyyu *et al.*, 2025), or where bordering countries have agreed otherwise or developed a cross-border or transboundary protocol for collaboration. Generally, in spite of challenges for transboundary or cross border wildlife movement and management, there is still great optimism and opportunities for partner states to collaborate in transboundary wildlife management.

CONCLUSIONS

It is concluded that there is high cross-border movement of elephants between northern Tanzania and southern Kenya. Therefore, there is a great need to strengthen cross-border collaboration for landscape and species conservation. Also, every country should improve water availability in the landscapes

as well as increase the size of protected areas (set more protected areas) in order to improve resources availability.

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MONITORING OF WILDLIFE COMPOSITION, ABUNDANCE AND DIVERSITY IN RELATION TO HABITATS IN MSOLWA SECTOR NYERERE NATIONAL PARK, TANZANIA

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ABSTRACT

Camera trapping is an objective and non-invasive technique that generates substantial data on the presence or absence of wildlife species, as well as on anthropogenic activities. Four grids, each containing 30 stations, were developed using ArcGIS version 10.8 software. In every station, a single camera trap was deployed resulting in a total of 120 cameras in the Msolwa sector. The cameras were deployed for over 60 days between September and November 2024 to maximize detection, including the most elusive species. PAST Version 4.03 software was used to calculate diversity indices (i.e. Shannon-Wiener, Simpson's Index) and relative abundance. The results showed that a total of 6,549 images were recorded representing 16 orders, 35 families, and 55 species. The open woodland exhibited high species diversity ($H' = 3.06$), then wooded grassland ($H' = 2.69$) and grassland habitat ($H' = 0.61$). Furthermore, elephants exhibited the highest naïve occupancy at a value of 0.60, followed by both common duiker and warthog who had the naïve occupancy value of 0.50. Illegal human activities, including wildlife poaching and unauthorized fishing were recorded during the survey. The study concludes that diverse habitats in the Msolwa sector support different mammal species. Therefore, the need for strengthening conservation efforts is important for improving biodiversity and making the area an important tourist destination within the park.

Keywords: Camera trapping, Composition, Diversity, Naïve occupancy, Species richness

INTRODUCTION

Camera traps have become the most useful non-invasive tools for studying wildlife ecology (Oliver *et al.*, 2023). They are essential when research involves elusive species inhabiting different habitats, which are difficult to access (Sittinger *et al.*, 2024). The data from camera traps are used to estimate animal abundance, understanding animal movement, assess species diversity, richness, distribution as well as exploring species presence/absence (Bassing *et al.*, 2023). This information provides critical insight into wildlife population trends, particularly in newly established or recently upgraded protected areas (Davis *et al.*, 2023).

Tanzania has several areas, with different conservation status, designated for protection of natural resources (Ract *et al.*, 2024). These include but are not limited to, National Parks, Game Reserves, Wildlife Management Areas, Conservation area and Forest Reserves (Ma *et al.*, 2024). National parks hold the strictest conservation status where human activities are highly restricted with the primary exception of photographic tourism (Mkwizu, 2024). Over time, some protected areas with minimal conservation status have been upgraded to National parks, thus receiving stricter Governmental control (Mi *et al.*, 2023). This modification significantly enhances wildlife conservation efforts, improves habitat

protection, and provides greater public access for photographic tourism while limiting potentially exploitive activities like hunting.

Nyerere National Park (NYENAPA) is one of the notable examples, which was established in 2019 after the upgrade of the former Selous Game Reserve (Foya *et al.*, 2023). This upgrade shifted the focus from consumptive use to non-consumptive use of wildlife resources (Foya *et al.*, 2023). As part of ongoing conservation and species monitoring efforts, Tanzania Wildlife Research Institute (TAWIRI) conducted a camera trap survey in Msolwa area, aiming at gathering data on biodiversity and providing valuable insights into the presence and absence of various species, thereby acting as the pioneering step for future conservation strategies, management practices, and tourism development potentials.

METHODOLOGY

Study Area Description

The camera trap survey was conducted at Msolwa sector, a part of NYENAPA, covering about 4,620 km² out of 30,893 km². Msolwa sector is located between 9°0'S and 37°24'E and receives annual rainfall ranging between 750 and 1300 millimeters. The area covers about 4,620 km² out of 30,893 km² of the total area of NYENAPA. The area is located between 9°0'S and 37°24'E with annual rainfall ranging between 750 and 1300 millimeters.

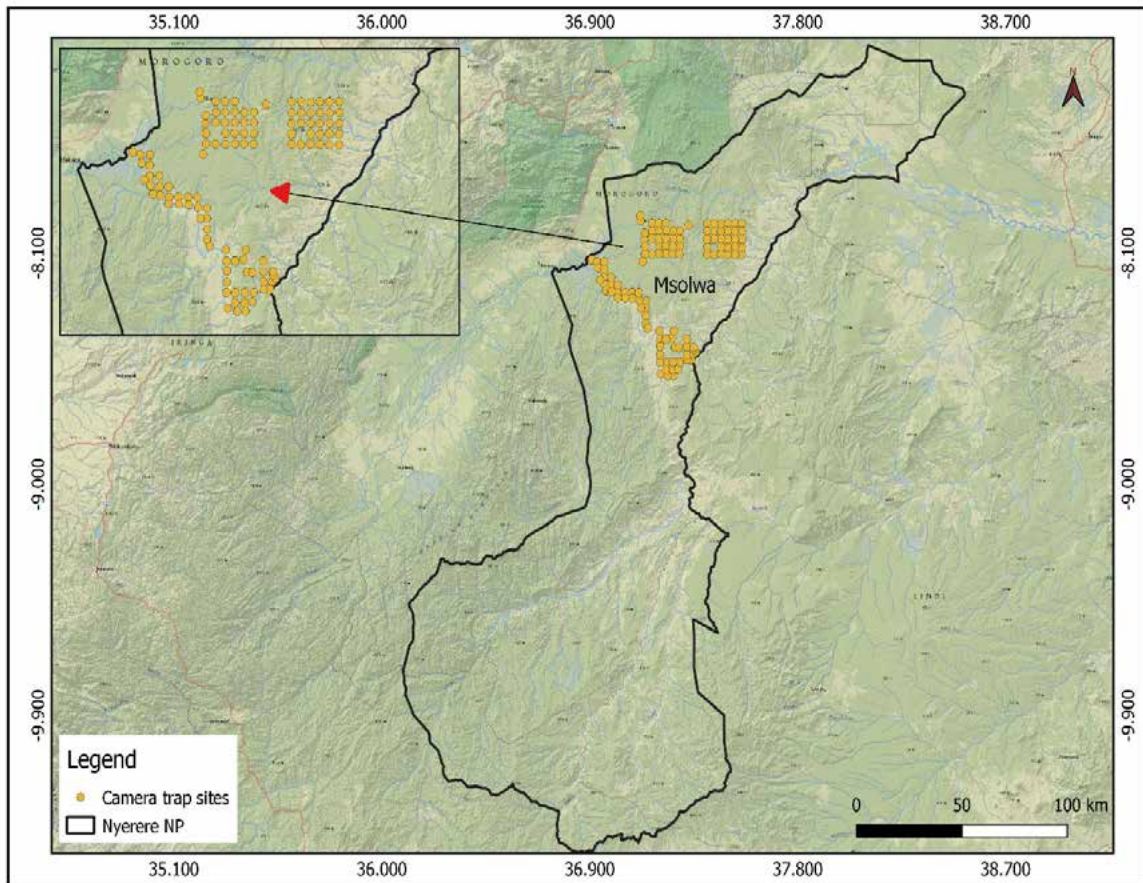
Msolwa area is dominated by extensive miombo woodland characterized by *Azelia quanzensis* and *Brachystegia spp* (Zhang *et al.*, 2024). This landscape is like an isolated island surrounded by two major rivers; Ruaha river to the East and North, and Kilombero river to the West and South (Tarimo *et al.*, 2024). Together, the two rivers serve as crucial water sources

for wildlife. The riparian influenced vegetation along these rivers ensures a year-round supply of quality forage for herbivores which in the end sustain the predators' population.

Msolwa area is home to a diverse of game species, including species such as lions (*Panthera leo*), leopards (*Panthera pardus*), wild dogs (*Lycaon pictus*), African elephant (*Loxodonta africana*), range of antelopes and buffalo (*Syncerus cafer*), together with several other bird species (John *et al.*, 2024). The area was renowned as a premier trophy-hunting destination in Africa.

Data collection

The survey was conducted between September and November 2024. Using ArcGIS version 10.8 software, the study area was divided into six systematic grid-based camera trap layouts with uniform 3km spacing. Expansion of the Nyerere Hydropower Dam resulted in the submerge of two grids. We then restructured the study area to remain with four grids, prioritizing habitat diversity and expected wildlife activity (Map. 1). A total of 30 stations were established within each grid, with a single camera trap deployed at each station, resulting in the installation of 120 camera traps. Two types of camera traps, i.e., Browning Dark Ops Max Plus Trail Camera (BTC-6HD-MXP) and Cuddeback 20MP IR Model H-1453. They remained operational for 24 hours with a capture interval of 30 seconds between consecutive photographs for 60 days. Prior to setting up camera traps, a reconnaissance survey of the study area was conducted to collect data on indirect evidence of mammalian species. In each grid, based on data on indirect evidence of mammalian species, a location was selected for camera trap deployment to increase the likelihood of detection.



Map 1: Map showing location of deployed cameras in the Msolwa Sector

DATA ANALYSIS

Upon removal of camera traps from the field, downloading of images from the SD cards was performed followed by extraction of metadata from the downloaded images by using a exifPro script in R-studio version 4.3.2. Captured species from the downloaded images were further identified and additionally cleaned in an excel sheet. Each species was checked for its conservation status using the International Union for Conservation of Nature (IUCN) database. Some animals could not be identified due to either a poor angle image captured, the animal was too far or moved too fast before was captured. Diversity indices of animal species obtained in the surveyed habitats were computed using PAST Version 4.03. PAST was used because it features a user-friendly interface for ecological data analysis and includes key

biodiversity metrics such as Shannon-Wiener, Simpson's Index, and rarefaction curves. It is much accepted in wildlife ecology research and has been used in numerous published studies. The graphs were summarized and drawn using Microsoft Excel for window 2021. Besides wild animals, anthropogenic activities were also observed following the captured images of humans doing poaching activities in the camera traps.

Relative Abundance Index (RAI), a measure of the number of captures of a species per 100 days of camera trapping was calculated as described by (Martin, 2023). Additionally, Naïve occupancy of each species, is the proportion of the study area occupied by the given species (Martin, 2023).

RESULTS

Camera traps

Out of 120 installed cameras, 107 (89%) were active throughout the study duration. Six units were stolen, 2 destroyed by elephants, and 5 cameras were inactive, making a total of 13 (11%) cameras that had no data.

Species composition

A total of 6,549 wildlife photographs consisting of 16 orders, 35 families and 55 (43 mammals and 12 avian) species were recorded. Order Cetartiodactyla had the highest recorded frequency of wild exposures. Hippopotamus (*Hippopotamus amphibious*) had the highest frequency of records with 1,460 exposures while eight bird species including Striped kingfisher (*Halcyon chelicut*), Superb starling (*Lamprotornis superbus*), Crowned hornbill (*Lophoceros alboterminatus*), White-browed Scrub-robin (*Cercotrichas leucophrys*), Little bee-eater (*Merops pusilus*), White-bellied-bustard (*Eupodotis senegalensis*), Small eared greater galago (*Otolemur garnettii*) and Angolan Black and White colobus monkey (*Colobus angolensis*) appeared with only one exposure (Table 1; Appendix 2). The endangered species African Wild dog was also recorded during the survey with 16 exposures.

The highest abundance of species (3,049 individuals) was recorded in bushland habitat;

likely, due to its diverse vegetation structure providing food and cover, and the least was closed woodland with 22 individual species (Figure 1). Anthropogenic activities were recorded, including poaching of game meat and illegal fishing within riparian and woodland habitats.

Species richness and diversity

Open woodland exhibited both highest species diversity ($H'=3.06$) and richness (6.237), followed by wooded grassland ($H'=2.697$), where grassland had the least species diversity ($H'=0.619$). Wild animals were evenly distributed in closed woodland habitat compared to the rest of habitats (Table 1).

RAI and naïve occupancy

Table 2 provides the RAI and Naïve occupancy values of wildlife species recorded during camera survey. Elephant exhibited the highest naïve occupancy at a value of 0.60, followed by both Common duiker and warthog who had the naïve occupancy value of 0.50. These values represent the proportion of camera trap stations where the respective species was captured relative to the total number of camera stations sampled. A value closer to 1 signifies a larger presence of the species across the sampled area, providing insight into the extent of their occurrence.

Table 1: Summary of wildlife Taxas captured during surveys, with species diversity (H'), richness (Margalef) and evenness (e^H/S) across each surveyed habitat.

	Bushland	Closed Woodland	Grass-land	Open woodland	Riparian	Wet land	Wooded Grassland
Taxa_S	13	5	10	49	37	7	32
Individuals	70	22	98	2199	3049	504	607
Shannon_H'	1.688	1.364	0.6195	3.06	2.03	0.723	2.697
Evenness_e^H/S	0.4161	0.782	0.1858	0.4355	0.2058	0.2944	0.4638
Margalef	2.825	1.294	1.963	6.237	4.487	0.9642	4.837

Table 2: List of species captured in the Msolwa Sector with number of capture locations, Relative Abundance Index and the naïve occupancy for each species.

COMMON NAME	SCIENTIFIC NAME	NUMBER OF CAPTURES	NUMBER OF LOCATIONS	RAI	NAÏVE OCCUPANCY
Aardvark	<i>Orycteropus afer</i>	30	19	50	0.18
African buffalo	<i>Syncerus caffer</i>	589	35	981.67	0.33
African Civet	<i>Civettictis civetta</i>	100	25	166.67	0.23
African gray hornbill	<i>Tockus nasutus</i>	2	1	3.33	0.01
African savannah hare	<i>Lepus victoriae</i>	11	6	18.33	0.06
African Wild Dog	<i>Lycaon pictus</i>	16	5	26.67	0.05
Angolan Black and White colobus monkey	<i>Colobus angolensis</i>	1	1	1.67	0.01
Banded Mongoose	<i>Mungos mungo</i>	13	5	21.67	0.05
Little bee-eater	<i>Merops pusilus</i>	1	1	1.67	0.01
Bushbuck	<i>Tragelaphus scriptus</i>	120	18	200.00	0.17
Bushpig	<i>Potamochoerus larvatus</i>	64	18	106.67	0.17
Bushy tailed mongoose	<i>Bdeogale crassicauda</i>	47	19	78.33	0.18
Cape hare	<i>Lepus capensis</i>	133	25	221.67	0.23
White-browed Scrub-robin	<i>Cercotrichas leucophrys</i>	1	1	1.67	0.01
Common duiker	<i>Sylvicapra grimmia</i>	31	54	51.67	0.50
Crested guineafowl	<i>Guttera pucherani</i>	77	10	128.33	0.09
Crowned hornbill	<i>Lophoceros alboterminatus</i>	1	1	1.67	0.01
Dwarf Mongoose	<i>Helogale parvula</i>	5	2	8.33	0.02
Egyptian goose	<i>Alopochpa aegyptiaca</i>	4	1	6.67	0.01
Eland	<i>Tragelaphus oryx</i>	76	20	126.67	0.19
African Savannah elephant	<i>Loxodonta africana</i>	449	64	748.33	0.60
Superb starling	<i>Lamprotornis superbus</i>	1	1	1.67	0.01
Greater Kudu	<i>Tragelaphus strepsiceros</i>	2	2	3.33	0.02
Hartebeest	<i>Alcelaphus lichtensteinii</i>	289	38	481.67	0.36
Helmeted guinea fowl	<i>Numida meleagris</i>	55	9	91.67	0.08
Hippopotamus	<i>Hippopotamus amphibius</i>	1460	36	2433.3	0.34
Honey badger	<i>Mellivora capensis</i>	16	7	26.67	0.07

COMMON NAME	SCIENTIFIC NAME	NUMBER OF CAPTURES	NUMBER OF LOCATIONS	RAI	NAÏVE OCCUPANCY
Human	<i>Homo sapiens</i>	6	3	10.00	0.03
Impala	<i>Aepyceros melampus</i>	956	32	1593.3	0.30
Kirk's dik-dik	<i>Madoqua kirkii</i>	2	1	3.33	0.01
Large-spotted Genet	<i>Genetta maculata</i>	31	15	51.67	0.14
Leopard	<i>Panthera pardus</i>	66	19	110.00	0.18
Lesser kudu	<i>Tragelaphus imberbis</i>	4	1	6.67	0.01
Lion	<i>Panthera leo</i>	15	10	25.00	0.09
African giant pouched rat	<i>Cricetomys ansorgei</i>	19	1	31.67	0.01
Marsh mongoose	<i>Atilax paludinosus</i>	21	9	35.00	0.08
Miombo genet	<i>Genetta angolensis</i>	13	5	21.67	0.05
Olive baboon	<i>Papio anubis</i>	8	8	13.33	0.07
Temmincks Ground Pangolin	<i>Smutsia temminckii</i>	2	2	3.33	0.02
Crested Porcupine	<i>Hystrix cristata</i>	23	10	38.33	0.09
Red billed oxpecker	<i>Buphagus erythrorhynchus</i>	2	1	3.33	0.01
Red-natal duiker	<i>Cephalophus natalensis</i>	26	6	43.33	0.06
Sable antelope	<i>Hippotragus niger</i>	10	6	16.67	0.06
Serval Cat	<i>Leptailurus serval</i>	5	5	8.33	0.05
Small-eared greater galago	<i>Otolemur garnettii</i>	1	1	1.67	0.01
Southern ground hornbill	<i>Bucorvus leadbeateri</i>	28	21	46.67	0.20
Spotted Hyena	<i>Crocuta crocuta</i>	206	44	343.33	0.41
Vervet Monkey	<i>Chlorocebus pygerythrus</i>	35	10	58.33	0.09
Warthog	<i>Phacochoerus africanus</i>	339	53	565.00	0.50
Waterbuck	<i>Kobus ellipsiprymnus</i>	158	24	263.33	0.22
White bellied bustard	<i>Eupodotis senegalensis</i>	1	1	1.67	0.01
Wildebeest	<i>Connochaetes taurinus</i>	124	11	206.67	0.10
Striped kingfisher	<i>Halcyon chelicut</i>	1	1	1.67	0.01
Yellow baboon	<i>Papio cynocephalus</i>	391	47	651.67	0.44
Zebra	<i>Equus quagga</i>	182	28	303.33	0.26

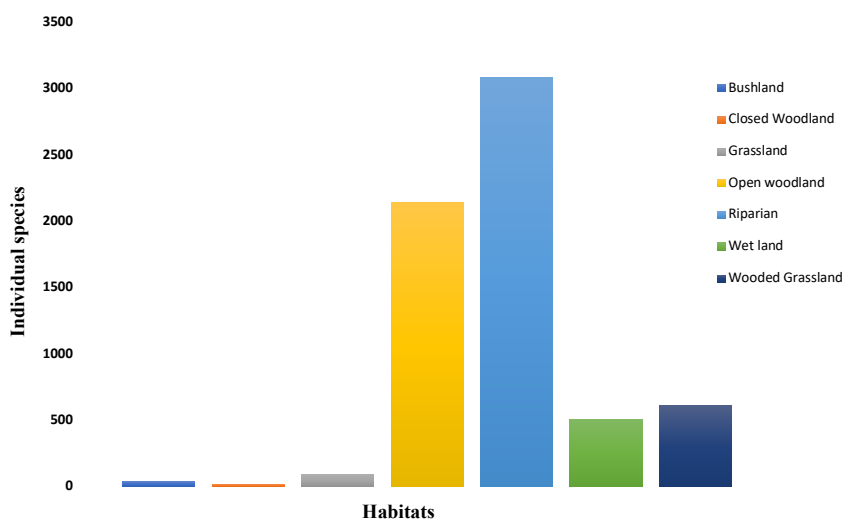


Figure 11. Individual species recorded across the surveyed habitats

DISCUSSION

The conducted camera trap survey yielded a substantial dataset of wildlife observation, capturing 56 species (43 mammals and 12 avian) across a diverse range of habitats. The high operational success rate of the camera traps (89%) provided a robust foundation for assessing species composition, relative abundance, richness, diversity and occupancy within the study area similarly to the results from Tanwar *et al.*, (2021) study in Ranthambhore National Park.

The dominance of the Order Cetartiodactyla in the photographic records, with Hippopotamus having the highest frequency, highlights the significant presence of large herbivores, availability of preferred habitats and forage resources in the surveyed landscape (Tefera *et al.*, 2024). Conversely, the single photographic capture of several species, including the Small-eared greater galago, Angola Black and white colobus monkey, may suggests potentially low densities, restricted distributions or unsuitable technique used for such species within the study area, drawing the need for further surveys (Mitterwallner *et al.*, 2024). The detection of the endangered African Wild dog, though with 16 exposures, is a significant finding, highlighting the importance these areas to this

species and emphasizing the need for continued monitoring and conservation effort (O'Neill *et al.*, 2020).

Habitat analysis revealed that the bushland supported the highest abundance of animals. This is likely attributable to its heterogeneous structure, offering a wider array of food resources and cover for a few species (Mango *et al.*, 2022). In contrast, the lower abundance and species richness observed in closed woodland might be linked to habitat specialization or potentially lower resources availability compared to the more open woodland. Even the distribution of animals was recorded in closed woodland, despite lower richness, could indicate a uniform habitat use by the species present (Kavana *et al.*, 2024).

The species' diversity and richness indices further support the importance of open woodland, which exhibited the highest diversity (H') and richness values. This suggests that open woodland provides a critical habitat for a wide range of species in Msolwa as the result from Kavana *et al.*, 2024 study. The lower diversity in grassland could be due to its less complex structure and potentially limited resources availability compared to the woodland habitats (Kavana *et al.*, 2024).

Naïve occupancy estimates provide valuable insights into the spatial distribution of key species (Morgan *et al.*, 2024). The high occupancy of African Savanna elephant indicates its widespread presence throughout the sample area, which has implications for habitat management and potential human-wildlife conflict mitigation. The relatively high occupancy of common duiker and warthogs also suggests their successful adaptation to the prevailing environmental conditions across the Msolwa area.

Finally, the documented anthropogenic activities, including poaching of game meat and illegal fishing, are concerning findings. These activities pose direct threats to wildlife populations and the integrity of the riparian and woodland habitats. The presence of such activities highlights the need for more frequent and effective patrols, which will increase the chance of detecting illegal activities and correcting them before the damage.

CONCLUSION AND RECOMMENDATION

The study revealed the presence of a wide range of habitats in the Msolwa area, which supports a diverse range of wild species ranging from mammals to avian group. The area requires long-term wildlife monitoring with increased effort using camera traps. This will ultimately provide comprehensive information on diversity, wealth and distribution of wild species. Additionally, it is recommended that immediate steps towards anti-poaching should be taken, and in this regard, we suggest the following measures.

- i. Increase ranger patrols – Deploy more teams in identified poaching hotspots areas to protect the camera traps.
- ii. Engage communities – Train and hire local scouts to report poaching activities.
- iii. Use of technology – Utilize GPS tracking collars on key species and

drones for real-time monitoring.

- iv. Strengthen law enforcement – Work with authorities for quicker prosecution of poachers.
- v. Offer alternative livelihoods – Support eco-tourism and sustainable farming to reduce poaching dependence.

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Appendix 1. Pictures of animal and bird species captured during 2024 survey at Msolwa area

 Hartebeest (<i>Alcelaphus lichtensteinii</i>)	 Hippopotamus(<i>Hippopotamus amphibious</i>)
 African civet (<i>Civettictis civetta</i>)	 Lion (<i>Panthera leo</i>)
 Honey badger (<i>Mellivora capensis</i>)	 Bushbuck (<i>Tragelaphus scriptus</i>)



Red-natal duiker (*Cephalophus natalensis*)



Yellow baboon (*Papio cynocephalus*)



Impala (*Aepyceros melampus*)



Large spotted genet (*Genetta maculata*)



African buffalo (*Syncerus caffer*)



Savanna elephant (*Loxodonta africana*)



Temminck's Ground Pangolin (*Smutsia temminckii*)



Crested Porcupine (*Hystrix cristata*)



Aardvark (*Oryctiropus afer*)



Waterbuck (*Kobus ellipsiprymnus*)



Bushpig (*Potamochoerus larvatus*)



Olive baboon (*Papio anubis*)



Leopard (*Panthera pardus*)



Cape hare (*Lepus capensis*)



African Wild Dog (*Lycaon pictus*)



Zebra (*Equus quagga*)

	
Marsh mongoose (<i>Atilax paludinosus</i>)	Serval cat (<i>Leptailurus serval</i>)
	
Wildbeest (<i>Connochaetes taurinus</i>)	Warthog (<i>phacochoerus africanus</i>)
	
Miombo genet (<i>Genette angolensis</i>)	Spotted hyena (<i>Crocuta crocuta</i>)
	
Vervet monkey (<i>Chlorocebus pygerythrus</i>)	Eland (<i>Tragelaphus oryx</i>)



Sable antelope (*Hippotragus niger*)



African savanna hare (*Lepus victoriae*)



Lesser Kudu (*Tragelaphus imberbis*)



Bushy-tailed mongoose (*Bdeogale crassicauda*)



Black-and-white colobus monkey (*Colobus guereza*)



Banded mongoose (*Mungos mungo*)



Kirk's dik-dik (*Madoqua kirkii*)



African gray hornbill (*Tockus nasutus*)

 African giant pouched rat (<i>Cricetomys ansorgei</i>)	 Greater kudu (<i>Tragelaphus strepsiceros</i>)
 Small-eared greater galago (<i>Otolemur garnettii</i>)	 Dwarf mongoose (<i>Helogale parvula</i>)
 Crested guineafowl (<i>Guttera pucherani</i>)	 Common duiker (<i>Sylvicapra grimmia</i>)
 Stripped kingfisher (<i>Halcyon chelicuti</i>)	 Superb starling (<i>Lamprotornis superbus</i>)



Southern ground hornbill (*Bucorvus leadbeateri*)



Crowned hornbill (*Lophoceros alboterminatus*)



White-browed Scrub-robin (*Cercotrichas leucophrys*)



Little Bee-eater (*Merops pusillus*)



Egyptian goose (*Alopochpa aegyptiaca*)



Red billed oxpecker (*Buphagus erythrorhynchus*)



White bellied bustard (*Eupodotis senegalensis*)



Helmeted guineafowl (*Numida meleagris*)

Appendix 2: Species check list in Msolwa - Nyerere National Park

S/N	CLASS	ORDER	FAMILY	COMMON NAME	SPECIES NAME	PHOTO RECORDS
1	AVES	Coraciiformes	Alcedinidae	Striped kingfisher	<i>Halcyon chelicuti</i>	1
2			Meropidae	Superb starling	<i>Lamprotornis superbus</i>	1
3			Bucerotidae	African gray hornbill	<i>Tockus nasutus</i>	2
4		Galliformes	Numididae	Crested guineafowl	<i>Guttera pucherani</i>	77
5				Helmeted guineafowl	<i>Numida meleagris</i>	55
6		Bucerotiformes	Bucerotidae	Crowned hornbill	<i>Lophoceros alboterminatus</i>	1
7		Bucerotiformes	Bucorvidae	Southern ground hornbill	<i>Bucorvus leadbeateri</i>	28
8		Anseriformes	Anatidae	Egyptian goose	<i>Alopochpa aegyptiaca</i>	4
9		Passeriformes	Muscicapidae	White-browed Scrub-robin	<i>Cercotrichas leucophrys</i>	1
10			Sturnidae	Little bee-eater	<i>Merops pusilus</i>	1
11			Buphagidae	Red billed oxpecker	<i>Buphagus erythrorhynchus</i>	2
12		Otidiformes	Otididae	White bellied bustard	<i>Eupodotis senegalensis</i>	1
13	MAMMALIA	Carnivora	Herpestidae	Bushy tailed mongoose	<i>Bdeogale crassicauda</i>	47
14			Viverridae	African Civet	<i>Civettictis civetta</i>	100
15			Hyaenidae	Spotted Hyena	<i>Crocuta crocuta</i>	206
16			Vivveridae	Large spotted Genet	<i>Genetta maculata</i>	31
17				Miombo Genet	<i>Genetta angolensis</i>	13
18			Herpestidae	Dwarf Mongoose	<i>Helogale parvula</i>	5
19			Felidae	Serval Cat	<i>Leptailurus serval</i>	5
20			Canidae	African Wild Dog	<i>Lycaon pictus</i>	16
21			Mustelidae	Honey badger	<i>Mellivora capensis</i>	16
22			Herpestidae	Marsh mongoose	<i>Atilax paludinosus</i>	21
23				Banded Mongoose	<i>Mungos mungo</i>	13
24			Felidae	Lion	<i>Panthera leo</i>	15
25				Leopard	<i>Panthera pardus</i>	66
26		Cetartiodactyla	Bovidae	Impala	<i>Aepyceros melampus</i>	956
27				Hartebeest	<i>Alcelaphus lichtensteinii</i>	289
28				Sable antelope	<i>Hippotragus niger</i>	10
29				Waterbuck	<i>Kobus ellipsiprymnus</i>	158
30				Lesser kudu	<i>Tragelaphus imberbis</i>	4
31				Greater Kudu	<i>Tragelaphus strepsiceros</i>	2
32				Common duiker	<i>Sylvicapra grimmia</i>	31
33				Kirk's dik-dik	<i>Madoqua kirkii</i>	2
34				Wildebeest	<i>Connochaetes taurinus</i>	426

S/N	CLASS	ORDER	FAMILY	COMMON NAME	SPECIES NAME	PHOTO RECORDS
35				Red-natal duiker	<i>Cephalophus natalensis</i>	6
36				African buffalo	<i>Syncerus caffer</i>	589
37			Suidae	Eland	<i>Tragelaphus oryx</i>	76
38				Bushbuck	<i>Tragelaphus scriptus</i>	120
39				Warthog	<i>Phacochoerus africanus</i>	339
40				Bushpig	<i>Potamochoerus larvatus</i>	64
41			Hippopotamidae	Hippopotamus	<i>Hippopotamus amphibius</i>	1460
42		Lagomorpha	Leporidae	Cape hare	<i>Lepus capensis</i>	133
43				African savannah hare	<i>Lepus victoriae</i>	11
44		Perissodactyla	Equidae	Zebra	<i>Equus quagga</i>	182
45		Pholidota	Manidae	Temminck's Ground Pangolin	<i>Smutsia temminckii</i>	2
47		Primates	Cercopithecidae	Vervet Monkey	<i>Chlorocebus pygerythrus</i>	35
48			Galagidae	Small eared greater galago	<i>Otolemur garnettii</i>	1
49			Homonidae	Human being	<i>Homo sapiens</i>	6
50			Cercopithecidae	Olive baboon	<i>Papio anubis</i>	8
51				Angola Black and White colobus monkey	<i>Colobus angolensis</i>	1
52				Yellow baboon	<i>Papio cynocephalus</i>	391
53		Proboscidea	Elephantidae	African elephant	<i>Loxodonta africana</i>	449
54		Rodentia	Muridae	African giant pouched rat	<i>Cricetomys ansorgei</i>	19
55			Hystriidae	Crested Porcupine	<i>Hystrix cristata</i>	23
56		Tubulidentata	Orycteropodidae	Aardvark	<i>Orycteropus afer</i>	30

PRELIMINARY RAPID ASSESSMENT ON POPULATION AND SPATIAL DISTRIBUTION OF ASHY RED COLOBUS MONKEY (*PILIOCOLOBUS TEPHROSCELES*) IN COMMUNITY FORESTS IN BIHARAMULO DISTRICT, NORTHWESTERN TANZANIA

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ABSTRACT

The ashy red colobus monkey (*Piliocolobus tephrosceles*) is classified as endangered by the IUCN and faces threats from ongoing habitat destruction due to agriculture and charcoal making in unprotected community forests. This study aimed to estimate the population and spatial distribution of the ashy red colobus monkey within three community owned forests in the Biharamulo District of North-western Tanzania. Conducted in 2024, the study utilized active observation searches and geospatial surveys to gather data on abundance of the individual monkeys and their spatial distributions. A total area of 404,357 square meters across the three forests was surveyed, resulting in the observation of 108 individuals. Employing negative binomial regression results to inform Bayesian negative binomial model prior, we estimated a total of 180 individuals, distributed across Nyamahanga (40), Karundi (47), and Mabila (93) forests. For spatial distribution analysis, we utilized ArcGIS Pro version 3.3.0. We began by delineating polygon boundaries for each sampled forest and clipping the 2020 land cover and land use raster data to match these extents. Subsequently, we overlaid both the probable observation points (POPs) and actual observation points (OPs) onto the maps. This spatial analysis revealed that the monkeys' distribution was largely confined to gullies and dense canopy patches, indicating significant habitat fragmentation and degradation. These findings underscore the urgent need for targeted conservation strategies to prevent further population decline, particularly in under-protected community-owned forests. Implementing habitat restoration and protection measures is essential to ensure the survival of this endangered primate species.

Keywords: *Piliocolobus tephrosceles*, Endangered, Red colobus monkey, Population, Habitat degradation, Conservation

INTRODUCTION

Researchers estimate that the current rate of species loss is around 1,000 times the background extinction rate, the rate of species extinction over a long geological timescale. The significant increase in extinction rate

marks an unprecedented environmental crisis driven by unsustainable human activity such as deforestation and fossil fuel consumption (De Vos *et al.*, 2015). Such a loss of biodiversity impacts the overall health of ecosystems, decreasing the stability of ecosystem services, which are benefits humans derive from

ecosystems functions. These services are vital to our planet and society, as they include the production of renewable resources, such as food and fresh water, climate regulation, and disease control (Cardinale *et al.*, 2012). In line with global trends of species loss, over 60% of primate species are now threatened with extinction (Estrada *et al.*, 2017). The destruction and fragmentation of forest habitats, largely due to logging, mining, infrastructure development, and the conversion of land for agriculture, are critical factors contributing to the decline of primate populations (Struhsaker, 1997; Estrada *et al.*, 2017; Kitegile *et al.*, 2021; Linder *et al.*, 2021). The red colobus monkey plays a significant role in its ecosystem. Primates like the ashy red colobus monkey occupy a unique niche in their ecosystem as both predator, prey, and a mutualistic species within food webs; in turn, influencing the structure and resilience of the ecosystems they inhabit (Estrada *et al.*, 2017). Many primates, including this species, are pollinators (Heymann, 2011) and seed dispersers through their interactions with flowering flora and their fruit foraging behaviour, thereby supporting forest regeneration and ecosystem functioning (Chapman *et al.*, 2013). These interactions underscore the essential ecological contributions of the red colobus monkey.

Despite their significant role within their ecosystems, red colobus monkeys are a highly threatened group, with more than three quarters of its taxa listed as either endangered or critically endangered according to the IUCN (International Union of Conservation of Nature) Red List (Linder *et al.*, 2021). The red colobus feeds primarily on young leaves but is known to consume fruits seasonally (Chapman & Chapman, 1999). Most taxa live in large groups comprised of both male and female individuals, with females frequently outnumbering males (Struhsaker, 2010). Group members spend most of their day feeding or resting (Struhsaker, 2010). Their primary threats include habitat loss, degradation and fragmentation driven by demand for natural resources, as well as illegal hunting for both

subsistence and commercial purposes and retaliatory killings linked to suspected crop raiding (McGraw *et al.*, 2020). Indeed, given their arboreal nature, canopy alteration affects the red colobus monkey's local abundance (Rovero & Struhsaker, 2007). For these reasons, they are likely the first species to die out in a highly disturbed habitat (Linder *et al.*, 2024).

The Ashy red colobus monkey (*Piliocolobus tephrosceles*) is listed as endangered according to the IUCN Red List with a total population of over 25,000 individuals (Linder *et al.*, 2021). The ashy red colobus has a highly fragmented distribution, found in five subpopulations across Tanzania and Uganda, with the vast majority of its population found in Uganda's Kibale National Park ($\geq 17,000$) (Struhsaker *et al.*, 2005). They occupy a wide variety of forest types including tropical forests, savanna woodlands, and degraded and regenerating forests (Struhsaker, 2010; Chapman *et al.*, 2013; Kibaja *et al.*, 2023). However, the ashy red colobus has been eliminated in forests surrounding Kibale National Park and in some forests on the Ufipa Plateau in southwestern Tanzania (Davenport *et al.*, 2007; Chapman *et al.*, 2013) and is designated as endangered owing to the real possibility of decline in populations located outside of well-protected national parks (Linder *et al.*, 2021). Little is known about the population in northwestern Tanzania, which is thought to occupy the Biharamulo Game Reserve and surrounding forest reserves. In the latter, consumptive human activities such as logging are permitted, threatening the existing population. This study aims to estimate the current population and distribution of the ashy red colobus monkey in three community owned forest reserves in Biharamulo district, Northwestern Tanzania, where no prior research on the species has been conducted. The research first involved delineating the boundaries of the community owned forest, which had not been previously documented. A Negative Binomial regression model with a log link function was then applied to quantify the relationship between

the monkey count and the predictor variables. The population of the Ashy red colobus monkey was subsequently estimated using Bayesian Negative Binomial model. Finally, species distribution was mapped using two types of spatial data, Probable Observation Points (POPs), sites identified by the local Villages Game Scouts (VGS) as common areas for the target species and direct Observation Points (OPs), representing locations where the monkeys were directly sighted and counted

MATERIALS AND METHODS

Study area

This study took place in three community owned forest reserves, known as Nyamahanga, Mabila, and Karundi, in Northwestern Tanzania’s Biharamulo District (Figure 1).

These forests encompass the villages of Nyamahanga, Ntungamo, and Mabila and lie south of the newly gazetted Burugi-Chato National Park and Lake Victoria (Makunga & Gobolo, 2020). The area is situated at an elevation of 1200-1500 m above sea level with coordinates at latitude 2° 37’ 55’’ S and longitude 31° 18’ 32’’ E. Temperatures vary only slightly due to proximity to the equator, with highs around 30°C and lows around 20°C. Rainfall ranges from 600 to 1200 mm. The primary agricultural crops cultivated in the district include bananas, coffee, maize, beans, yams, and cassava. Practices close to the forests include excavation for brick making and pastoralism in addition to the agricultural activity listed above. Areas cleared for agriculture are also planted with pine trees for commercial charcoal burning or fuel wood.

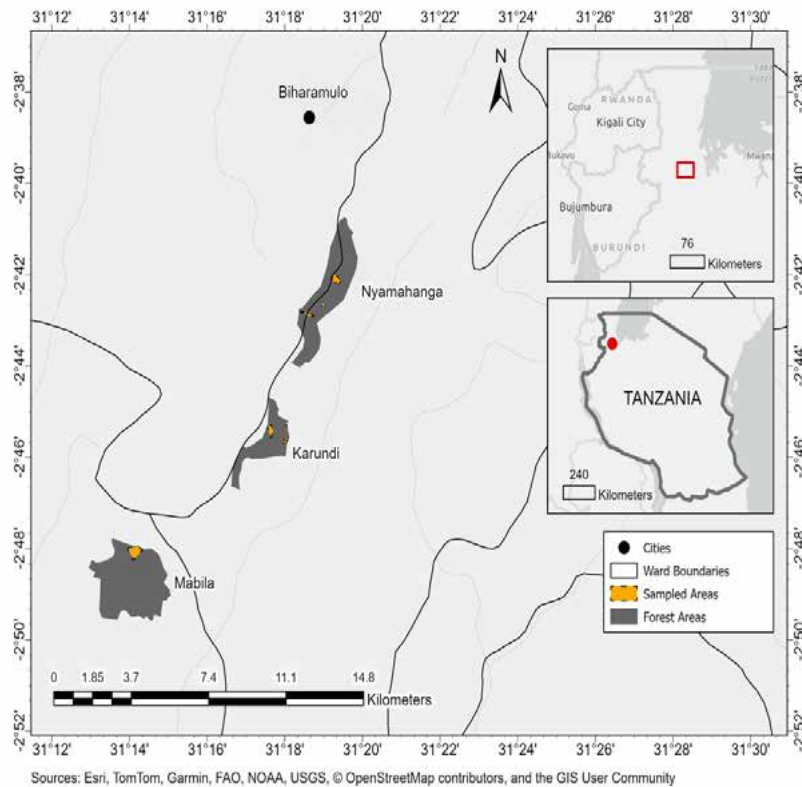


Figure 1: Map of study areas in Northwestern Tanzania.

Ecologically, the area is characterized by cliffs and hills covered in dense tropical forest with dominant tree species *Combretum molle*, *Pericopsis angolensis*, and *Markhamia obtusifoli*. Biharamulo District is home to multiple notable biodiversity hotspots, such as Ruiga Forest Reserve in the east and Biharamulo and Nyantakara Forest Reserves to the south. These areas are facing increasing pressure from anthropogenic stressors such as agriculture, livestock keeping, and unsustainable land use and management practices, driven, in part, by an increasing human population and climate change.

Methods

Two complementary methods were used to estimate the population and spatial distribution of the ashy red colobus monkey (*Piliocolobus tephrosceles*) across three community forests in Biharamulo District. The first method was active observation search, to estimate the species' abundance in each forest. Active observation search is a wildlife monitoring or ecological survey method where researchers deliberately and systematically search for the target species by moving through their habitat while actively observing signs of presence. The second method employed was geospatial survey, which involves collecting data on location, shape and spatial relationships of natural or man-made features on the Earth's surface. This approach was used to map the total forest cover, delineate the portion of the forest surveyed, and to record point distribution data for the Ashy red colobus monkey. Mapping was necessary since no prior spatial information existed for these forests, making it necessary to determine both the overall extent of the forest and the areas actually covered during data collection. The resulting spatial information were then used to estimate the species' population within each surveyed forest. These surveys were conducted with the support and guidance of local knowledge. Both methods were carried out for two days in each forest in November 2024.

Active Observation Search

Active observation search is widely regarded as one of the most reliable methods for conducting primate census, especially in fragmented habitats (Davenport *et al.*, 2007). In this study, the active observation searches were conducted as recces surveys, we explored the forests to locate the Ashy red colobus groups, record their abundance, and collect their georeferenced data. Routes were flexible and adapted based on the forest structure and terrain and sightings, enabling the identification of new areas used by the monkeys. This technique was similarly applied by Kitegile *et al.* (2021) to estimate the population of the ashy red colobus monkey (*Piliocolobus tephrosceles*) in the Ufipa Plateau, Southwestern Tanzania. Observations were conducted during the species' peak activity periods, including early morning (from ~7:30 am to ~11:30 am), and late evening (from ~4:30 pm to ~6:00 pm) to record the number of observed red colobus monkeys and other metadata. Given that the ashy red colobus is known for its shy behaviour, which can introduce detection bias in ground-based count, direct observations were supplemented with unmanned aerial vehicle (UAV; drone) surveys to enhance the accuracy and minimal disturbances (Semel *et al.*, 2019). The drone was deployed each time the targeted species was detected to confirm group count and capture distinguishing group's visual features. Only one licensed team member operated the drone, with each flight lasting 10-15 minutes on average.

Group sizes were determined by counting all identified and encountered groups within the forests. To avert double counting, these observations were cross-validated using previous captured drone imagery whenever a new group was detected. Moreover, the field team used distinctive physical characteristics (e.g coloring, deformities, leaping and missing body parts), group location and composition throughout the survey period to ensure that each group was recorded only once.

Geospatial surveys

The compiled GPS data were used to delineate the overall forest extent, the actual surveyed forest area, and the specie's distribution location. Using a geospatial survey device (Global Positioning System; GPS), we recorded multiple georeferenced points along the boundaries of the forest. These points were later joined to form a polygon mapping forest's boundary. The same approach was applied to delineate the sampled portions of the forest area. In addition GPS coordinates were collected to record the spatial distribution of the ashy red colobus monkey (*Ptilocolobus tephrosceles*). Two types of species' distribution points were recorded in the field including the Probable Observation Points (POPs) and the Observation Points (OPs). POPs were sites identified by the local Villages Game Scouts (VGS) as areas frequently used by the targeted species based on their local ecological knowledge, whereas OPs represented location where the monkeys were directly observed during field surveys. All spatial data were georeferenced and integrated into the subsequent spatial analyses.

Data analysis

The population estimate for the ashy red colobus monkey were based on the count data representing the number of individuals observed per forest. Two predictor variables the Area_scaled (scaled total forest area) and the Sampled_area_scaled (scaled area actually surveyed) both derived from the geospatial mapping data, were used for the estimate. A negative Binomial regression model with a log link function was first applied to quantify the relationship between the monkey count and the predictor variables. The model was defined as:

$$(\log(\mu_i) = \beta_0 + \beta_1(\text{Area\_scaled}_i) + \beta_2(\text{Sampled\_area\_scaled}_i))$$

Where Y_i is the observed count of the ashy red colobus monkey in the forest, μ_i is the expected mean count, θ is the dispersion parameter, and β_0 , β_1 , β_2 are model coefficients. This model was used to estimate how the forest area and sampled area influence monkey abundance

while addressing overdispersion in the count data. The resulting coefficients were then used as priors in a Bayesian Negative Binomial model, expressed as:

$$(\log(\mu_i) = \alpha + \beta_1(\text{Area\_scaled}_i) + \beta_2(\text{Sampled\_area\_scaled}_i) + \delta^2)$$

Where α are the regression-derived priors from the frequentist model, and δ^2 represents prior uncertainty. The Bayesian model refined population estimates by integrating parameter uncertainty and data variability.

Traditional distributional species mapping formed the basis for our spatial analysis. We used ArcGIS Pro (version 3.3.0) to delineate forest boundaries, clipped the 202 Land use/Land Cover (LULC) raster to the forest extents, and overlaid the Probable Observation Points (POPs) and the Observation Points (OPs) from the geospatial survey data to visualize the species' spatial distribution across the study site and LULC.

RESULTS

Ashy red colobus monkey population

Of the total area of 6.57 km², approximately 0.15 km² of Nyamahanga Forest Reserve was canvassed during animal counts. Two groups of ashy red colobus monkeys were encountered in the sampled area, totalling 29 individuals. In Karundi Forest Reserve, approximately 0.04 km² out of total area 3.14 km² was canvassed, and a total of two groups of ashy red colobus monkeys were encountered in this area, containing 21 individuals. Of the total area of 8.58 km², approximately 0.21 km² of Mabila Forest Reserve was canvassed during animal counts. Three groups of ash red colobus monkeys were encountered in this area, totalling 58 individuals. Across all three forests, a total of 108 individuals were observed in 7 groups. The average group size of the three forests was 15.43 individuals. Table 1 provides information on group size and geographical location (GPS coordinates) where the monkeys were encountered in each forest.

Table 1: Group ID, location of observation, group size, and total individuals observed in each forest canvassed in Biharamulo District.

Forest		Group ID	Location (GPS coordinates)		Group size
Nyamahanga		1	-2.714259	31.308777	23
		2	-2.711064	31.316115	6
<i>Total individuals</i>					29
Karundi		3	-2.75376	31.300941	6
		4	-2.760261	31.300151	15
<i>Total individuals</i>					21
Mabila		5	-2.803443	31.235389	13
		6	-2.80067	31.23288	35
		7	-2.800245	31.233668	10
<i>Total individuals</i>					58

The negative binomial regression model produced coefficients of 3.4908 for the intercept, 5.9553 for the area scaled, and -5.4719 for the sampled area scaled. The intercept coefficient suggests that when both predictor variables equal zero, the baseline population of ashy red colobus monkeys equals ~32.9 individuals ($\exp(3.4908)$). The area-scaled coefficient suggests that for every additional 1,000,000 m² in the scaled forest area, the log population size increases by 5.96, or the population increases by a factor of 390.5 once exponentiated. However, this effect was not statistically significant ($p = 0.111$). On the other hand, the sampled area-scaled coefficient suggests that for every additional 1,000,000 m² in the scaled sampled area, the log population size decreases by 5.47, or the population decreases by a

factor of 0.0042 when exponentiated. Again, this effect was not statistically significant ($p = 0.146$). Therefore, the negative binomial regression model estimates the abundance of ashy red colobus monkeys in each of the three forests to be the same as observed in the field. These results were then used to inform the priors of the Bayesian negative binomial model. The Bayesian model estimated the ashy colobus monkey population to be 40, 47, and 93 individuals in Nyamahanga, Karundi, and Mabila forests, respectively (Figure 2). Therefore, we predict a total of 180 individuals residing in all three forests, which cover a total of 18.28 km² together. We therefore, estimate a population density across all three forests (number of individuals divided by total area) of 9.84433266 individuals/km².

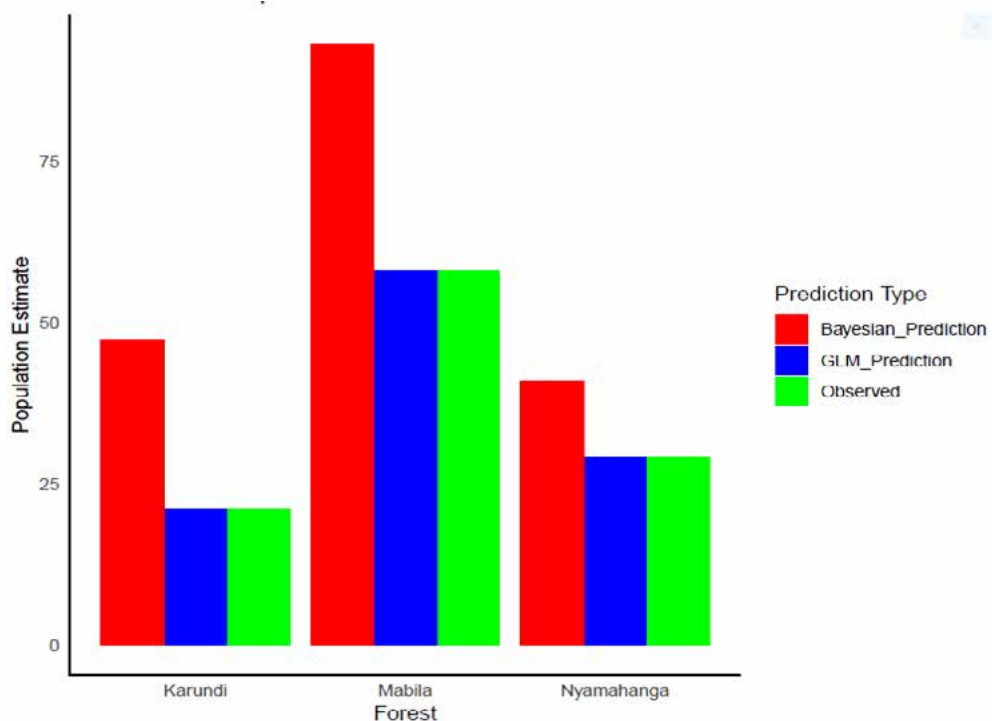


Figure 2: Population estimate of each forest canvassed in Biharamulo District. Red indicates results of the Bayesian negative binomial model (Bayesian_Prediction); blue indicates the results of the negative binomial regression model (GLM_Prediction); and green indicates the number of individuals observed in each forest (Observed).

Ashy red colobus monkey distribution

Generally, groups of monkeys were found in tall, dense stands of forest, typically along riverbeds flowing down the hills. They were found to roost in cliff caves and were never seen in disturbed areas where agroforestry or human development took place. A total of 9 observations of ashy red colobus monkeys were recorded in Nyamahanga Community

Forest, and 15 probable points of observation were recorded. A total of 9 observations of ashy red colobus monkeys were recorded in Karundi Community Forest, and 35 probable points of observation were recorded. A total of 13 observations of red colobus monkeys were recorded in Mabila Forest Reserve, and 13 probable points of observation were recorded. Figure 3 shows forest maps with these features plotted.

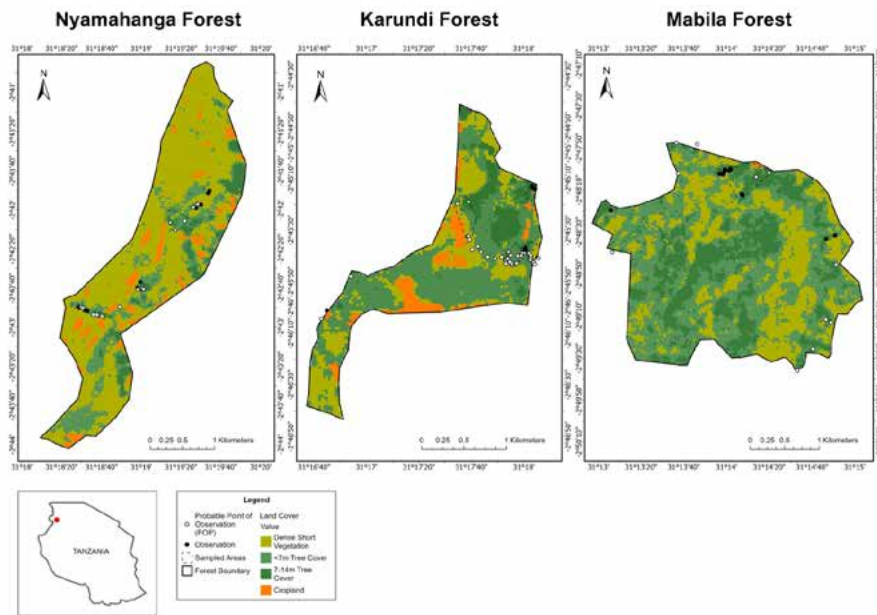


Figure 3: Maps of Nyamahanga, Karundi, and Mabila forests showing land cover data and collected observation points and probable points of observation. The white circles indicate a probable point of observation. The black circles indicate a point of observation. For land cover, orange indicates cropland, light green indicates dense short vegetation, middle green indicates tree cover less than 7m in height, and dark green indicates tree cover 7-14m in height. Esri, ArcGIS Pro 3.3.

DISCUSSION

Population

The negative binomial regression model produced coefficients of 5.9553 for the area scaled and -5.4719 for the sampled area scaled, suggesting that changes in area scaled and sample area scaled may influence the population. However, neither of these effects was statistically significant. Further investigation with additional variables or data might be needed to refine these estimates. The Bayesian negative binomial model estimated the ashy colobus monkey population to be 40, 47, and 93 individuals in Nyamahanga, Karundi, and Mabila forests, respectively. These numbers are low compared to similar studies. Kitegile *et al.* (2021) found a population of 528 ashy red colobus monkeys in Mbizi forest, which has an area of 29.3 km². Comparing basic population density (number of individuals divided by total forest area), we report a density of 9.84433266 individuals per square kilometre compared to

Kitegile's 18.0204778 individuals per square kilometre, which is nearly half of the density reported in the threatened Mbuzi forest in the Ufipa Plateau. In Kibale National Park, where a healthy group of ashy red colobus monkey is reported, there is a conservative estimate of 25–50 red colobus per km², which more than triples the density found in this study. While this could indicate an underestimate, it's also possible that the population of ashy red colobus monkeys in the studied forests is that much smaller and more vulnerable. The smaller population size may also be interpreted through the lens of antagonistic pleiotropy hypothesis, a concept in evolutionary biology which proposes that some genes can have opposing effects on an organism's fitness by offering benefits early in life but incurring costs later. In this context, genes that enhance the survival of Ashy red colobus monkey under harsh or resource limited conditions such as promoting delayed reproduction or increased longevity might be favoured by natural selection. Although these traits reduce the immediate reproductive

output, they can increase the likelihood of survival until more favourable conditions arise, thereby influencing the overall population size observed in less optimal habitats.

The average group size for all three forests was 15.43 individuals, ranging from 6-35 individuals per group. Red colobus monkey group size typically ranges from 25-40 individuals, but group sizes are known to vary from 3 to more than 80 (Struhsaker, 1975; Oates, 1994). Our observations are not outside of this range, but they do appear on the lower side. Indeed, in Kitegile *et al.* (2021), group size of ashy red colobus monkeys averaged a higher 26.37 individuals in areas of the Ufipa Plateau. One explanation for this lower trend is that we could have identified subgroups and groups indiscriminately. This challenge was documented in Chapman & Chapman (1999), where on observing groups of red colobuses, researchers were able to identify subgroups within them, leading to concerns of counting subgroups as groups and introducing bias to population counts. This seems less likely to apply to our observations, as each group was observed at considerable distance from each other. However, it's possible that groups split off during foraging and reunited to roost together; intragroup competition reduces foraging efficiency, favoring smaller groups when food is scarce (Chapman & Chapman, 1999; Gillespie & Chapman, 2001). As such, smaller groups may be indicative of a lower density of potential food resources (Chapman & Chapman, 1999; Mbora & Meikle, 2004). In addition, in declining populations of red colobus monkeys, group size has been observed to decrease. For example, the critically endangered Tana River red colobus monkey experienced more than a 50% decrease in group size from 1970 to 2004, largely due to habitat fragmentation (Mbora & Meikle, 2004). When a habitat is patchy, smaller forests may contain smaller groups (Struhsaker, 1975; Gatinot, 1978). The relatively small group sizes we observed could be explained by increasing habitat fragmentation in these unprotected community forest reserves.

Distribution

Results from this study indicate that the ashy red colobus monkey is primarily distributed in riverine habitats, along rivers and streams, or within dense forest stands with tree cover reaching up to 14 meters in height (Fig. 3). Notably, their presence was restricted to these areas; although some individuals were observed near cropland, they were never seen within cleared or cultivated areas. This pattern may be explained by the species' canopy-dependent ecological niche and its arboreal locomotion. Moist riverine areas provide optimal conditions for the growth of dense forests, which the ashy red colobus relies on for both movement and foraging (Kitegile *et al.*, 2021). These findings align with existing knowledge of the species' habitat requirements. Additionally, the monkeys' localized restriction to gullies has been similarly observed in previous studies (Kitegile *et al.*, 2021). The restriction of these monkeys to specific habitats, coupled with observed signs of habitat degradation, suggests increasing fragmentation. This fragmentation may act as a barrier to movement, leading to the isolation of groups (Fahrig, 2003).

CONCLUSION

The populations of ashy red colobus monkeys in the community forests of Nyamahanga, Karundi, and Mabila in Biharamulo District are highly limited, with density nearly half of similarly threatened populations in Southwestern Tanzania and more than one third the density of healthy populations in Kibale National Park (Struhsaker *et al.*, 2005; Kitegile *et al.*, 2021). Additionally, observed group size was considerably smaller than recorded in the literature (Struhsaker, 1975; Oates, 1994; Kitegile *et al.*, 2021). Groups were restricted to relatively undisturbed patches of forest, primarily in gullies in riverine habitat. Such dire findings are reproduced across the literature as results of habitat degradation are felt and recorded in declining primate populations around the world (Srivastava *et al.*, 2001; Arroyo-Rodríguez & Dias, 2010;

Estrada *et al.*, 2017; Kitegile *et al.*, 2021). Indeed, these situations are undoubtedly connected to continuing habitat destruction and fragmentation by humans in the unprotected community forests of Nyamahanga, Karundi, and Mabila. If destructive activities such as livestock overgrazing in community forests, agriculture expansion, expansion of commercial tree plantations, illegal hunting by livestock herders, school children and farmers, as well as the continued spread of human settlements, aren't effectively controlled, the Ashy red colobus monkey is likely to face local extinction in these forests.

RECOMMENDATIONS

Preliminary results from this study indicate that populations of ashy red colobus monkeys persist in Nyamahanga, Karundi, and Mabila forests. However, it is necessary to scale up the survey to cover the remaining area of the forest reserves to determine the true status of the ashy red colobus monkey in Northwestern Tanzania. From these preliminary results, immediate management interventions are recommended to preserve the remaining habitat patches that exist there to slow the assumed decline of the ashy red colobus population. This management should be twofold: on one hand, legal protection, but on the other, community involvement and engagement, which is crucial to promote adherence to law and observance of recommendations. Given the species' restriction to undisturbed riverine habitats and the ongoing habitat degradation in community forests, a comprehensive study should be conducted to i) assess habitat suitability of the ashy red colobus monkey across its range and to ii) understand the long-term viability of isolated populations with a genetic and isotopic study looking at population structure, gene flow, and dietary shifts in response to habitat degradation. The latter should take into account that it's likely the nearby community forests and reserved areas are home to the ashy red colobus monkey, and their study would benefit the bank of knowledge on this species, which informs its protection.

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ETHICAL STATEMENT

This research abided by the Oryx guidelines on ethical standards, was approved country SIT review board and adhered to the all other in country research protocols.

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GENETIC BARCODING UNCOVERS HIDDEN DIVERSITY OF HAPLOCHROMINE CICHLIDS IN SHIRATI BAY, LAKE VICTORIA

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ABSTRACT

Haplochromine cichlids in Lake Victoria have undergone significant population declines following the introduction of the predatory Nile perch (*Lates niloticus*). These declines have been further exacerbated by eutrophication, overfishing, and habitat degradation, which have collectively contributed to the disappearance of many *Haplochromis* species from the catch. However, recent studies show that some species previously thought to have disappeared from catches may be recovering, possibly due to the recent decline in Nile perch populations. Despite these indications, the actual composition of *Haplochromis* species remains uncertain due to a lack of recent inventory data. Additionally, morphological similarities and intraspecific variations complicate identification, leaving populations vulnerable to overexploitation. To address this gap, the present study amplified a 620-base pair fragment of the cytochrome oxidase subunit I (COI) gene from 94 haplochromine individuals collected from Shirati Bay, Lake Victoria to assess species composition and distribution. Seven species were identified: *Haplochromis nyererei*, *H. alluaudi*, *H. erutus*, *H. simpsoni*, *H. sauvagei*, *H. nubilus*, and *H. pyrrhocephalus*. However, *H. nubilus* could not be matched in the NCBI database, highlighting the need to update existing genetic reference libraries. Phylogenetic analysis confirmed the accuracy of morphological identifications, emphasizing the value of genetic barcoding in clarifying taxonomy and supporting fisheries monitoring. Among the identified species, *H. erutus* and *H. simpsoni* were recorded in Tanzania for the first time. Similarly, *H. nyererei* was the most widespread, occurring at all sampling sites. In contrast, *H. erutus* and *H. alluaudi* were rare, despite being classified as Not Evaluated and Least Concern, respectively. These findings highlight a mismatch between conservation status and local abundances, emphasizing the need for periodic, site-specific reassessments of conservation status, particularly for rare and newly recorded species to ensure that management and conservation strategies are based on up-to-date local population data.

Keywords: DNA barcoding, Haplochromine cichlids, Lake Victoria, Mitochondrial cytochrome oxidase subunit I gene (COX1), Species diversity

INTRODUCTION

Lake Victoria, the world's largest tropical freshwater lake is globally recognized for its exceptional biodiversity, particularly its remarkable assemblage of haplochromine cichlids (Kishe-Machumu *et al.*, 2015). These fish formed one of the richest vertebrate radiations known, with over 500 documented species, more than 95% of which are endemic to the Lake Victoria Basin (LVB) (Outa *et*

al., 2020). The ecological dominance of haplochromines in the Lake Victoria was abruptly disrupted following the introduction of exotic fish species in the 1950's, specifically the predatory Nile perch (*Lates niloticus*) and Nile tilapia (*Oreochromis niloticus*), which were introduced to enhance fishery productivity and economic returns (Kishe-Machumu *et al.*, 2015; Witte, Wanink, Kishe-Machumu, *et al.*, 2007). At first, the introduced Nile perch constituted a very small portion of the fish biomass, making

up less than 0.5% (Witte, Wanink, & Kishe-Machumu, 2007). However, by the early 1980's, its population had exploded, triggering a severe collapse in haplochromine diversity (Mkumbo & Marshall, 2015). Previous studies reported nearly 200 haplochromine species became extinct, marking one of the most substantial species losses in the LVB (Outa *et al.*, 2020). The decline of haplochromines was influenced by factors such as their smaller size and habitat overlap with Nile perch, which made them highly susceptible to predation (Witte, Wanink, & Kishe-Machumu, 2007). Additionally, the introduction of exotic Nile tilapia is suspected to have negatively impacted haplochromine biodiversity through competition for food and the introduction of diseases and parasites (Awiti, 2011). While destructive fishing practices played a role, Nile perch predation and eutrophication were the primary factors that caused the decline of haplochromine populations (Witte, Wanink, & Kishe-Machumu, 2007).

Despite efforts to manage the fishery, including the formation of Beach Management Units (BMUs) under a co-management framework, current conservation and policy efforts largely prioritize economically valuable species like Nile perch and Nile tilapia (Njiru *et al.*, 2014; Outa *et al.*, 2020). In contrast, haplochromines continue to face threats such as habitat loss, overfishing, and are used as bait in longline fisheries (URT, 2021b). Although some studies suggest a potential partial recovery of haplochromine populations, driven by a decline in Nile perch catch from 40% in 2011 to 30% in 2020 (Mkumbo & Marshall, 2015; URT, 2021a), the actual extent of haplochromine composition is uncertain due to a lack of recent inventory data. Furthermore, the morphological similarities and intraspecific variations among haplochromines complicate their identification and conservation, leaving the remaining

populations vulnerable to overexploitation (Witte, Wanink, & Kishe-Machumu, 2007). This data gap limits accurate species identification and constrains the determination of their species composition and biodiversity hotspots, limiting effective haplochromine conservation strategies. Therefore, this study assessed the composition of Haplochromine cichlids in Shirati Bay using DNA barcoding for precise species identification. The results will support more inclusive biodiversity assessments and inform efforts to restore the ecological integrity of Lake Victoria's fish communities.

MATERIALS AND METHODS

Study area description

This study was conducted in Shirati Bay (SB), located in the Mara region in Tanzania along the south eastern shore of Lake Victoria (Figure 1). The site was selected because is among the main landing site in Lake Victoria (URT, 2020). Also, the SB was selected due to its role as an important breeding and nursery ground supporting the recruitment of *Haplochromis* species (Waya *et al.*, 2017). Shirati Bay spans approximately 14 km² and has an average depth of 10 meters (Kashindye *et al.*, 2015). The region is characterized by a bimodal rainfall pattern, comprising two rainy seasons and a prolonged dry season from July to September (Waya *et al.*, 2017). Fishing is the predominant economic activity within the bay, where local communities depend heavily on the lake's resources.

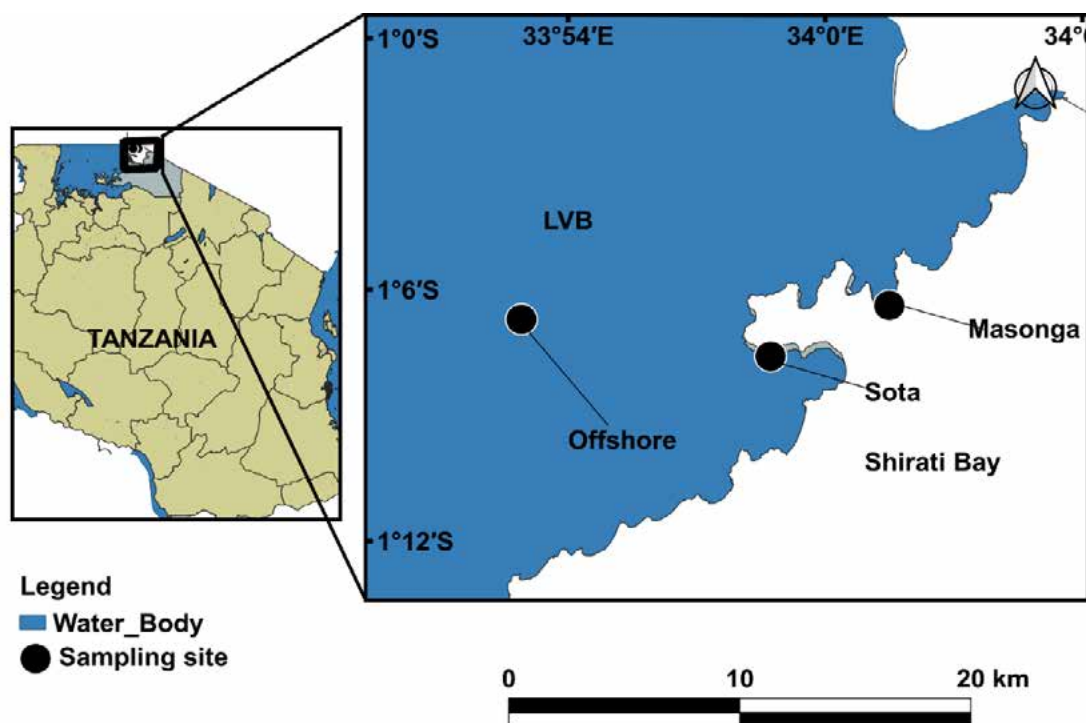


Figure 1. Map of the study area showing locations where haplochromine samples were collected. The map was created using QGIS software version 3.10.10 and shapefiles from the Database of Global Administrative Areas (GADM) (<https://gadm.org/maps/TZA.html>, accessed on 28th July 2025).

Fish sampling

Fish sampling was conducted at two primary landing sites within Shirati Bay: Sota and Masonga with an average distance of 12.87 km between sites, during February and March 2025 (Figure 1). These sites were purposively selected due to their high fishing activity and known presence of *Haplochromis* species. In addition, samples of haplochromine cichlids were obtained from local fishers fishing offshore of Shirati Bay to capture species from different habitats. Fish were collected using a beach seine net measuring 200 meters in length and 4 meters in width, with a cod-end mesh size of 4 mm. At each station, three separate hauls were conducted per sampling event. The harvested catch was sorted immediately to isolate *Haplochromis* species. A total of 32 individuals from each site were collected and identified using *A Guide to the Freshwater Fishes of Tanzania* (Eccless, 1992). Subsequently, approximately 2 g of

muscle tissue was dissected from the pelvic fin of each individual and preserved in 1.5 ml microcentrifuge tubes containing 99.9% ethanol for further analyses.

DNA extraction, COI amplification and sequencing

Genomic DNA was extracted from fin clip tissues using the FlAPure Animal Tissue/Cell/Blood DNA Extraction Kit (Genesand Biotech Co., Ltd, Beijing, China) in accordance with the manufacturer's guidelines. The integrity of the extracted DNA was assessed using a 1% agarose gel (Saiperaki *et al.*, 2024a). Afterwards, a 620 base pair fragment of the mitochondrial Cytochrome c Oxidase Subunit I (COI) gene was amplified using the primers FishF1 (5'-TCAACCAACCACAAAGACATTGGCAC-3') and FishR1 (5'-TAGACTTCTGGGTGGCCAAAGAATCA-3') (Saiperaki *et al.*, 2024b). PCR amplification was conducted in 40 µL reaction

volumes comprising 3.2 µL template DNA, OneTaq 2X Master Mix with Standard Buffer (New England BioLabs Inc., MA, USA), 5 mg bovine serum albumin (BSA), and 0.3 µM of each primer (Rumisha *et al.*, 2023). The thermal cycling profile included an initial denaturation at 94°C for 5 minutes, followed by 35 cycles of denaturation at 94°C for 40 seconds, annealing at 54°C for 45 seconds, and extension at 72°C for 60 seconds. A final extension step at 72°C for 15 minutes was included to ensure complete DNA elongation (Morandus & Rumisha, 2024). PCR products were visualized using 1.5% agarose gels stained with SafeView DNA stain and visualized under a blue-light transilluminator (Mbilinyi *et al.*, 2023). Successfully amplified products were sent for Sanger sequencing at MacroGen Europe Laboratory using the ABI 3730XL DNA Analyzer (Applied Biosystems, Foster City, USA) (Saiperaki *et al.*, 2024a).

Data analysis

A total of 94 haplochromine cichlid specimens collected from Shirati Bay were successfully sequenced. The resulting sequences were first trimmed and translated into amino acid sequences using the vertebrate mitochondrial genetic code in MEGA version 11 (Tamura *et al.*, 2021) to detect and exclude sequencing errors and pseudogenes (Morandus *et al.*,

2025). The refined sequences were then collapsed into distinct haplotypes using FaBox DNA Collapser version 1.61 (Villesen, 2007). Each unique haplotype was subsequently compared to sequences in the GenBank Nucleotide Database using the Basic Local Alignment Search Tool (BLAST) to assess similarity and confirm species identity (Fabiani *et al.*, 2023). A sequence match of ≥ 99.0% to known references was considered a successful identification (Saiperaki *et al.*, 2024a). Bayesian phylogenetic analysis was then performed using BEAST version 1.10.4 to reconstruct evolutionary relationships among the identified taxa (Morandus *et al.*, 2025). Additionally, the conservation status of each species was compiled from the IUCN Red List to support ecological interpretation and inform conservation measures (Rumisha *et al.*, 2024).

RESULTS

Identified *Haplochromis* species

A total of 94 COI sequences were obtained from the analysed samples. DNA barcoding confirmed the identity of nine haplotypes, with percentage identities ranging from 99.57 to 100 % (Table 1). Similarly, the expected value for all sequences analysed was 0, query coverage was 100 while the maximum score ranged from 857 to 869.

Table 1: Summary of the results obtained after comparing the obtained COI sequences of *Haplochromis* species from the SB, Tanzanian with those in the GenBank and BOLD databases.

Haplo type	Species	Base pair	Maximum score	GenBank		BOLD	
				Query cover (%)	E- value	% Identity	% Identity
H1	<i>H. nyererei</i>	470	869	100	0	100	100
H2	<i>H. erutus</i>	470	869	100	0	100	100
H3	<i>H. sauvagei</i>	470	863	100	0	99.79	99.79
H4	<i>H. nubilus</i>	470	No match	No match	No match	No match	100
H5	<i>H. nyererei</i>	470	863	100	0	99.79	99.79
H6	<i>H. simpsoni</i>	470	869	100	0	100	100
H7	<i>H. alluaudi</i>	470	857	100	0	99.57	99.57
H8	<i>H. nyererei</i>	470	857	100	0	99.57	99.57
H9	<i>H. pyrrhocephalus</i>	470	863	100	0	99.79	99.79

The constructed Bayesian phylogenetic tree provided additional confirmation to the identified fish species (Figure 2). Closely

related species were clustered under the same node implying that the amplified barcodes correctly identified the species.

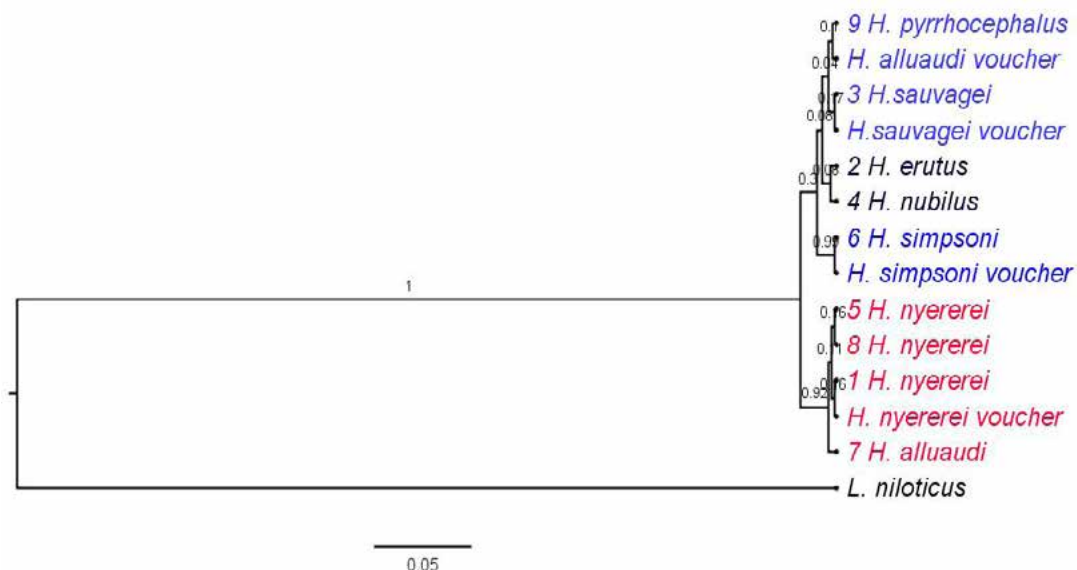


Figure 2. Bayesian phylogenetic tree showing the evolutionary relationships among cytochrome oxidase subunit I haplotypes of the sampled *Haplochromis* species

Haplochromis species composition and conservation status

The composition of the sampled *Haplochromis* species varied from 0% to 28.72% (Table 2). *Haplochromis nyererei* was the most abundant species, accounting for 28.72, 15.96% and 10.64% of samples collected from Sota, Masonga and Offshore site, respectively. Of

the seven *Haplochromis* species sampled, three (including *H. nyererei*, *H. alluaudi*, and *H. pyrrocephalus*) were classified as Least Concern, two (*H. sauvagei* and *H. nubilus*) as Vulnerable, one (*H. simpsoni*) as Endangered, and one (*H. erutus*) as Not Evaluated. Although *H. alluaudi* is categorized as Least Concern and *H. erutus* as Not Evaluated, both were recorded at very low abundances in the catch (Figure 3).

Table 2: Composition of sampled *Haplochromis* species and their conservation status.

Specie name	% Sota	% Masonga	% Off-shore site	IUCN Red list
<i>H. nyererei</i>	28.72	15.96	10.64	LC
<i>H. erutus</i>	1.06	0	0	NE
<i>H. sauvagei</i>	9.57	0	0	VU
<i>H. nubilus</i>	10.64	8.51	0	VU
<i>H. simpsoni</i>	1.06	0	0	EN
<i>H. alluaudi</i>	3.19	0	0	LC
<i>H. pyrrocephalus</i>	0	3.19	7.45	LC

VU - Vulnerable, LC - Least Concern, EN - Endangered, and NE - Not Evaluated.

Figure 3. Percentage composition of each *Haplochromis* species in the total catch

DISCUSSION

Haplochromis species landed in the Shirati Bay

This study identified seven distinct *Haplochromis* species from the SB: *H. nyererei*, *H. alluaudi*, *H. erutus*, *H. simpsoni*, *H. sauvagei*, *H. nubilus*, and *H. pyrrhocephalus*. All species, except *H. erutus* and *H. simpsoni*, have previously been reported in Tanzanian freshwater ecosystems (Froese & Pauly, 2025). However, this is the first record of *H. erutus* and *H. simpsoni* in the country and at the landing site. Previous studies reported *H. erutus* and *H. simpsoni* are endemic to Lake Edward and Lake Nabugabo respectively (Ogutu-ohwayo, 1993; Vranken *et al.*, 2020). Their occurrence in Shirati Bay could be attributed to hydrological connectivity, seasonal migrations, or potential range expansions facilitated by environmental changes, allowing these previously restricted species to colonize new habitats (Nagl *et al.*, 2000). However, because previous studies relied solely on morphological identification, it is possible that these species were present in Lake Victoria but were overlooked in earlier regional surveys due to their cryptic or morphologically similar characteristics. The barcoding approach used in this study effectively resolves the taxonomic uncertainties that have complicated earlier surveys, where *Haplochromis* species were frequently grouped together because of identification difficulties. Additionally, the phylogenetic analysis clustered sequences of the sampled *Haplochromis* species into well supported clades, confirming the taxonomic validity of the morphologically identified species. This molecular confirmation enhances taxonomic resolution, particularly for morphologically similar species, and supports more effective conservation strategies by ensuring that species-level data guide management decisions. Additionally, the results stress the importance of updating global taxonomic databases such as FishBase and BOLD Systems to reflect this refined understanding, thereby improving the accuracy and relevance of

conservation planning, ecological monitoring, and biodiversity reporting.

Composition and Conservation Status of the sampled *Haplochromis* species

The findings revealed that *H. nyererei* was the most abundant haplochromine species across all sites. Previous studies did not report this species individually, often grouping *Haplochromis* species due to the morphological challenges in distinguishing species (Kishe-Machumu *et al.*, 2015). The high abundance of *H. nyererei* may be attributed to its ecological versatility, allowing it to occupy both shallow and deeper habitats. Its feeding plasticity and aggressive reproductive behavior likely contribute to its ecological success and dominance (Katunzi *et al.*, 2003). Conversely, the low representation of *H. simpsoni* and *H. erutus* suggests that these species may be more habitat-specific or sensitive to environmental stressors such as overfishing and habitat modification. Their limited presence in the catch highlights a potential population decline and calls for urgent conservation action. Furthermore, the observed low abundances of *H. simpsoni*, *H. erutus*, and *H. alluaudi* highlight an important discrepancy between IUCN conservation categories and local catch composition. *H. simpsoni*, although classified as Endangered, appears underrepresented at the landing site, emphasizing the potential for localized overexploitation and the need for targeted monitoring. Conversely, *H. alluaudi*, categorized as Least Concern, was also recorded at very low abundance, suggesting that broad conservation assessments may not fully capture site-specific population dynamics. The rarity of *H. erutus*, currently Not Evaluated, highlights the importance of reassessing its conservation status in Lake Victoria, particularly given its low representation in catches. These findings indicate that conservation status at a global or regional scale may not always align with local abundance patterns, highlighting the need for site-specific surveys to inform management and conservation strategies

CONCLUSION

This study recorded seven haplochromine cichlid species in Shirati Bay, including the first national records of *Haplochromis erutus* and *H. simpsoni*, species previously considered endemic to Lake Edward and Lake Nabugabo, respectively. Their detection may reflect range expansions or distributional shifts, but it is also possible that these species were previously present and overlooked due to limitations in earlier surveys. The dominance of *H. nyererei* across all sampling sites suggests a shift in species composition, potentially linked to the decline of Nile perch populations in Lake Victoria. However, the limited number of reference barcodes in nucleotide databases highlights the need to update and expand these libraries to facilitate accurate species detection and effective conservation monitoring. Similarly, the low abundances of *H. simpsoni*, *H. erutus*, *H. alluaudi* and other rare *Haplochromis* species, combined with the mismatch between their conservation status and local catch composition indicates the need for periodic, site-specific reassessments of conservation status, particularly for rare and newly recorded species to ensure that management and conservation strategies are based on up-to-date local population data. Additionally, management strategies should prioritize the protection of these rare and newly detected species through sustainable fishing practices, habitat preservation, and community engagement to raise awareness about practices that minimize impacts on fish populations. The establishment of protected areas or fish refugia within Shirati Bay is recommended to safeguard vulnerable *Haplochromis* species and prevent further declines. Furthermore, future studies should utilize hypervariable genetic markers such as simple sequence repeats (SSRs) and single nucleotide polymorphisms (SNPs) to examine population structure, genetic diversity, and connectivity. Such targeted research is essential for informed conservation planning, monitoring the spread of newly detected species, and ensuring the long-term ecological

sustainability of haplochromine cichlids in Tanzania.

ETHICAL STATEMENT

All handling of haplochromine cichlids and experimental procedures were conducted in strict accordance with the Animal Ethics Guidelines established by the Sokoine University of Agriculture Research and Ethics Committee, ensuring minimal stress and ethical compliance throughout the study.

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HOW PREY THREATENS SURVIVAL OF PREDATOR: A FIELD EXPERIENCE ON AFRICAN LION IN NYERERE-SELOUS COMPLEX

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ABSTRACT

The African Lion (*Panthera leo*), a vulnerable species as per the IUCN Red List, faces numerous threats across its range like other large wild carnivores which subject them to increase extinction risks. While human-lion conflict and habitat loss are well-documented, the direct mortality and morbidity inflicted upon lions by their prey are less understood. This report presents the findings on the monitoring of health events concerning lions in Nyerere National Park (NNP) and Selous Game Reserve (SGR) between 2023 and 2025. Through information gathered from rangers, tour guides and alerts from GPS collars deployed on lions, we have managed to make follow up on nine incidences out of 32 lions monitored. Three events were received as mortality alerts and six as injuries, hence treatment of wounds, autopsy and collection of samples for toxicology were conducted. The case fatality rate was 44.4% (n=4), while 55.6% (n=5) of individuals were successfully treated and recovered. Of the nine cases, eight (89%) involved physical injuries, with 37.5% (n=3) caused by snares and 62.5% (n=5) identified as traumatic wounds inflicted by prey. One mortality was of unknown cause, and all dead lions were solitary females with no signs of poisoning. Asymmetrical appearance, sharpness and depth of the wounds, lack of scratches and bites, presence of herbivore footprints and struggling signs on the sites qualifies for traumatic wounds due to prey. Our findings indicate that in the Nyerere-Selous Complex, prey species particularly large ungulates like the African Buffalo (*Syncerus caffer*) constitute a significant, and often overlooked, source of lion mortality and injury. We recommend intensified ecological studies on predator-prey dynamics, enhanced veterinary surveillance, and rapid-response protocols to mitigate these health risks, which are likely exacerbated by habitat fragmentation and changing ecosystem dynamics.

Key Words: African lions, Nyerere-Selous, Prey-induced mortality, Traumatic injuries, Tanzania.

INTRODUCTION

The African Lion (*Panthera leo*) is an apex predator that plays a critical role in maintaining the structure and function of savanna ecosystems (Bauer *et al.*, 2015; Davies-Mostert *et al.*, 2018). Despite their ecological dominance, lion populations are in decline across Africa, leading to their classification as Vulnerable on the IUCN Red List (Bauer *et al.*, 2016). The primary threats to their survival are widely recognized as habitat loss, prey base depletion, and retaliatory killing due to human-lion conflict (Packer *et al.*, 2013; Bauer *et al.*, 2016).

The Nyerere-Selous-Mikumi ecosystem in southern Tanzania represents one of the continent's most significant lion strongholds, supporting a robust population within a vast landscape inhabited by over 268,000 wild animals, including substantial numbers of formidable prey species like the African Buffalo (*Syncerus caffer*) (TAWIRI, 2023). While this ecosystem provides a relative sanctuary, lions are still subject to population dynamics influenced by various mortality factors.

Traditionally, intraguild competition with other large carnivores such as Spotted Hyenas (*Crocuta crocuta*) is considered a significant

natural cause of lion mortality (Swanson *et al.*, 2014). However, the risk posed by their prey is often underestimated. Large herbivores are not passive victims; they are powerful and dangerous animals that vigorously defend themselves and their young. Encounters with species like buffalo, giraffe, and even large antelope can result in serious or fatal injuries for predators (Hayward & Kerley, 2005).

Rather than seeing predators as the only danger, this research reveals a more complex arms race between lions and their prey. Field data from the Nyerere-Selous Complex show that prey can seriously injure lions, and these wounds often lead to sickness or death, posing a real threat not just to individual animals but to the health of the entire population.

MATERIALS AND METHODS

Study area

The study was conducted in the Nyerere-Selous Complex, which encompasses the Nyerere National Park (formerly the northern sector of the Selous Game Reserve) and the Selous Game Reserve proper. This is a UNESCO World Heritage Site covering about 50,000 km² of miombo woodland, savanna, and riparian habitats. It hosts high densities of wildlife, including key lion prey species such as African Buffalo, wildebeest, zebra, warthogs and various antelopes.

Lion monitoring and incident reporting

Between January 2023 and March 2025, we monitored 32 lions (7 males, 25 females) of which 8 were fitted with GPS satellite collars (Vectronic VERTEX PLUS IRIDIUM collars-Aerospace GmbH). Collars were programmed to transmit location data daily

and to send mortality/immobility alerts if the collar remained stationary for over 12 hours. A network of park rangers and tour guides was requested to report sightings of injured or behaving abnormally lions. These reports, combined with GPS alerts, formed the basis for initiating field investigations. Sometimes a vet team conducted unplanned visits to the prides for health monitoring.

Field investigation and data collection

Upon receiving an alert or report, a rapid response team comprising a wildlife veterinarian and rangers, sometimes with ecologists was dispatched. For live animals, the team attempted to approach from a safe distance for visual assessment. If intervention was deemed necessary, the animal was immobilized using a combination of 200-400mg ketamine or 100-150mg Zoletil and combined with 4-6mg medetomidine delivered via a dart gun.

A thorough physical examination was conducted, including documentation and photography of all wounds (Photo 1), assessment of wound dimensions and depth, collection of tissue and hair samples from wound margins and collection of blood samples for baseline serology. For dead animals, a full field necropsy was performed. The examination focused on determining the cause of death, documenting the nature, location, and severity of all injuries, assessing body condition, collecting visceral samples (liver, kidney, stomach content) for poison screening using laboratory-based toxicology assays (at Government Chemist Laboratory Authority), and searching for signs of other predators (bite marks, scratches) or human activity (snare, bullet wounds, spears).



Photo 2: The traumatic wounds on two lionesses that were inflicted by preys in Nyerere National Park

Categorization of wounds

The wounds were categorized based on forensic evidence at the scene and on the animal. For instance, snare wounds were characterized by circumferential, deep lacerations typically around the neck, torso, limbs, or pelvic, often with embedded wire or signs of severe infection. Prey-induced traumatic wounds were identified by deep, penetrating, and often asymmetrical wounds consistent with goring by horns or tusks. Key supporting evidence included the absence of carnivore tooth marks or scratches, the presence of specific herbivore footprints and struggle signs (disturbed soil, broken vegetation) at the location, and the pattern of internal trauma (e.g., punctured organs, internal haemorrhage). Cases of animals died some days ago, already predated and with no clear cause could be determined, were classified as of unknown cause.

RESULTS

Incident overview

Over the 27-month monitoring period, nine significant health incidents were investigated from the cohort of 32 monitored lions (Incidence Rate: 28.1%). Three incidents (33.3%) were initial mortality alerts, and six (66.7%) were injury reports. The overall outcome was a 44.4% (n=4) mortality rate and a 55.6% (n=5) treatment and recovery rate.

Causes of mortality and morbidity

The primary causes of injury and death are summarized in Table 1. The vast majority of incidents (88.9%, n=8) involved physical trauma. Prey-induced injuries were the most frequent cause, accounting (Photo 2) for 62.5% of all trauma cases (n=5). Snare-related injuries accounted for 37.5% (n=3). One mortality had no clear cause as the carcass was already predated; the cause of death remains undetermined (Photo 2).



Photo 3: The collared lioness died of unknown cause (Left), and the dead buffalo (right) with tooth bites on the neck (middle) that was associated with severe wounding of one of the lionesses as was found aside of the injured lioness.

Table 1: Summary of health incidents and outcomes for monitored lions in the Nyerere-Selous Complex (2023-2025).

Case ID	Sex	Group size	Pride status	Incident Type	Cause	Outcome
SGR-1	M	2	Collared	Injury	Snare	Treated & released
NNP-1	M	13	Collared	Injury	Prey	Treated & released
NNP-2	F	5	Non-collared	Injury	Snare	Treated & released
NNP-3	F	13	Collared	Injury	Prey	Treated & released
NNP-4	F	9	Non-collared	Mortality	Prey	Died
NNP-5	M	4	Non-collared	Injury	Snare	Treated & released
NNP-6	F	2	Collared	Mortality	Unknown	Died
NNP-7	F	1	Collared	Mortality	Prey	Died
NNP-8	F	4 (2 cubs)	Collared	Mortality	Prey	Died

Demographics of affected lions

Out of nine affected lions, eight were adults and one was sub-adult, of which 33.3% were males and 66.7% were females. All four lions that died (100% of mortalities) were females. Furthermore, 75% (n=3) of these mortalities were a direct result of encounters with prey.

Characteristics of prey-induced injuries

Investigative analysis of the five prey-induced trauma cases revealed consistent

patterns: *Wound morphology:* Wounds were deep, penetrating, and often singular or asymmetrically placed, consistent with the horn of a buffalo or tusk of warthog. They were notably «clean» and sharp, lacking the tearing associated with carnivore bites.

Internal damage: Necropsies on fatal cases revealed severe internal trauma, including punctured lungs, livers, inguinal and massive internal haemorrhage.

Scene evidence: In all five cases, the incident

site was characterized by signs of a violent struggle, trampled vegetation, disturbed soil and the presence of fresh African Buffalo footprints. No tracks of other large carnivores were found in the immediate vicinity. However, in one large female with signs of lesions due to warthog tusk in inguinal area, her gut contents were dominated with warthog flesh.

DISCUSSION

Our findings provide compelling evidence that prey species, particularly the African Buffalo and warthogs, represent a major and underappreciated threat to lion survival in the Nyerere-Selous Complex. Prey-induced trauma was the leading cause of verified injury and mortality in our monitored population, exceedingly even the impacts of illegal snaring. The vulnerability of females in prides with two adults or less is a critical finding. The individual hunting is a known strategy, especially for sub-adults dispersing or adults separated from their pride, it appears to carry an exceptionally high-risk including injuries from prey and reduced hunting efficiency (Hayward & Kerley, 2005). Hunting large, dangerous game like buffalo is a cooperative endeavour for lions (Funston *et al.*, 2001). A solitary hunter lacks the support of pride-mates to distract and subdue the prey, thereby increasing its exposure to defensive counterattacks. The death of these females has a disproportionate impact on population growth, as they represent the core reproductive unit of the population.

The African Buffalo is renowned for its defensive ferocity and will often stand its ground and counter-attack when threatened (Estes, 1991). Our scene investigations consistently pointed to buffalo as the primary antagonist in these traumatic encounters followed by warthogs. This dynamic underscore the perpetual “arms race” between predator and prey, where the success of one is constantly balanced by the defensive capabilities of the other. The high incidence of such events in the Nyerere-Selous Complex could be influenced by several factors. The ecosystem supports a very high density of

buffalo, increasing the frequency of dangerous encounters (TAWIRI, 2023). Furthermore, ecological pressures such as seasonal resource availability or habitat fragmentation may be forcing lions, particularly more vulnerable individuals, to take greater risks when hunting.

CONSERVATION IMPLICATIONS

The significant mortality from natural prey conflicts has direct implications for conservation management.

Veterinary intervention: The successful treatment and release of 55.6% of reported cases highlights the invaluable role of proactive veterinary services. Rapid response to health threats can convert potential mortalities into survivors, directly bolstering population numbers.

Enhanced monitoring: Our study demonstrates the efficacy of a combined monitoring system using GPS telemetry and human intelligence networks. Strengthening this system is crucial for the early detection of health threats. This can be improved by opening more tourism circuits in NNP especially in northern area of the park where there are large populations of preys including wildebeests, African buffalos, hartebeests, elands, and zebras. Presence of these preys attracts predators to the area, and therefore increased tourism activities in the area not only will improve surveillance on health threats but also anti-poaching operations.

Ecological research: There is a pressing need for focused research on predator-prey dynamics, specifically investigating whether certain environmental conditions or individual lion behaviours predispose them to a higher risk of injury. This should be linked with health monitoring studies to unravel the common lion silence killers, diseases and genetic complications.

CONCLUSION

This field experience from the Nyerere-Selous

Complex reframes the conventional narrative of the predator-prey relationship. The prey, in its struggle for survival, becomes a potent threat to the predator. For the Vulnerable African Lion, traumatic injuries inflicted by large herbivores like the African Buffalo are not merely occasional hazards but a significant source of mortality, particularly for solitary individuals. As global habitats continue to fragment, potentially altering predator-prey interactions, understanding and mitigating these natural, yet population-impacting, risks become paramount. We advocate for the integration of robust health surveillance and rapid veterinary response as essential components of lion conservation strategies in this and similar ecosystems.

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ASSESSING THE INFLUENCE OF WILDLIFE HERBIVORY, FIRE AND CLIMATE ON GRASS GROWTH IN MIKUMI NATIONAL PARK, TANZANIA.

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ABSTRACT

Understanding how herbivory, fire, and climate influence grass productivity is essential for managing and sustaining wildlife and ecosystem services in semi-arid savanna landscapes. This study aimed to assess the effects of wildlife herbivory, fire occurrence, and climatic factors on grass growth in Mikumi National Park from 2020 to 2024. Sixty permanent vegetation plots were monitored quarterly using systematic step point transect method, in which grass height was measured and recording evidence of herbivory and fire. Rainfall and temperature were concurrently recorded. Results indicated that grazing intensity was high (57%) during wet season and low during dry season (43%). The results indicate that no single factor (rainfall, temperature, fire, and season) acts in isolation to influence grass growth. Instead, grass growth is influenced by synergistic non-linear combination of interactions among these factors. Effective management of grass growth as potential rangelands, therefore, requires an integrated approach that recognizes the ecological linkages connecting herbivore distribution, burning practices, and climatic variability. The findings emphasize the importance of managing fire regimes and monitoring forage availability to maintain ecological balance and support wildlife populations. Integrating environmental monitoring of climate elements with adaptive management strategies can enhance resilience of savanna ecosystems under changing climatic conditions.

Key Words: Climate variability, Fire effects, Grass–herbivore interactions, Habitat management, Savanna vegetation.

INTRODUCTION

Savanna ecosystems compose the primary foundation of East African protected-area food webs (Scholes & Archer, 1997; Veldhuis *et al.*, 2016), covering approximately 20% of earth's land surface, supporting immense biodiversity providing essential ecological services for wildlife and livestock including forage, carbon sequestration, and drive nutrient and carbon cycling across landscapes (Le Stradic, 2017; Veldhuis *et al.*, 2016; Stevens *et al.*, 2021). Tanzania's savanna ecosystems like other ecosystems face escalating pressures from intensive herbivore pressures, recurrent wildfires, and climate variability which alter vegetation dynamics and ecosystem integrity

(Bond & Keeley, 2005; Riginos *et al.*, 2012). In many of the Tanzanian protected areas, the quantity and quality of grass directly influence wildlife population dynamics and the delivery of ecosystem services that support wildlife, livestock and neighbouring communities (McNaughton *et al.*, 1998). Herbivores, in particular large mammals, exert top-down control on grass productivity through selective grazing, potentially limiting fuel loads for fires while influencing nutrient cycling (Ritchie *et al.*, 2011; Karp *et al.*, 2024). Research has shown that managed grazing and seasonal resting can substantially enhance grass regrowth and maintain herbaceous diversity in semi-arid rangelands (Baumgartner *et al.*, 2024). However, excessive grazing can

degrade grasslands, reducing their resilience to drought and fire (Zaher *et al.*, 2023), while climate change exacerbates these stressors, with projections indicating declines in rainfall and temperature increases, disrupting grass phenology and productivity (World Bank, 2022).

The empirical and experimental studies across East Africa demonstrate that grazing can both suppress biomass and stimulate compensatory regrowth and foraging quality under grazing and fire intensity, and climate variability (McNaughton *et al.*, 1998). Although general patterns on grazing regimes including early burns to promote rapid post-fire growth and rainfall variability controls interannual biomass are well established, there remain important local gaps on how wildlife grazing pressure, fire and climate influence grass growth in Mikumi National Park (MNP) to predict observed grass growth and quality across the park's heterogeneous habitats. This study, therefore, seeks to understand the influence of wildlife grazing pressure, fire regimes and climate variability on grass growth in MNP. To achieve this, we compared grass growth alongside assessing factors influencing grass growth across seasons of the year.

MATERIALS AND METHODS

Study area

Mikumi National Park is located in south-eastern Tanzania's Morogoro Region, covering 3,230 km², and lies within 7°05' to 7°50' South, and 36°50' to 37°35' East, about 300 km west of Dar es Salaam city, forming the northern part of the expansive Nyerere-Selous-Mikumi ecosystem (Gunn *et al.*, 2013). The park spans lowland savannas, miombo woodlands and floodplain grasslands (Backéus *et al.*, 2004). Mean annual rainfall is 1,000 mm with bimodal wet season spanning from November to May and dry season from June to October, whose temperatures average 24°C during wet season and above 35°C during drier seasons. The elevation in the park ranges from 500 metres above sea level on the floodplain to 2743 metres in the mountain (Maige 1990). The area is home to diverse herbivores including elephants and zebras, MNP experiences variable grazing pressures influenced by seasonal migrations and water availability (Gereta *et al.*, 2002). Within miombo systems, local nutrient hotspots such as termite mounds have been found to influence forage quality and attract grazing herbivores, further shaping vegetation patterns (Moyo *et al.*, 2020). The study was conducted on permanent vegetation plots; within the Viramba ranges located approximately 10 km north-northeast to the northern border of MNP (Fig. 1).

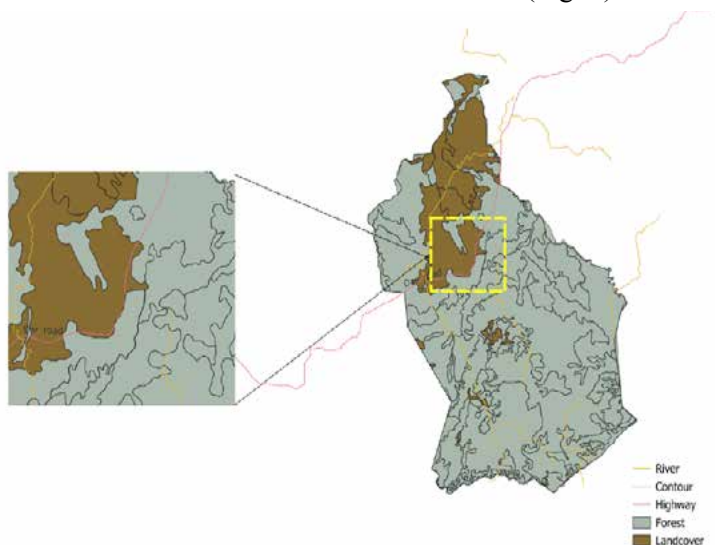


Figure 1: Map indicating locations of vegetation plots monitored in Mikumi National Park (from 2020-2024).

Study design

We used longitudinal observational study design using permanent vegetation plots, monitoring grass growth. A total of 60 permanent plots, each measuring 40m x 40m, were monitored quarterly according to four climatic seasons namely, Early Wet (December-February), Late Wet (March-May), Early Dry (June - August), and Late Dry (September- November) from January 2020 - December 2024. Monitoring was conducted from January 2020 to December 2024. Each plot had a central reference point marked with a steel post to guide all sampling activities. From this fixed point, vegetation sampling was conducted along four transects extending in the cardinal directions (North, East, South, and West).

Data Collection

We used step transect method for data collection within vegetation plots, with a central steel post marking the sampling origin. Data collection was conducted on the 44 systematic steps in cardinal directions from the centre, where at each step-point grass height was measured to the nearest 0.1 cm using standardized vegetation meter ruler, using visual indicators on fire burn status was recorded as burned/unburned, grazing status on grazed/ungrazed through observation of cropped blades, animal tracks were also recorded, and fire based on visual indicators like charred stems and ash residues) and grazing evidence (documenting grazed/ungrazed status through observation of cropped blades, animal tracks). We further incorporated temperature and rainfall data collected at MNP.

Data Analysis

A descriptive analysis was conducted using R version 4.3.1 to produce frequency distribution graph on seasonal comparison of grass grazing pressure, inferential statistics were run by checking the normality assumptions using Shapiro-Wilk test, where the data were not normally distributed. Wilcoxon rank-sum

test was conducted to test for the correlation between grazing and season. Furthermore, multiple linear regression model was used to predict interactive relationship between grass growth, rainfall, grazing, and season.

RESULTS

Comparison of grass growth across seasons under different grazing levels

Figure 2 demonstrates wildlife grazing intensity recorded during dry and wet seasons within MNP. Wet season had the highest grazing intensity of 57%, while dry season has the lowest grazing intensity of 43% (Fig. 2). The Wilcoxon rank-sum test for the correlation between grazing and season ($W = 9,206$, $P = 0.4749$), indicated that there is no statistically significance difference between wildlife grazing pressure and seasonality. This implies that wildlife's grazing pressure is not statistically different across dry and wet seasons.

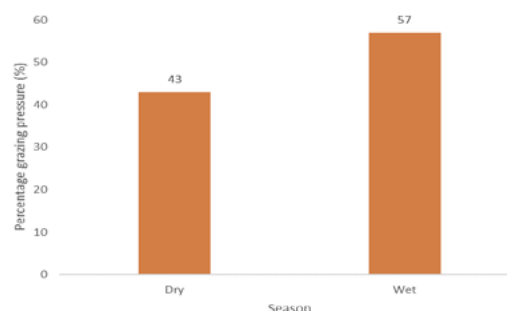


Figure. 2: Wildlife grazing intensity across seasons in 60 permanent plots monitored in Mikumi National Park from 2020 - 2024.

Factors influencing grass growth

The analysis shows that grazing intensity by itself does not significantly influence grass height. However, the interactive combination of rainfall, burning percentage (fire), and Season wet ($p = 0.017$), significantly influence grass height. Also, the combination of temperature, rainfall, burning percentage (fire), and Season

wet ($p=0.009$), influence grass height. Multiple regression analysis results on predictors of grass growth are shown in Annex 1.

DISCUSSION

Comparison of Grass Growth across Seasons under different Grazing Levels

Grass growth in Mikumi National Park (MNP) was influenced by both natural and prescribed fires that commonly occur during the dry season, leaving much of the rangeland sparsely vegetated and less available for wildlife grazing. Limited rainfall during this period further reduces soil moisture, constraining grass regeneration and overall productivity. Consequently, grazing pressure on vegetation plots remains low due to the limited availability of forage. In contrast, the wet season provides adequate rainfall that nourishes the rangeland, promotes vigorous grass growth, and increases forage availability for herbivores. Wildlife grazing pressure thus rises during the wet season when vegetation is abundant and accessible.

Several studies support these observations, indicating that seasonal variation strongly influences grassland productivity and grazing intensity. For example, Frank *et al.* (2016) in Yellowstone National Park reported that grazing intensity and ungulate species composition significantly affected aboveground net primary production (ANPP), with elk-dominated communities removing less biomass than bison-dominated ones. Similarly, research in East African savannas has shown that rainfall patterns and fire regimes shape vegetation structure, influencing herbivore distribution and foraging behavior (Augustine & McNaughton, 2004; Sinclair *et al.*, 2015; Archibald & Hempson, 2016). Recent Tanzanian studies also reveal that managed grazing during wet seasons enhances grass regrowth and maintains forage quality in semi-arid savannas (Baumgartner *et al.*, 2024; Moyo *et al.*, 2020).

These findings align with the present study, highlighting that rainfall variability and fire occurrence are central drivers of grass growth and wildlife grazing pressure. The results underscore the importance of maintaining ecological balance through appropriate fire management and water conservation. Park authorities should regulate fire frequency and timing to prevent excessive vegetation loss during the dry season while promoting sustainable regrowth before the onset of rains. Monitoring grazing intensity is recommended to avoid overutilization during the wet season, and integrating local weather data with vegetation monitoring could enhance predictive management of forage availability, thereby supporting both wildlife populations and ecosystem stability.

Factors influencing grass growth

Grass growth in this ecosystem is highly synchronized with numerous factors like rainfall, fires, temperature and season leading to grass growth dynamics. However, the results explain very little of the variation in grass height, suggesting that other variables such as soil conditions, grazers density, and feeding guild composition should be considered as important drivers of grass growth. Grass growth in the study plots appears to be primarily influenced by the combined interaction of environmental factors rather than grazing pressure alone. These patterns suggest that combinations of variables such as rainfall, burning percentage, temperature, and wet-season timing were significant predictors. This demonstrates that grass growth is multifactorial, determined by the synergistic effects of climatic conditions, fire, and seasonal dynamics.

Previous studies corroborate the importance of interacting environmental drivers in shaping grassland productivity. Frank *et al.*, (2016) reported that ungulate grazing impacts vary with species composition. Similarly, Archibald *et al.*, (2010); Augustine & McNaughton, (2004) observed that rainfall and fire regimes regulate vegetation recovery and structure,

influencing herbivore foraging patterns. Sinclair *et al.* (2015) also emphasized that seasonal timing and resource availability determine herbivore distribution and grazing intensity. Recent evidence from Tanzanian miombo systems supports this, showing that combined fire–grazing management promotes vegetation heterogeneity and resilience (Nyakunga, 2016; Zaher *et al.*, 2023).

The findings highlight the need for integrated rangeland management that considers fire regimes, seasonal, rainfall, temperature, and grazing dynamics as drivers of grass growth. Early and controlled burnings should be timed to optimize post-fire regrowth prior to peak grazing periods, while water and soil management strategies should maintain forage availability during dry seasons. Incorporating variables such as soil fertility, grass species composition, and post-fire recovery into predictive models could further improve understanding of grass growth patterns for maintaining vegetation resilience, supporting wildlife forages, and sustaining ecosystem stability. Also, monitoring grazing intensity is also recommended to avoid overutilization during the wet season, while integrating local weather data with vegetation monitoring could enhance predictive management of forage availability and support both wildlife populations and ecosystem stability.

CONCLUSION AND RECOMMENDATIONS

This study demonstrate that grass growth dynamics in Mikumi National Park and other similar areas are influenced by a complex interplay of interactive ecological drivers namely rainfall, fire regimes, rainfall, and seasonal–interannual climatic variability rather than thinking of grazing pressure alone. Fire occurrence and limited rainfall during the dry season reduce vegetation cover and restrict forage availability, while the wet season enhances grass regeneration and attracts higher grazing activity. The net effect of grass growth on seasonal grass

production could potentially depend on other factors like soil conditions, grazers density, and feeding guild composition. Sustaining Mikumi’s grasslands requires managing the balance between herbivores, fire, and climate variability. Effective management, therefore, requires an integrated approach that recognizes the ecological linkages connecting herbivore distribution, sustainable burning practices, and climatic rhythms. Potential recommendations include strengthening adaptive management strategies that incorporate climate variability to improve the resilience of rangeland ecosystems. Further, spatial-temporal monitoring of grazing intensity is also recommended to avoid overutilization, while integrating local weather data with vegetation monitoring could enhance predictive management of forage availability and support both wildlife populations and ecosystem stability.

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Annex 1: Regression Analysis Model Results on predictors (rainfall, temperature, fire, and seasons) to influence grass productivity

Predictors	Estimates	Std error	Statistic	P. value
Mean Monthly Temperature`	1.781241e-01	5.306421e+00	0.03356766	0.973248807
`Annual Rainfall`	4.154337e+00	1.222511e+01	0.33981995	0.734278587
`Grazing pressure`	5.255275e-02	6.456073e+00	0.00814005	0.993511769
`Seasonal Burning Percentage`	-4.496602e-01	1.812560e+00	-0.24808013	0.804276870
Season _Wet	1.861475e+02	1.461487e+02	1.27368561	0.203962499
`Combined; Temperature& Rainfall`	-2.537823e-01	4.673945e-01	-0.54297225	0.587634608
`Combined; Temperature & Grazing Pressure`	-3.246687e-02	2.489488e-01	-0.13041586	0.896342721
`Combined; Rainfall & Grazing Pressure`	-6.187355e-03	5.770440e-01	-0.01072250	0.991453432
`Combined; Temperature & Burning Percentage`	-9.715771e-03	7.149587e-02	-0.13589277	0.892015851

Predictors	Estimates	Std error	Statistic	P. value
`Combined; Rainfall & Burning Percentage`	-6.527415e-02	1.647875e-01	-0.39611107	0.692362304
`Combined; Grazing pressure & Burning Percentage`	2.282126e-02	8.968605e-02	0.25445723	0.799352358
`Combined; Temperature & Season_Wet`	-8.895642e+00	5.856116e+00	-1.51903449	0.130022500
`Combined; Rainfall & Season_Wet`	-2.252528e+01	1.451145e+01	-1.55224152	0.121874504
`Combined; Grazing pressure & Season_Wet`	-4.258041e+00	6.880503e+00	-0.61885608	0.536576920
`Combined; Burning percentage & Season_Wet`	-3.695309e+00	2.218147e+00	-1.66594414	0.096982034
`Combined; Temperature, Rainfall & Grazing pressure`	2.778841e-03	2.145424e-02	0.12952411	0.897047512
`Combined; Temperature, Rainfall & Burning percentage`	3.496545e-03	6.246275e-03	0.55978083	0.576132152
`Combined Temperature, Grazing pressure & Burning percentage`	-5.231176e-04	3.419984e-03	-0.15295910	0.878554407
`Combined Rainfall, Grazing & Burning percentage`	-1.934716e-03	8.639189e-03	-0.22394647	0.822982590
`Combined Temperature, Rainfall & Season Wet`	1.025698e+00	5.478833e-01	1.87210958	0.062363608
`Combined Temperature, Grazing percentage & Season Wet`	2.133764e-01	2.646488e-01	0.80626233	0.420860961
`Combined Rainfall, Grazing percentage & Season Wet`	3.862295e-01	6.426289e-01	0.60101479	0.548376844
`Combined Temperature, Burning percentage & Season Wet`	1.729695e-01	8.881916e-02	1.94743410	0.052606764
`Combined Rainfall, Burning percentage & Season Wet`	6.474949e-01	2.700230e-01	2.39792510	0.017225257

Predictors	Estimates	Std error	Statistic	P. value
Combined Grazing pressure, Burning percentage & Season Wet	-1.355459e-01	7.837324e-02	-1.72949241	0.084960281
`Combined Temperature, Rainfall, Grazing pressure & Burning percentage	4.277591e-05	3.184261e-04	0.13433544	0.893245848
`Combined Temperature, Rainfall, Grazing & Season Wet	-1.948574e-02	2.380060e-02	-0.81870784	0.413735967
`Average Temperature`: average Rainfall`: Burn: Season Wet	-3.211112e-02	1.226031e-02	-2.61911151	0.009356861
`Average Temperature`: Grazed: Burn: Season Wet	1.030181e-02	6.444722e-03	1.59848733	0.111202702
`average Rainfall`: Grazed: Burn: Season Wet	-3.633402e-03	5.403971e-03	-0.67235779	0.501979370
`Average Temperature`: `average Rainfall`: Grazed: Burn: Season Wet	NA	NA	NA	NA

ENVIRONMENTAL AND ANTHROPOGENIC DRIVERS OF SPATIAL DISTRIBUTION OF COMMON ELAND IN BURIGI CHATO NATIONAL PARK, TANZANIA.

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ABSTRACT

Eland (*Tragelaphus oryx*) is the second-largest antelope species in Africa and is a highly adaptable mixed feeder, depending on seasonal resource availability. It occupies a wide range of habitats, including open savanna, grasslands, acacia woodland, and wooded bushlands. Across Southern Africa, eland distribution is shaped by a combination of environmental and human factors. Major environmental factors include rainfall, habitat type, water availability, land surface temperature (LST), and vegetation greenness (NDVI), while human-related factors include cultivation, roads, livestock, and settlements. This study examined how environmental and human-related factors influence the spatial distribution patterns of elands in Burigi-Chato National Park (BCNP). Specifically, the study aimed to; (i) assess influence of environmental variables: Land Surface Temperature (LST), vegetation greenness (NDVI), and distance to water on eland distribution; (ii) evaluate the influences of human activities: livestock keeping, cultivation, and settlements; and (iii) integrate these factors within a spatial modelling framework to identify suitable habitats and potential human-wildlife conflict zones within the park. The study used eland and human activities data from a Total Count (TC) aerial survey conducted by TAWIRI over the BCNP in September 2025. Elevation data were extracted from a 30m resolution Digital Elevation Model (DEM), NDVI and rainfall were derived from Moderate Resolution Imaging Spectroradiometer (MODIS) data. The zero Inflated Negative Binomial Model was used to determine the influence of environmental and anthropogenic variables on the spatial distribution of eland. Eland distribution is positively influenced by vegetation productivity NDVI ($\beta = 95.254$; $p = 0.0003$) and favorable microclimatic conditions LST ($\beta = 1.624$; $p = 0.0006$), while being strongly constrained by the presence of livestock ($\beta = -5.254$; $p = 0.0004$), there were more sightings in the grasslands ($\chi^2 = 272.87$, $df = 3$, $p < 0.05$). To ensure the sustainability and distribution of elands in BCNP, management should prioritize forage-rich habitats and control encroachment into park boundaries.

Keywords: Eland, human activities, modelling, NDVI, protected areas, spatial distribution

INTRODUCTION

Eland (*Tragelaphus oryx*) is the second-largest antelope species in Africa (Hillman, 2008). It is a highly adaptable mixed feeder that grazes and browses depending on seasonal resource availability (Gagnon & Chew, 2000). The common eland occupies a wide range of habitats, including open savanna, grasslands,

acacia woodland, and wooded bushlands. According to the IUCN Red List, eland is listed as of Least Concern (IUCN, 2016) with an estimated global population of 136,000 individuals (IUCN, 2016). Approximately 50% of this population lives in protected areas, 30% on private land, and the remaining 20% in unprotected areas (IUCN, 2016).

Across Southern Africa, eland distribution is shaped by a combination of environmental and human factors over time and space (Akanyang *et al.*, 2017; Patel *et al.*, 2019). Major environmental influences include rainfall, habitat type, water availability, land surface temperature, and vegetation greenness (NDVI), while human-related factors involve cultivation, roads, livestock, and settlements (Akanyang *et al.*, 2017). Seasonal rainfall patterns determine the availability of forage and water, necessitating shifts in range between wet and dry seasons (Watson & Owen-Smith, 2001). During the wet season, elands graze on grasses, whereas during the dry season, they switch to browsing when grasses become scarce (Watson & Owen-Smith, 2001). Variations in rainfall, NDVI, and land surface temperature serve as reliable indicators of habitat quality and forage availability, influencing eland spatial distribution patterns during periods of resource abundance or scarcity (Patel *et al.*, 2019). On the other hand, elands are sensitive to livestock grazing intensity and human-induced risk, hence suggesting that human-induced risk, such as human disturbance, is among the most significant factors influencing eland distribution (Akanyang *et al.*, 2017; Patel *et al.*, 2019).

Moreover, anthropogenic pressures, such as increasing human populations and the conversion of natural habitats into agricultural land, have increased competition for resources between humans and wildlife in protected areas (Snyder, 2016). In the Ukhahlamba-Drakensberg Park, South Africa, long-term eland population data indicate a stable population within the protected area but a decline in surrounding human settlements (Patel *et al.*, 2019). Similarly, in Botswana's Kalahari rangelands, elands have been observed avoiding grazed communal areas and concentrating in wildlife management areas far from human settlements, where human disturbance and livestock competition are minimal (Akanyang *et al.*, 2017). This has significant implications, as the ongoing

expansion of human activities threatens to cause habitat loss for elands.

Tanzania supports one of the largest eland populations in East Africa (Okello *et al.*, 2015), underscoring the importance of eland conservation. Despite the species' ecological significance and contribution to tourism, few studies have focused specifically on elands. Most existing research has focused on habitat use and feeding ecology within the Ngorongoro Conservation Area, revealing that elands primarily occupy grassland habitats and feed mainly on selected grass species, with limited browsing on woody plants (Mtega *et al.*, 2023). A more recent study by Massawe *et al.* (2025) used Species Distribution Models (SDMs) for 20 mammal species, including elands, and found that land surface temperature and maximum NDVI were the dominant factors influencing eland distribution in Nyerere National Park, Tanzania. However, there remains a limited understanding of how environmental and human-related factors interact to determine eland distribution patterns across other ecosystems in Tanzania, especially in recently upgraded protected areas.

Burigi Chato National Park (BCNP), formed by upgrading the Biharamulo, Burigi, and Kimisi (BBK) Game Reserves, provides a valuable opportunity to address this knowledge gap. Historically, the area experienced intense anthropogenic pressure, particularly during the early 1990s, when over 600,000 refugees from Rwanda and Burundi settled in Ngara District (Masalu, 2008). This sudden population surge increased demand for firewood, building materials, and food, driving extensive deforestation and heavy poaching that sharply reduced large herbivore populations, including the common eland (Jambiya *et al.*, 2007). In 1997, Jambiya *et al.* (2007) recorded 1,414 elands and 753 carcasses. Recent records from TAWIRI indicate the presence of about 250 elands (sightings = 15) across Burigi-Chato NP (formerly the Burigi-Biharamulo-Kimisi GR),

suggesting continued population decline of this species (TAWIRI, 2025, unpublished).

Increased wildlife protection following the gazettelement of Burigi-Chato National Park was expected to improve wildlife populations there. However, this has not been the case, raising questions about whether the continued decline may be driven by confounding environmental and human-related factors. Therefore, this study aims to examine how environmental and anthropogenic factors influence the spatial distribution patterns of elands in BCNP. Specifically, the study aims to; (i) assess how environmental variables: Land Surface Temperature, vegetation greenness (NDVI), and distance to water affect eland distribution; (ii) evaluate the influences of human activities: livestock keeping, cultivation, and settlements; and (iii) integrate these factors within a spatial modelling framework to identify suitable habitats and potential human-wildlife conflict zones within the park.

MATERIALS AND METHODS

Study area

The study was conducted in Burigi-Chato National Park (BCNP), located in north-western Tanzania, between latitudes 1.94°-2.5°S and longitudes 30.9°- 31.6°E (Figure 1A). The park lies at elevations ranging from 1,100m to 1,500m above sea level (AGL), and covers an area of 4,702 km². Administratively, BCNP is situated in the Muleba, Ngara, and Biharamulo Districts of Kagera, as well as in Chato District of Geita (Jambiya *et al.*, 2007). It is the third-largest national park in Tanzania,

after Ruaha and Serengeti National Parks (Tanzania National Parks, 2021). Burigi-Chato National Park was officially gazetted in 2019 following the upgrading of the former Burigi-Biharamulo-Kimisi (BBK) Game Reserves, which were originally established in the 1970s. The park extends from Bunkwe Bay of Lake Victoria in the east to the Kagera swamps in the west. Its landscape is characterized by a series of north-south-oriented ridges interspersed with broad plains, giving rise to a predominantly undulating to gently topographic landscape. The dominant soil type within the park is Haplic Ferralsols, which are generally low in fertility and influence vegetation distribution and productivity.

The BCNP contains several permanent water bodies that play a critical role in sustaining wildlife and ecological processes within the park. These include Lakes Burigi, Kisanga, Nyamalebe, and Ngoma, as well as significant portions of Lake Victoria. These aquatic systems provide essential water resources for both terrestrial and aquatic species, particularly during dry seasons, and support a diverse range of fish and other aquatic organisms. The park supports a wide variety of vegetation types reflecting its heterogeneous landscape and soil conditions. The major vegetation formations include forests, thickets, woodlands, shrublands, bushlands, grasslands, and swamps. This diversity of habitats contributes to the park's high biodiversity and ecological importance, supporting numerous wildlife species and maintaining ecosystem functions.

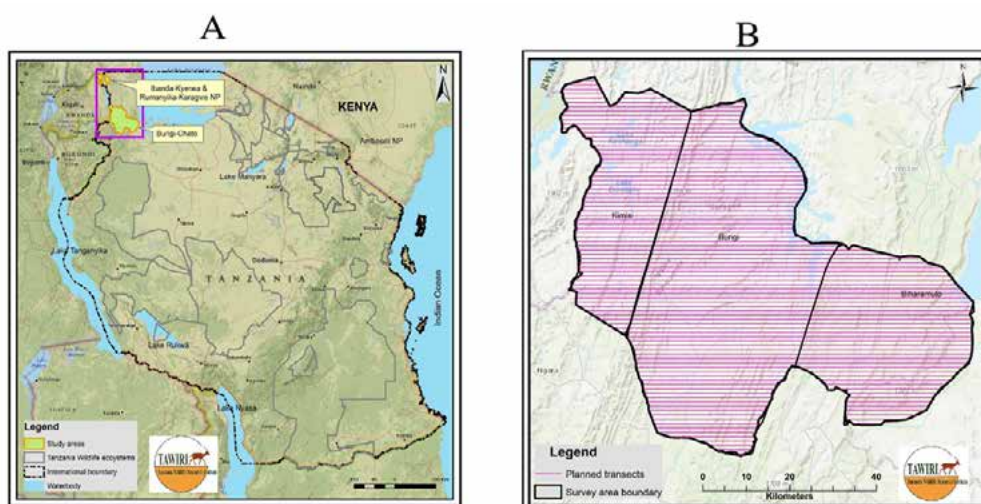


Figure 1: (A): Map showing the location of the study area in relation to other protected areas in Tanzania; (B): Design and layout of the planned transect spaced 1km during the total count census

Data collection

Eland records and environmental data

The study utilized data on eland GPS locations (easting and northing) collected from a Total Count (TC) aerial survey conducted by TAWIRI over the BCNP. The survey included 171 parallel flight transects spaced 1 km apart, totaling 1,001 km (Figure B). The transects were systematically flown in an east-west direction, with observers recording all eland sightings along each flight path. The elevation data for identified hotspot areas were extracted from a 30m resolution Digital Elevation Model (DEM) (<https://www.earthdata.nasa.gov/data/instruments/srtm>) (Wang *et al.*, 2011). The Normalized Difference Vegetation Index (NDVI) and rainfall were derived from Moderate Resolution Imaging Spectroradiometer (MODIS) data (<https://modis.gsfc.nasa.gov>), whilst the land surface temperature datasets were acquired from (<http://www.usgs.gov>).

Vegetation cover data

Vegetation cover for BCNP in 2025 was analyzed using Landsat 8 Operational Land Imager (OLI) and Thermal Infrared Sensor

(TIRS). All images were visually interpreted and processed to create a vegetation map for BCNP. The 30m resolution satellite images were freely sourced from the USGS Earth Explorer platform (<http://www.usgs.gov>). To reduce seasonal variability in vegetation cover, images were selected from the dry season, ensuring clear sky conditions and less than 10% cloud cover (Kija *et al.*, 2020). The satellite images were imported into ArcGIS version 10.8, georeferenced, and projected to Universal Transverse Mercator (UTM) zone 36S. Image preprocessing, including layer stacking, radiometric correction, and geometric correction, was applied to enhance image quality and reduce errors. False-color composite images were generated using band combinations 5-4-3 for TM and ETM+, and 6-5-4 for OLI, which are optimal for vegetation analysis (Wang *et al.*, 2015).

Data analysis

A zero-inflated negative binomial model was used to determine the influence of environmental and anthropogenic variables on the spatial distribution of eland. The predictor variables included NDVI, elevation, the proportion of land converted to incompatible uses such as agriculture and settlements,

distance to protected area boundaries, proximity to main road networks and rivers, and distance to human settlements. These variables were selected based on prior studies identifying them as important determinants of eland distribution (Lendelvo, 2018). Before running the model, fitting, and assessing multicollinearity among predictor variables, the Variance Inflation Factor (VIF) was used, and variables with VIF values exceeding 10 were excluded from the analysis (Bayman & Dexter, 2021).

Supervised image classification was performed using the maximum likelihood classification (MLC) algorithm to assign each pixel to the class with the highest probability of membership (Patil *et al.*, 2012). The training signatures were developed through visual interpretation and high-resolution reference imagery from

Google Earth (<https://earthengine.google.com>). Classification accuracy was assessed using 250 ground-truth points (GCPs) distributed across BCNP, and the results were adjusted accordingly. The final classified images were organized into standard LULC categories: cultivation, settlements, bushland, grassland, woodland, and wetland (Kija *et al.*, 2020; Msofe *et al.*, 2019) (Table 1). Chi-square goodness-of-fit tests were subsequently employed to determine differences in eland habitat preferences across different habitats within the BCNP.

All statistical analyses were conducted in R version 3.4.1, while spatial data visualization and mapping were carried out using ArcGIS version 10.8 and QGIS version 3.6 software.

Table 1: Land use and land cover (LULC) types adapted from a classification system by Kija *et al.* (2020) and Msofe *et al.* (2019) in the BCNP by 2025.

LULC types	LULC Description
Cultivation	Land actively used to grow crops, seasonally and permanently.
Scrubland	Predominantly comprised of multi-stemmed plants from a single root base, woody cover < 50% canopy cover of woody plants of < 5 m height.
Woodland	> 60% canopy cover of woody plants of ≥ 5 m height.
Grassland	< 10% cover of sparse woody plants, dominated by continuous herbaceous cover.
Settlement	Urban and rural settlements (including houses, roads, infrastructure).
Wetlands	Land consisting of marshes or swamps; saturated land.

RESULTS

Influences of environmental and ecological variables

Results indicate that environmental and ecological variables strongly influence eland distribution. Land surface temperature had a positive and highly significant effect ($\beta = 1.624$; $p = 0.0006$), suggesting that eland prefer relatively warmer areas, possibly due to associated habitat types or favorable microclimatic conditions. Normalized Difference Vegetation

Index (NDVI) exhibited a strong, positive, and highly significant relationship ($\beta = 95.254$; $p = 0.0003$), highlighting the importance of vegetation greenness and forage availability in determining eland habitat use. Additionally, elevation and rainfall were not significant predictors ($\beta = -0.37$, $p = 0.232$) and ($\beta = -6.21481$, $p = 0.1481$), respectively. The effect of rainfall is probably captured through NDVI, and the eland distribution in BCNP is not strongly structured along elevational gradients (Table 2).

Influences of anthropogenic variables

Findings show that the distance to livestock grazing areas had a strong and highly significant effect on eland distribution ($\beta = -5.254$; $p = 0.0004$), suggesting that eland occurrence decreases with increasing distance from livestock areas and indicating spatial overlap between eland and livestock. This pattern may reflect shared use of productive grazing areas along the park's borders. Moreover, distance to cultivation indicated a significant negative effect ($\beta = -0.134$; $p = 0.0152$), suggesting that eland are more likely

to occur closer to cultivated areas. This could be associated with the presence of palatable crops or higher-quality forage near agricultural fields. In contrast, distance to settlements had a marginally non-significant negative effect ($\beta = -0.14$; $p = 0.0556$), suggesting a weak negative relationship between eland and settlements. Distance to roads had a significant positive effect ($\beta = -0.22$; $p = 0.0239$), indicating that elands are more likely to be found farther from roads, likely due to avoidance of disturbance, vehicle traffic, and associated human activities (Table 2).

Table 2: Environmental and anthropogenic drivers of spatial-temporal distribution of eland in BCNP.

Responsible variable	Estimate	SE	z value	Pr ($> z $)
(Intercept)	-92.04	4.81	-0.43	0.6654
Livestock	-5.25	3.38	1.61	0.0004***
Distance from cultivation	-0.134	0.14	-2.43	0.0152*
NDVI	95.25	8.38	3.61	0.0003***
Distance from settlements	-0.14	0.07	-1.91	0.0556
Rainfall	-0.37	0.32	-1.19	0.232
Elevation	-6.21	1.29	-0.42	0.1481
Land surface temperature	1.62	0.36	4.53	0.0006***
Distance from roads	-0.22	0.09	2.26	0.0239*

Land Use and Land Cover (LULC) analysis

The land-use and land-cover analysis reveals that BCNP is predominantly covered by natural vegetation, with limited anthropogenic land use (Table 3). Woodland was the dominant habitat type, covering approximately 2,821.2 km² (60.2%) of the park area. Grassland accounted for 1,410.6 km² (30%), while scrubland occupied 187 km² (0.04%) of the total park area, and water bodies together constituted less than

0.05% of the area (Figure 2). Anthropogenic land uses, covering only 47.02 km² (0.01% of the total area), are mostly concentrated along the park boundary. Spatially, woodland habitats are widely distributed across the central and southern part of the park, while grasslands occur in patches in the northern section of the BCNP, frequently associated with rough terrain. Shrublands are primarily located in zones between woodland and grassland habitats.

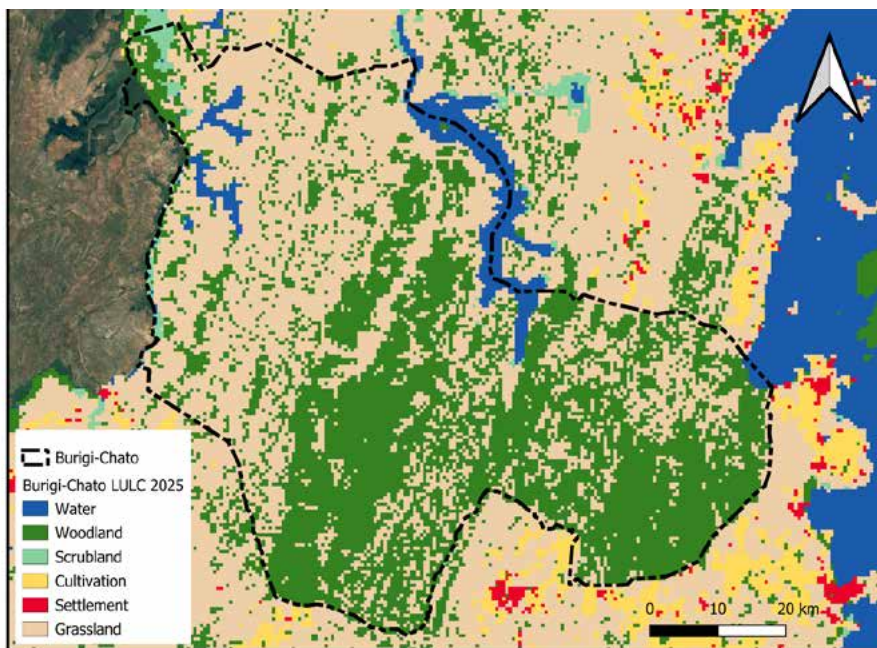


Figure 2: Map of vegetation cover in Burigi Chato NP, satellite imagery, 2025.

Habitat type on eland distribution in BCNP

A total of 153 common eland sightings were recorded, with 48.57% occurring in grassland habitats and 47.93% in woodland areas, while only nine sightings were observed in scrubland. According to Mtega *et al.* (2023), eland use diverse habitats, predominantly occupying grasslands during the wet season

and shifting to woodland habitats in the dry season when forage is scarce. However, A chi-square goodness-of-fit test revealed a significant difference in the overall distribution of sightings among habitat types ($\chi^2 = 272.87$, $df = 3$, $p < 0.05$), with grassland being the most frequently used habitat and wetlands being the least represented (Figure 3).

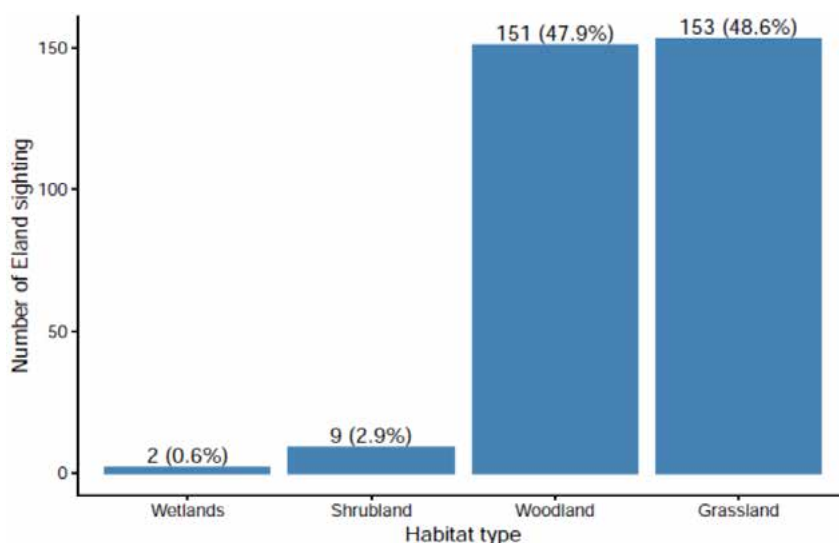


Figure 3: Number of common eland sightings in different vegetation habitats in BCNP

DISCUSSION

Generally, findings indicate that the spatial distribution of eland in BCNP is driven by interactions among environmental and ecological variables and human-modified landscapes. Forage availability and proximity to anthropogenic land uses emerged as the primary determinants of eland habitat use, while avoidance of roads indicates sensitivity to direct human disturbance, pinpointing the need to integrate both environmental factors and human land-use patterns into eland conservation and park general management plans.

Furthermore, environmental and ecological variables strongly influenced eland distribution. Findings indicate that land surface temperature had a significant positive effect, suggesting that eland preferentially use relatively warmer areas, possibly due to associated vegetation. Warmer areas may also reduce increased thermoregulatory costs, particularly in open areas (Yager *et al.*, 2024). The NDVI showed the strongest positive relationship with eland occurrence, highlighting the importance of vegetation forage availability (Borowik *et al.*, 2013). As a large-bodied mixed feeder, eland require abundant, high-quality forage, and areas with high NDVI provide greater biomass and nutritional value (Weeber, 2018). In contrast, elevation and rainfall were not significant predictors, probably due to limited variation across BCNP, indicating that local-scale vegetation and microclimatic conditions are more influential than broad climatic gradients (Kidane, 2022).

Also, human activities play an important role in shaping eland distribution. The strong association between eland occurrence and proximity to livestock grazing areas suggests spatial overlap, driven by shared use of productive grazing areas bordering the BCNP boundary with rich forage. While areas close to the park boundary enhance forage availability, this overlap increases the potential for competition and disease transmission.

Eland also occurred near cultivated areas, possibly attracted by palatable crops or early-successional vegetation, as reported by Kariuki *et al.* (2021), though this pattern increases the risk of crop raiding and negative interactions with local communities. Distance to settlements showed a weak, marginal effect, indicating that settlements have less influence on eland distribution than grazing areas and cultivation. In contrast, eland avoided roads, possibly due to vehicle disturbance and increased human activity.

Land-use/land-cover analysis shows that BCNP is dominated by woodland vegetation, covering over 60.2% of the park. Despite this, eland disproportionately used grassland habitats, which occupy a small fraction of the area but supported nearly half of all sightings. This indicates a strong habitat preference for grasslands, driven by higher forage quality and accessibility. Seasonal shifts in habitat use further reflect eland's adaptive foraging strategy, with greater use of grasslands during the wet season and increased reliance on woodlands in the dry season for browse and shade.

Management and conservation implications

Eland distribution in BCNP is influenced by a combination of ecological suitability and human land-use patterns. Maintaining heterogeneous landscapes that provide high-quality forage across the park is therefore essential. Such habitat management not only supports eland populations but also helps to reduce potential conflicts arising from livestock grazing and agricultural activities. In addition, minimizing disturbance from road vehicles and carefully planning infrastructure development are critical for conserving suitable habitats. These measures will help maintain ecological connectivity, which is necessary for the movement and long-term persistence of eland populations within the park.

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LARGE-MEDIUM SPECIES COMPOSITION, DIVERSITY, AND ABUNDANCE IN THE ARUSHA NATIONAL PARK, NORTHERN TANZANIA

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ABSTRACT

Mammals are crucial components of global biodiversity. Therefore, understanding the diversity of mammalian species and their habitat associations is a baseline for developing conservation plans. This study investigated mammalian species composition, distribution, diversity, and abundance in Arusha National Park (ANAPA) in June 2025. Both direct and indirect observation methods were employed, using ground vehicle and walking transect surveys, as well as distance sampling techniques for data collection. The results from the direct observation method revealed that a total of 1,079 individual mammals, representing 16 species, eight families, and six orders, were recorded in the area. Only nine additional species were recorded through the indirect observation method. The order Artiodactyla contributed the highest number of species ($n = 8$). Of the 16 species recorded, only the cane rat (*Thryonomys swinderianus*) and the African savanna hare (*Lepus victoriae*) were small mammals; the remaining species ranged from medium to large mammals. Although the African buffalo (*Syncerus caffer*; $n = 414$; 38.37%) was the most abundant species compared to others across different habitat types, the differences were not statistically significant ($P = 0.3785$). Moreover, woodland habitat exhibited the highest mammalian diversity ($H' = 1.684$) compared to other habitats, and the difference was statistically significant ($P = 0.04695$), likely because some species are highly restricted habitat specialists in the area. Additionally, the results show that mammalian species diversity decreases with increasing altitude in the area. High species diversity and abundance were recorded at low altitudes (1,400–1,767 m asl) ($n = 1,045$; $H' = 1.877$), whereas lower diversity was observed at higher altitudes (2,135–2,500 m asl) ($n = 10$; $H' = 0.950$), and the difference was statistically significant ($P = 0.03638$). These findings provide valuable insights for enhancing tourism potential and underscore the importance of continued conservation efforts in the area.

Keywords: Biodiversity, Conservation, Habitats, Location, and Mammals

INTRODUCTION

Mammals are a diverse group of vertebrates that range in size from the largest whales to the smallest rodents, such as mice. Small mammals form a significant proportion, and among these, rodents comprise approximately 43% (Moges *et al.*, 2016). They are a highly versatile group due to their diversity in species, morphologies, ecologies, and behaviors, with approximately 5,487 species globally (Tefera *et al.*, 2025). On

the African continent alone, there are more than 1150 distinct mammalian species, with 360 of those species found in eastern Africa (Tefera *et al.*, 2025). Mammals are integral to terrestrial ecosystems, serving essential roles that enhance habitat structures, shape trophic dynamics, and maintain various ecological processes (Sharma, 2024). Their influence extends to promoting biodiversity and maintaining ecosystem functions, primarily through mechanisms such as herbivory, seed dispersal, and nutrient

cycling (Canale *et al.*, 2012; Sharma, 2024). However, significant anthropogenic activities, such as habitat destruction, overexploitation, and climate change, have severely impacted the natural distribution, density, and compositional dynamics of large mammal populations, ultimately threatening biodiversity conservation efforts globally (Crees *et al.*, 2016; Rios *et al.*, 2021).

In sub-Saharan Africa, protected areas, including national parks, are experiencing increasing anthropogenic activities, specifically livestock encroachment and grazing pressures, which threaten habitat quality and biodiversity conservation efforts (Ogutu *et al.*, 2019; Legese *et al.*, 2019). These areas provide essential resources for wildlife but are increasingly threatened by human encroachment and infrastructural development, which can compromise the ecological integrity of the habitats upon which these animals depend (Saito & Koike, 2013). For instance, the legacy of land-use changes and community interactions can significantly affect large mammal populations, leading to shifts in community structure and biodiversity (Rios *et al.*, 2021).

Arusha National Park (ANAPA) in Tanzania exemplifies these dynamics, serving as a biodiversity hotspot that offers various habitats, including montane forests and grasslands, which are crucial for numerous large mammal species. However, like other globally threatened protected areas, the surrounding agricultural landscapes exert considerable pressure on the park, affecting the diversity, abundance, and distribution of mammalian species (Bellón *et al.*, 2022). Given this threat, it is therefore crucial that rigorous monitoring of large mammal populations within ANAPA is essential for evaluating the effectiveness of conservation strategies and providing data to anticipate ecological shifts resulting from both environmental and anthropogenic changes (Andermann *et al.*, 2020).

Despite ANAPA's rich biodiversity, there is a lack of rigorous information about faunal diversity due to inadequate resources, dense vegetation, poor visibility, and an inaccessible and mountainous landscape. The majority of assessments of ANAPA fauna are confined to forest areas, which necessitate the use of indirect sign surveys to confirm the presence and abundance of mammalian species, such as African elephants, Buffalo, giraffes, and zebras. The inventory of the mammalian assemblage is essential in understanding the distribution of species in an area. In ANAPA, for example, no recent studies have been conducted regarding the composition, distribution, diversity, and abundance of mammal species in the area. Hence, the findings from this study will provide crucial information for informed decision-making and planning processes in large mammal conservation at ANAPA. Therefore, the present study aimed to document the mammalian species composition, diversity, and abundance in the area.

Furthermore, it was hypothesized that mammalian species diversity would differ significantly across different habitats and levels of attitudes in the area. This can be attributed to the fact that, following natural selection and evolutionary adaptation, some species have become highly specialized habitat specialists (Sharma, 2024). Therefore, findings from this study provide valuable insights for promoting tourism potential and highlighting the importance of ongoing conservation efforts in the area.

MATERIALS AND METHODOLOGY

Study area

This study was conducted in Arusha National Park, located in northern Tanzania within the Arusha Region. The park extends from latitudes 03°12' to 03°18' South and longitudes 36°45' to 36°56' East (Map. 1), approximately 32 km northeast of Arusha town, and covers an area of 552 km² (Scoon, 2018). Elevations range from

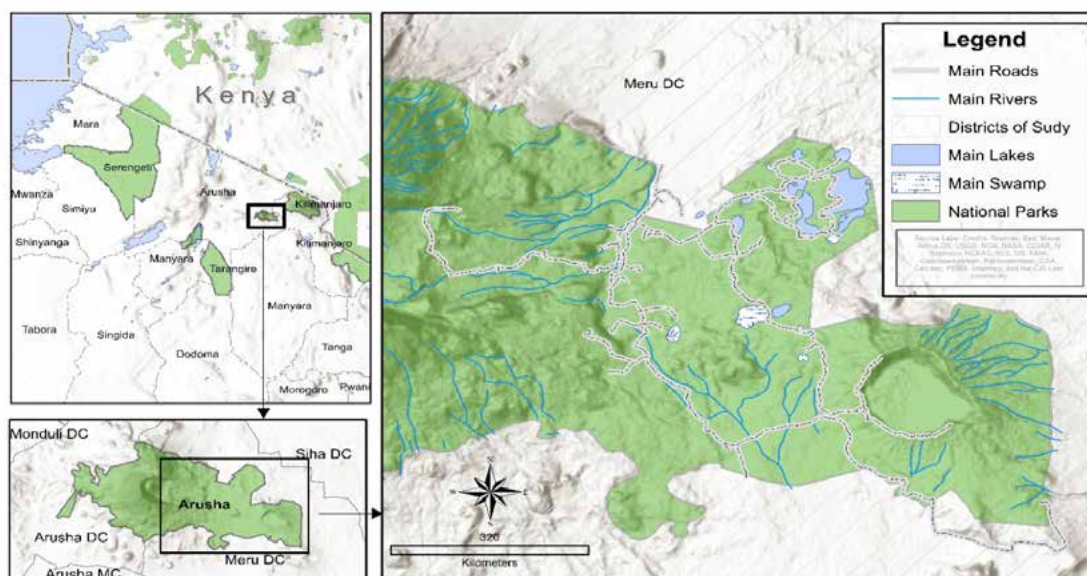
1,400 m to 4,565 m, encompassing a diverse mosaic of habitats, including lakes, swamps, grasslands, shrublands, forests, heathlands, and rocky outcrops (Lee *et al.*, 2023).

Arusha National Park experiences a bimodal rainfall pattern, characterized by short rains from October to January and long rains from February to May, with a dry season from June to September (Prins & Loth, 1988; Lee *et al.*, 2023). Annual rainfall varies considerably, ranging from 600 mm to 2,400 mm, with an average of around 1,100 mm near the Momella Lakes, an area known for its high giraffe densities (Martinoli *et al.*, 2006; Lee *et al.*, 2023). Mean maximum and minimum temperatures are 25.4°C and 12.8°C, respectively (Kahana *et al.*, 2015; Lee *et al.*, 2023).

Vegetation varies markedly across the park. Montane forests dominate Mount Meru and the Ngurdoto Crater, with prominent tree species such as *Podocarpus*, *Xymalous*, and *Ilex* spp. On stony hillsides, *Juniperus capensis* and *Dodonia viscosa* are common. Bushlands and thickets are primarily composed of *Diospyros abyssinica* and *Olea hochstetteri*. Grasslands and forest glades support species such as *Cynodon dactylon*, *Pennisetum clandestinum*, *Eleusine jaegerii*, and *Cymbopogon afronadus*. The montane forests of Mount Meru function as critical catchments, capturing moisture and supplying water to the surrounding densely populated and intensively farmed areas.

The park hosts a rich faunal diversity, with over 40 mammal species, approximately 500 bird species, 24 reptile species, and 10 amphibian species documented (Maleko *et al.*, 2012). Large herbivores include Buffalo (*Syncerus caffer*), warthog (*Phacochoerus africanus*), hippopotamus (*Hippopotamus amphibius*), plains zebra (*Equus burchellii*), and waterbuck (*Kobus ellipsiprymnus defassa*). Common browsers include the African elephant (*Loxodonta africana*), the giraffe (*Giraffa tippelskirchi*), the dik-dik (*Madoqua kirkii*), and the small red forest duiker (*Cephalophus natalensis*). Predators are scarce, with only small populations of leopards (*Panthera pardus*), spotted hyenas (*Crocuta crocuta*), and isolated cheetahs (*Acinonyx jubatus*); lions (*Panthera leo*) are absent (Maleko *et al.*, 2012). Among the most frequently observed mammals are Buffalo, bushbuck, giraffe, and primates, such as olive baboons, blue monkeys, and black-and-white colobus monkeys (Nyangusi, 2008).

The areas surrounding ANAPA are predominantly agricultural, with local communities engaging in both crop farming and livestock rearing (Nyangusi, 2008). Permanent crops, notably bananas and coffee, are supported by small-scale irrigation systems, while maize, beans, and vegetables constitute important seasonal cash crops. Livestock keeping is extensive, involving cattle, goats, sheep, donkeys, and poultry. The park's western margins face particular pressure from intensive grazing and land scarcity, which have contributed to increased illegal grazing and encroachment into protected forest reserves (Nyangusi, 2008).



Map 4: Location of Arusha National Park

Data collection

Data on large mammals in ANAPA were collected using both primary field methods and secondary sources. Primary data collection combined qualitative and quantitative approaches, including consultations with key informants, systematic ground transects (by vehicle and on foot), and opportunistic observations. These methods were adapted from established wildlife survey protocols (Norton-Griffiths, 1978). Secondary data comprised historical monitoring reports, ecological records, and archived park datasets, which supported planning and contextual interpretation.

Consultative meeting with key informants

Before fieldwork, a meeting was held with ANAPA personnel, including the Park Warden, ecologists, protection officers, and experienced rangers, to define survey objectives, identify priority sampling areas, and incorporate expert knowledge on wildlife distribution. These discussions also provided critical spatial data, such as GIS layers and past survey records, which informed the placement of transects.

Ground transects

Ground Vehicle transects were conducted along selected administrative and patrol roads to record direct sightings of large mammals. Surveys maintained a consistent speed of 20 km/h, with observations recorded within 500 m on either side of the vehicle. The data included species identity, group size, sex, age composition, GPS coordinates, and habitat type. Additional behavioral notes captured activities such as feeding, resting, or vigilance. Other records were obtained through opportunistic sightings made outside formal transects while traversing the park and walking transects. We established walking transects parallel to the road transects, spaced every 5 kilometers. These parallel transects aimed to document animal species both directly and indirectly by looking for signs such as faeces, pellets, footprints, feeding marks, breeding indicators, and other relevant evidence. These incidental observations helped broaden species inventories, capturing data on less frequently encountered or more elusive taxa.

Data Analysis

The collected data were organised and entered into an Excel worksheet for the analysis. All statistical analyses were performed using the PAST 4.03 software. PAST 4.03 was used to analyze the Diversity indices, including species richness (the number of species present) and abundance (the number of individuals), species evenness, the Simpson similarity index, and the Shannon-Wiener index. The Shannon-Wiener index (H') was used to quantify species diversity; this index assigns higher values to communities with a greater number of species and a more even distribution of individuals among them. A one-way ANOVA test of independence was used to examine differences in diversity and the abundance of mammal species among habitats, locations, and between altitudinal levels. A p-value of less than 0.05 was considered statistically significant for differences in species diversity and composition. The relative abundance index of species (RAI) was calculated by dividing the number of records of each species by the total number of records of all species in an Excel sheet. Moreover, the conservation status of the mammals identified from ANAPA was described according to the IUCN Red List standards. Indirect observation data supported

additional information on the presence of mammal species.

RESULTS

Mammalian Species Composition and Abundance

A total of 1,079 individual of large-medium sized mammals, representing 16 species, eight families, and six orders, were recorded during the study. The order Artiodactyla contributed the most species ($n = 8$). Out of the 16 species recorded, only the cane rat and the African savanna hare were small mammals; the remaining species ranged from medium to large mammals (Table 1). Moreover, Buffalo ($n = 414$; 38.37%) were the most abundant species, followed by olive baboon ($n = 159$; 14.74%) and black-and-white colobus ($n = 111$; 10.29%). The least abundant species were the vervet monkey, African savanna hare, and cane rat, each represented by a single individual (0.09%; Table 1).

Based on the IUCN Red List, 12.5% of the species recorded are endangered, 12.5% are near threatened, and 75% are of least concern (Table 2).

Table 1: Composition, Abundance, and Relative Abundance (RA) of observed Mammalian species.

SN	Common name	Scientific name	Order	Family	IUCN Status	Field count	RA (%)
1	African Buffalo	<i>Syncerus caffer</i>	Artiodactyla	Bovidae	NT	414	38.37
2	Olive baboon	<i>Papio anubis</i>	Primates	Cercopithecidae	LC	159	14.74
3	Black and White Colobus	<i>Colobus guereza</i>	Primates	Cercopithecidae	LC	111	10.29
4	Warthog	<i>Phacochoerus africanus</i>	Artiodactyla	Suidae	LC	107	9.92
5	Masai giraffe	<i>Giraffa tippelskirchi</i>	Artiodactyla	Giraffidae	EN	83	7.69
6	Blue monkey	<i>Cercopithecus mitis</i>	Primates	Cercopithecidae	LC	82	7.60
7	Plains zebra	<i>Equus quagga</i>	Perissodactyla	Equidae	NT	67	6.21
8	African elephant	<i>Loxodonta africana</i>	Proboscidae	Elephantidae	EN	17	1.58

9	Bushbuck	<i>Tragelaphus scriptus</i>	Artiodactyla	Bovidae	LC	15	1.39
10	Natal red duiker	<i>Cephalophus natalensis</i>	Artiodactyla	Bovidae	LC	12	1.11
11	Common waterbuck	<i>Kobus ellipsiprymnus</i>	Artiodactyla	Bovidae	LC	3	0.28
12	Dik-dik	<i>Madoqua kirkii</i>	Artiodactyla	Bovidae	LC	3	0.28
13	Suni	<i>Neotragus moschatus</i>	Artiodactyla	Bovidae	LC	3	0.28
14	Vervet monkey	<i>Chlorocebus pygerythrus</i>	Primates	Cercopithecidae	LC	1	0.09
15	African savannah hare	<i>Lepus victoriae</i>	Lagomorpha	Leporidae	LC	1	0.09
16	Cane rat	<i>Thryonomys swinderianus</i>	Rodentia	Thryomyidae	LC	1	0.09
	Total					1079	100

Note: EN= Endangered, NT= Near threatened, LC= Least concern, RA= Relative abundance

Table 2: IUCN Red List of the observed species

IUCN Status	Number of Species	Percentage (%)
Endangered (EN)	2	12.5
Near Threatened (NT)	2	12.5
Least Concern (LC)	12	75
Total	16	100

In addition to the previously observed species, nine other mammal species: leopard, cheetah, porcupine, armadillo, striped mongoose, black-backed jackal, hippopotamus, spotted hyena, and common genet, were detected indirectly through footprints, dung/scat, or

burrows, bringing the total number of observed species to 25. Among these, 23 species were identified through indirect observations, with fourteen species also confirmed through direct observation (Table 3).

Table 3. Number of signs and percentage occurrence of observed mammal species

S. N	Common name	No. of signs encountered	Percentage
1	African Buffalo	119	18.71
2	African elephant	102	16.04
3	Masai Giraffe	71	11.16
4	Olive baboon	59	9.28
5	Natal red duiker	50	7.86
6	Common waterbuck	46	7.23
7	Bushbuck	44	6.92
8	Plains zebra	41	6.45
9	Dik-dik	18	2.83
10	Warthog	16	2.52

11	Aardvark	15	2.36
12	Common genet	13	2.04
13	Leopard	9	1.42
14	Spotted hyena	9	1.42
15	Hippopotamus	5	0.79
16	Suni	5	0.79
17	African savanna hare	3	0.47
18	Crested porcupine	3	0.47
19	Black backed jackal	2	0.31
20	Cheetah	2	0.31
21	Black-and-white Colobus	2	0.31
22	Banded mongoose	1	0.16
23	Blue monkey	1	0.16
	Grand Total	636	100

Distribution and abundance of mammals observed in different habitat types

Despite the differences in mammalian species composition across various habitat types, these differences were not statistically significant ($F(15, 96) = 1.002$, $P = 0.3785$). For instance, the Olive baboon ($n = 159$; 14.74%), warthog ($n = 107$; 9.92%), bushbuck ($n = 15$; 1.39%), and giraffe ($n = 83$; 7.69%) were the most

widely distributed species across most habitats in the area. Other species that were broadly distributed included the red duiker and black-and-white colobus, which were observed in more than three habitats. Although the common waterbuck ($n = 3$; 0.28%), suni ($n = 3$; 0.28%), African savanna hare ($n = 1$; 0.09%), and Cane rat ($n = 1$; 0.09%) were the least distributed species in the area (Table 4).

Table 4: Abundance of large-medium sized mammalian species in different habitats

Species	BGL	BL	FRT	GL	WET	WGL	WD	Total	RA
African Buffalo	15	0	0	395	0	4	0	414	38.37
African elephant	0	0	3	0	0	0	14	17	1.58
Giraffe	35	0	0	27	0	14	7	83	7.69
Plains zebra	13	0	0	54	0	0	0	67	6.21
Common Waterbuck	0	2	0	1	0	0	0	3	0.28
Bushbuck	0	3	8	0	2	1	1	15	1.39
Natal Red duiker	0	4	6	0	0	0	2	12	1.11
Olive baboon	19	0	83	26	21	3	7	159	14.74
Black and White Colobus	0	12	59	0	0	0	40	111	10.29
Blue monkey	0	0	69	0	0	0	13	82	7.6
Vervet monkey	1	0	0	0	0	0	0	1	0.09
Warthog	17	1	0	72	2	11	4	107	9.92
Dik-dik	0	3	0	0	0	0	0	3	0.28
Suni	0	0	1	0	0	0	2	3	0.28

Species	BGL	BL	FRT	GL	WET	WGL	WD	Total	RA
African savanna hare	1	0	0	0	0	0	0	1	0.09
Cane rat	0	0	1	0	0	0	0	1	0.09
Total	101	25	230	575	25	33	90	1079	100
Percentage %	9.4	2.3	21.3	53.3	2.3	3.1	8.3	100	

Note: BGL= Bushed grassland, BL= Bushland, FRT= Forest, GL= Grassland, WET= Wetland, WGL= Wooded grassland, WD= Woodland, RA= Relative Abundance

Distribution and Diversity of mammals at different habitat types

The highest diversity of mammalian species was observed in the woodland habitat ($n = 9$; $H' = 1.684$), followed by bush grassland ($n = 7$; $H' = 1.620$) and bushland ($n = 6$; $H' = 1.485$) and lowest in the grassland habitat ($n = 3$; $H' = 1.035$) and wetlands ($n = 3$; $H' = 0.5506$) and their differences were statistically significant ($F(6,105) = 1.617$, $P = 0.04695$). The wooded grassland had the most even distribution ($H' = 1.31$), while the grasslands had the least evenness ($H' = 0.4691$) (Table 5).

Table 5: Diversity indices of mammalian species observed in the seven habitats

Habitat	Taxa S	Individuals	Dominance D	Simpson 1-D	Shannon H	Evenness $e^{H/S}$
Bushed grassland	7	101	0.2226	0.7774	1.62	0.7218
Bushland	6	25	0.2928	0.7072	1.485	0.736
Forest	8	230	0.2881	0.7119	1.394	0.5038
Grassland	6	575	0.5007	0.4993	1.035	0.4691
Wetland/ lakes	3	25	0.7184	0.2816	0.5506	0.5781
Wooded grassland	5	33	0.315	0.685	1.31	0.741
Woodland	9	90	0.2578	0.7422	1.684	0.5987

Distribution and diversity of mammals at different locations

Additionally, these mammal species were distributed across four locations within the park: Maziwani, Ngarenanyuki, Ngurdoto, and Miriakamba. In those locations, the highest Shannon-Wiener index (H') was recorded in the Maziwani ($n = 12$; $H' = 1.877$), followed by Ngarenanyuki ($n = 13$; $H' = 1.725$), Ngurdoto ($n = 9$; $H' = 1.474$), and lastly Miriakamba ($n = 4$; $H' = 1.327$; Table 5) (table 6), however, their differences were not statistically significant ($F(3,60) = 1.725$, $P = 0.1487$).

Table 6: Diversity indices of species in different locations

	Ngarenanyuki	Ngurdoto	Maziwani	Miriakamba
Taxa_S	13	9	12	4
Individuals	462	238	340	39
Dominance_D	0.2629	0.2649	0.1931	0.2794
Simpson_1-D	0.7371	0.7351	0.8069	0.7206
Shannon_H	1.725	1.474	1.877	1.327
Evenness_ $e^{H/S}$	0.4318	0.4851	0.5446	0.9421

Altitudinal distribution of mammals

The abundance and diversity of species were recorded at altitudes ranging from 1,400 m to 2,500 m above sea level. These altitudes were divided into three groups: low altitude (1,400–1,767 m), moderate altitude (1,768–2,134 m), and high altitude (2,135–2,500 m).

The highest number of species ($n = 16$; $H' = 1.877$) was found at low altitude, while the fewest species ($n = 3$; $H' = 0.950$), were found at high altitude (Table 6), and their differences were statistically significant ($F(2,20) = 1.067$, $P = 0.03638$). The results indicate that the mammalian species diversity decreases as the altitude increases in the area (Table 7).

Table 7: Diversity indices of mammalian species observed at different altitude levels

Altitude	Low altitude	Medium altitude	High altitude
Taxa_S	16	4	3
Individuals	1045	24	10
Dominance_D	0.2152	0.3646	0.44
Simpson_1-D	0.7848	0.6354	0.56
Shannon_H	1.877	1.137	0.9503
Evenness_e^H/S	0.4085	0.7793	0.8621

DISCUSSION

Mammalian Species Composition, Abundance and Distribution

Although ANAPA is deemed to harbor iconic species of large mammals that are endangered in the country, few studies have been conducted on the species composition, diversity, and abundance of mammals in the area. Understanding the diversity, abundance, and habitat preferences of the mammalian species is essential for determining their status and suggesting effective conservation actions (Derebe *et al.*, 2023). In the present study, a total of 16 species, representing eight families and six orders, were recorded through direct observation in the study area. These findings are similar to those of the study by Worku *et al.* (2025), which reported the discovery of 16 species of large mammals in Nensebo Forest, southern Ethiopia, that unlike savanna areas, forest areas are species specific and thus they harbor few large-medium species. According to Worku *et al.* (2025), other studies in Tiski waterfall and Dati Wole National Park recorded 10 and 28 mammalian species, respectively, an

area of similar characteristics to Arusha park. However, this study focused only in Eastern part of the park which might limit the number of species recorded. A need to have a complete study for the entire park with additional night transects surveys to detect nocturnal species is a research gap that need an attention.

The order Artiodactyla (8 species) was the most abundant mammal group observed in the current study area. This finding aligns with the study by Tefera *et al.* (2025) in Dhati Wole National Park, which reported that, with the largest diversity of ungulates on Earth, the African savanna environment has supported multispecies animal production systems for thousands of years.

However, recently, both natural and anthropogenic factors have significantly threatened the survival of mammals by causing declines in mammal species populations, including habitat loss and fragmentation, overexploitation, global climate change, and other anthropogenic pressures that threaten mammals worldwide (Tefera *et al.*, 2025).

Despite the 16-mammalian species observed in ANAPA through direct observation methods, nine other mammal species account to 25 mammalian species, including leopard, cheetah, porcupine, armadillo, striped mongoose, black-backed jackal, hippopotamus, spotted hyena, and common genet, were detected indirectly through either their signs of footprints, dung/scat, or burrows, but their abundance was not estimated. Presence of some carnivore species such as cheetah that were recorded through signs may need further investigation to confirm their presence.

Although the abundance of mammals varied from one habitat to another in the present study, the differences were not statistically significant. The variation in the abundance of mammal species among different habitats is related to the quality of the habitat and the species' preferences (Kasso & Bekele, 2017). However, in our study, it was observed that some mammals (i.e., the Olive baboon, warthog, bushbuck, and giraffe) were more widely distributed across all habitats compared to others. While other mammals, such as the common waterbuck, suni, African savanna hare, and Cane rat, were the least distributed species in the area. Species abundances can be influenced by several factors, such as biotic factors (food quality and availability, number of predators/preys, competition), abiotic factors (soil, water, and climate), species traits, and habitat quality (Steinbeiser *et al.*, 2019). These findings corroborate those of the previous study, which reported that the quality of food and its availability were essential factors in determining the presence of the mammalian species across different habitats in an area (Kasso & Bekele, 2017).

Mammalian Species Distribution and Diversity at different habitats and elevations

Previous studies have found that mammals can colonize a variety of habitat types due to their diverse morphological, physiological, and behavioral adaptations (Derebe *et al.*, 2023) and serve as ecological regulators in

natural habitats (Moges *et al.*, 2016). In the present study, mammals were not found to be uniformly distributed across all habitat types, and their diversity varied from habitat to habitat. The highest diversity of mammalian species was observed in the woodland habitat compared to other habitats present in the area. This is attributed to the presence of sufficient vegetation cover and resource availability for the dependent mammal species. These findings also support those of the study by Moges *et al.* (2016), who reported that the nature and density of vegetation determine the abundance and distribution of mammals. Moreover, mammal density in a given area correlates with the availability of food resources (Moges *et al.* 2016). Additionally, habitats with high vegetation coverage, compared to other habitats, would provide ample food, cover, and space to attract large numbers of mammalian species (Derebe *et al.*, 2023). According to Derebe *et al.* (2023), the presence of water, food, and cover is often associated with the distribution and habitat preferences of animals. However, one of the primary causes of biodiversity loss is believed to be the conversion of natural habitats into agricultural land use (Derebe *et al.*, 2023).

The distribution patterns along elevation gradients have been widely studied among animal taxa, including mammals (McCain, 2007). In those studies, the elevation gradients hold enormous potential for understanding general properties of an area's biodiversity (McCain, 2007). This pattern of decreasing or variable mammalian species diversity with increasing altitude is supported by Chen *et al.* (2022), who report changes in species richness along gradients similar to those observed in this study. The decreasing mammalian species diversity pattern in ANAPA is likely a result of the unique climate and vegetation habitats on Mount Meru, particularly at altitudes of 1,768–2500 m (Chen *et al.*, 2022). These results also concur with the previous one, which reported that the mammalian species diversity and composition in particular areas are influenced by their habitat, altitudes, and other

environmental factors such as temperature and rainfall

CONCLUSION AND RECOMMENDATIONS

The current study provided update information on the presence and abundance of mammals in ANAPA, northern Tanzania. ANAPA is unique in that it is home to animals of great conservation value, including large mammals such as African elephants and Buffalo, which together underscore the importance of the area for biodiversity conservation. Twenty-five mammalian species were identified in the study area. The order Artiodactyla had the highest number of species compared to the others. Mammal species distribution and abundance vary across habitats due to differences in vegetation types and resource availability. Woodland had the highest number of species in the area. Variations influenced the habitat selection of the mammalian species in the quality and abundance of forage. The distribution and utilisation of different vegetation communities by mammal species could be explained in terms of altitudinal changes. Therefore, we recommend that effective conservation and management strategies are necessary to ensure the sustainable conservation of mammals in the ANAPA. Again, proper conservation and mitigation measures should be implemented, as conflicts between humans and mammals, specifically African elephants, are likely to increase in the area if not.

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A SURVEY OF BIRDS OF KITWE WOODLAND IN KIGOMA-UJIJI MUNICIPALITY, WESTERN TANZANIA

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ABSTRACT

Urban and periurban forests and woodlands are crucial for maintaining biodiversity and can host many species of fauna, including birds. However, in East Africa, few studies have examined birds in urban and periurban forests and woodlands. Such studies are important for our understanding of how urban expansion and densification affect bird species. Using fixed-radius point counts, bird surveys were conducted in Kitwe woodland, which had not been surveyed previously. Sixty-nine species, comprising 35 families, were observed. The majority of the species observed were those that may utilize forest habitats, though are not dependent upon them (e.g., non-forest species). Of the species observed, only three were forest-dependent species; others were generalists. Six species were Palaearctic migrants. The results suggest that Kitwe woodland, albeit small in size and found in an urban environment, is important for the conservation of bird species in an expanding urban landscape and provides a “stopover” or “wintering area” for some Palaearctic migrants. Continued conservation of Kitwe woodland is recommended.

Keywords: *Birds, Conservation, Kitwe woodland, Point count, Urban landscape*

INTRODUCTION

The urban development leads to the fragmentation of large areas of natural vegetation into smaller patches. It creates islands of habitat patches in urban landscapes that vary in size, shape, edge circumference, vegetative make-up, connectivity to other patches, and surrounding matrix (Adams & Lindsey, 2010). Effects of urbanization on fauna can follow one of the three paradigms: diversity decay, homogenization, or community-composition turnover (Schultz *et al.*, 2012). In the species-decay paradigm, species are lost due to habitat loss. This leads to a decrease in species richness (Emlen, 1974; Walcott, 1974; Werema *et al.*, 2019) and functional diversity (Filippi-Codaccioni *et al.*, 2009), and to local species extinction (Smart *et al.*, 2006; Werema *et al.*, 2019). In the homogenization paradigm, non-native invaders replace local, native diversity. These invading species are usually habitat

generalists (McKinney & Lockwood, 2006). In the third paradigm, community-composition turnover, species composition varies with the type of habitat present at any given time (Schultz *et al.*, 2012). In some cases, patches of natural vegetation in urban areas have been considered crucial for protecting birds from urban alterations (Cicero, 1989; Mlingwa, 1992; Werema *et al.*, 2019; 2025). In some urban landscapes, these patches have been regarded as biodiversity hotspots (Zhou & Chu, 2012), and the wildlife species inhabiting them are determined by patch size, proximity, and access to other patches or to landscapes within the surrounding matrix. In some instances, these patches can provide refuge for species whose native habitats have been largely lost (Angold *et al.*, 2006), even allowing forest specialists to inhabit them (Mlingwa, 1992; Park & Lee, 2000; Werema *et al.*, 2025). The patches of forest, thickets, or woodlands in urban landscapes may also provide dispersal

corridors or stepping stones among habitat patches (Sandström *et al.*, 2006; Bolger *et al.*, 2001).

In Tanzania, few bird studies have been conducted in urban green spaces to assess their conservation importance. The existing studies have been mainly conducted in Tanga (Kakoko & John, 2013), Morogoro (Rija *et al.*, 2014), and Dar es Salaam (Harvey & Howell, 1987; Mlingwa, 1992; Werema *et al.*, 2019; John & Kagembe, 2022; Werema *et al.*, 2025). As Tanzania becomes more densely populated, with some cities growing in size, and as anthropogenic activities continue to fragment natural vegetation, reducing green space in urban areas, it is important to continue assessing which avian species remain in these patches.

Kitwe woodland, located in a small peninsula bordering Lake Tanganyika within Kigoma-Ujiji Municipality, is one of the green spaces found in an urban environment. It is currently a woodland “island” surrounded by urban settlement and water, and has not been investigated ornithologically. Since urban and periurban forests and woodlands are crucial for maintaining biodiversity and can host many species of fauna, it is important to have an inventory of avian species occurring in the Kitwe woodland. The study aims to assess the role of this woodland in the conservation of birds. It also provides baseline data that can be compared in the future, such as a century from now (e.g., Schultz *et al.*, 2012).

This study in Kitwe woodland is still important in its own way because avian studies in western Tanzania remain few. The available ones include those by Archer (1994), who reported birds observed from Kigoma to Kalambo Falls on the eastern side of Lake Tanganyika. Stanford & Msuya (1995) in Gombe National Park, and more recently by John *et al.* (2019) covering Gombe National Park, Mahale

National Park, Ntakata Forest, and Malagarasi Ramsar site. Other avian studies in this region were conducted by Stanford (1994) on raptor migration in western Tanzania. From these studies, it has been revealed that the western rift area of western Tanzania appears to be a major migratory route for many Palaearctic migrants (Stanford, 1994; Stanford & Msuya, 1995). As such, Kitwe woodland, albeit small, can be an important stopover habitat for migratory birds.

MATERIALS AND METHODS

Study area

The study was conducted in the miombo woodland at Kitwe within the Kigoma-Ujiji Municipality along Lake Tanganyika (4 ° 54' S, 29 ° 36' E). This patch of miombo woodland was set aside in 1981 by the Wildlife Division of Tanzania as a Game Sanctuary for conservation education and recreation. The study area is characterized by a tropical rainy climate, with rainfall varying from 900-1050 mm, and experiences one long wet season from November to May and one long dry season from June to October (Kigoma Water Master Plan, 1980). Kitwe woodland is about 120 ha and provides an example of the potential for recovery of degraded miombo woodland, while also serving as a ‘living classroom’ in agroforestry issues and water source protection. The general area is a mosaic of closed and open woodlands of the following tree species: *Dalbergia melanoxylon*, *Khaya anthotheca*, *Rhus natalensis*, *Ficus sp.*, *Julbernadia globiflora*, and *Pterocarpus angolensis*. Also, the area has some small patches of savanna grassland dominated by *Hyparrhenia filipendula*, *Eragrostis pateus*, and *Pennisetum polystachyon*. The woodland borders Lake Tanganyika in the south, west, and east. To the north, it borders newly established settlements within Kigoma-Ujiji Municipality (Fig. 1).

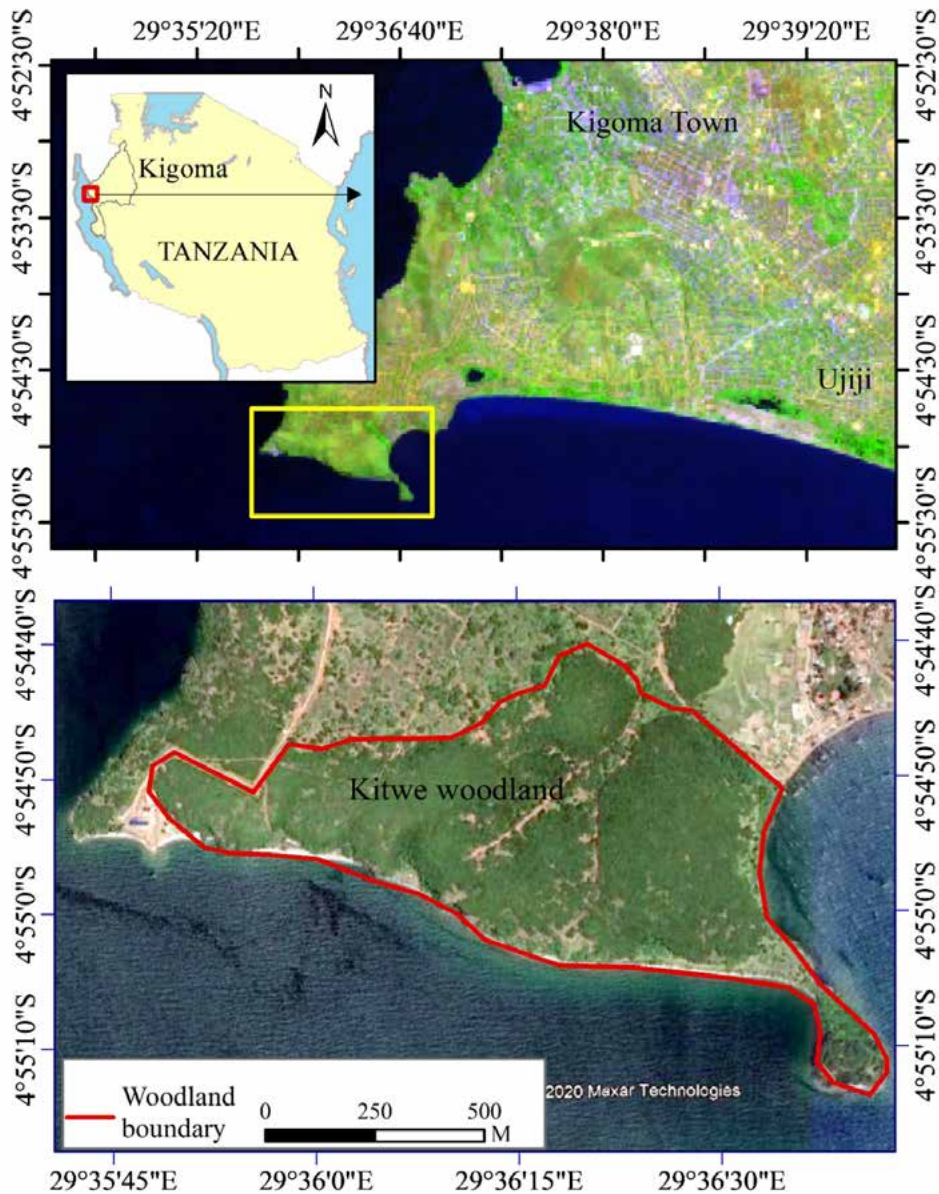


Figure 1: Map of the study area showing the location of Kitwe woodland within Kigoma - Ujiji Municipality (source: Google Earth).

The study area was once characterized by pristine Miombo woodland (Chepstow-Lusty *et al.*, 2006; Mganga & Lyaruu, 2016). It was later encroached upon by local communities who collected firewood to smoke fish, a practice that led to further deforestation and woodland degradation. In 1994, the Wildlife Division offered occupancy to the Jane Goodall Institute (JGI), which provided protection of the woodland from anthropogenic disturbances for about thirty years. This was

done in collaboration with the United Republic of Tanzania and the United States Agency for International Development (USAID) under the management of the JGI.

Data collection

We surveyed birds using the fixed-radius point-count method (Hutto *et al.*, 1986). A total of thirty-one point counts were carried

out on June 5-6, 2018, November 8, 2018, and September 4-5, 2019. Point counts were located at least 150-200 m apart, and each was surveyed for 10 minutes during the morning (0700-1100 hrs) and the evening (1600-1800 hrs). Point counts were randomly positioned such that each was counted once. Birds flushed from the measurement area on approach, as well as those using the airspace, such as aerial insectivores or raptors flying low over the canopy that appeared to be searching for prey within the study site, were considered to use the woodland. Birds seen foraging on the beach side of the lake, which were using the woodland to perch (e.g., Pied Kingfisher), were recorded. Birds solely using the beach (e.g., sandpipers and herons) were not recorded.

Data analysis

A list of species found in the study area was obtained by summing the number of species detected. Occurrence of any given species was expressed in terms of its relative abundance and detectability. Species relative abundance was expressed as the percent composition of the number of individuals of a particular species divided by the total number of individuals across all species. Species detectability was obtained by dividing the maximum number of point counts in which a species was detected by the total number of point counts (in this case, $n = 31$). This value was expressed in percentage. A combination of these two assessments (i.e., species relative abundance and species detectability) was used to assess whether a species was common, fairly common, uncommon, or rare (Also see Stanford & Msuya 1995, Appendix 1).

While some bird species have intrinsic conservation interest because they are rare, endangered, or endemic, the importance of a particular bird species can indicate habitat condition and value (Bennun *et al.*, 1996). The presence of forest-dependent bird species in a woodland habitat would thus indicate the conservation importance of such a woodland. As such, the conservation importance of Kitwe

woodland (which has some parts with closed woodland) was also assessed based on whether it is used by forest-dependent bird species. To assess whether Kitwe woodland was used by forest species, the birds were divided into two groups: forest-dependent and non-forest-dependent species using Bennun *et al.* (1996). Forest-dependent species were further divided into *FF* species (forest specialists), which are 'true' forest birds of the interior undisturbed forest, and *F* species, which may occur in undisturbed forest but are in most cases found in forest strips, gaps, and edges. Non-forest species are usually recorded in open habitats, including wooded grasslands, and are not dependent on forests.

Families, common names, and scientific names follow the International Ornithological Community World Bird Names 9.2 checklist (Gill & Donsker, 2019).

RESULTS AND DISCUSSION

Eight hundred and ten individual birds comprising 69 species and 35 families were observed (Appendix 1). None of the species recorded are globally threatened, and no forest specialist species (*FF*) were detected. The five most abundant species, in terms of their relative abundances and detection probabilities, were Common Bulbul (*Pycnonotus barbatus*), Tawny-flanked Prinia (*Prinia subflava*), Black-backed Puffback (*Dryoscopus cubla*), White-browed Scrub Robin (*Cercotrichas leucophrys*), and Bronze Mannikin (*Lonchura cuculata*) (Appendix 1). These are generalists and adapt well to disturbed habitats. The remaining species were classed as either fairly common, uncommon, or rare (see Stanford & Msuya, 1995). Despite the brevity of the study, the small size of the woodland, and its location within an urban environment, the results indicate that Kitwe woodland may host a disproportionately high number of species. However, with 69 species, the avian assemblage at Kitwe woodland is depauperate compared to existing forests in western Tanzania, where birds have been surveyed.

For example, compared with Gombe National Park to the north, Stanford & Msuya (1995) recorded 225 species. The presence of fewer bird species in Kitwe woodland compared to Gombe National Park could be due to its small size and the lack of a closed forest or a large closed woodland. Gombe National Park is relatively large and has a mosaic of habitats (i.e., higher habitat diversity), including high-canopy riverine forest, open woodlands, and lower-canopy woodlands (Stanford & Msuya, 1995), resulting in higher bird species richness (see Bett *et al.*, 2016). The current isolation of Kitwe woodland certainly hinders the movements of some bird species that would have originated from any remnant forests or woodlands in the surrounding landscape. This is because the presence of some bird species (e.g., forest species) in isolated forests or woodlands depends, among other factors, on the distance to remnant forest or woodland patches and on local structural diversity in the surrounding habitats (Laube *et al.*, 2008).

In a degraded small woodland surrounded by urban areas, we would expect the avifauna to be a diminished version of what one can find in less disturbed nearby regions. For example, about 50% of the species recorded during this study have been recorded by Archer (1994) to occur between Kigoma and Kalambo Falls on the eastern side of Lake Tanganyika. Furthermore, about 74% of the species recorded in this study have been reported by Stanford & Msuya (1995) to occur in Gombe National Park. All species recorded during this study have been reported by John *et al.* (2019) as occurring in western Tanzania (including Gombe National Park). The presence of the Lesser Masked Weaver *Ploceus intermedius* in Kitwe woodland suggests a range extension for this species as it has not been recorded in the northern section of Lake Tanganyika (Stevenson & Fanshawe, 2002), though there are records from Mahale National Park (John *et al.*, 2019).

With only three forest-dependent species (F species; the Narina Trogon - *Apaloderma*

narina, Red-capped Robin Chat - *Cossypha natalensis*, and Black-backed Puffback - *D. cubla*), Kitwe woodland hosts some species which can also be found in undisturbed forest (although in most cases found in forest strips, gaps, and edges) (see Bennun *et al.*, 1996). Since forest-dependent bird species appear to be less adaptable than forest visitors, many of them also have relatively limited distributions (see Bennun *et al.*, 1996). Notwithstanding, the presence of three forest-dependent species at Kitwe woodland underscores its conservation importance, despite its small size. Thus, the low proportion of forest-dependent species reflects that the area is small, disturbed, and still recovering, and could be unsuitable for habitation by most forest-dependent species. The majority of the species observed were forest-inhabiting but not dependent on it (e.g., generalists). The dominance of non-forest dependent species over forest dependent ones probably suggests homogenization (Schultz *et al.*, 2012) due to human disturbance during the past and the fact that Kitwe is almost an 'island patch' being bounded by water on the western, southern, and eastern sides and settlements/buildings on the northern side.

During the study, seven Palaearctic migrant species, including European Bee-eater (*Merops apiaster*), Barn Swallow (*Hirundo rustica*), Willow Warbler (*Phylloscopus trochilus*), Garden Warbler (*Sylvia borin*), Spotted Flycatcher (*Muscicapa striata*), Red-backed Shrike (*Lanius collurio*), and Isabelline Shrike (*Lanius isabellinus*), were recorded. The presence of Palaearctic migrants corroborates Stanford's (1994) findings that the western rift area of western Tanzania is a migratory route for many transcontinental migrant bird species. Since the study was conducted for a limited number of field days, it is expected that more Palaearctic migrant species use this woodland than is currently known. Stanford & Msuya (1994), who observed birds at Gombe National Park for 4 years, recorded more Palaearctic migrants than our study. A larger list of Palaearctic migrants would certainly be compiled if more time were devoted throughout

the year of observations, particularly during the migration season (see Stanford & Msuya, 1995). Thus, Kitwe woodland, albeit small in size and found in an urban environment, provides a “stopover” or “wintering area” for Palaearctic migrants.

At a local scale, the study has demonstrated the role of one of the remnant woodlands found in urban landscapes in the conservation of local birds. As recommended by Werema *et al.* (2019), it is high time for city/municipal planners and managers planning new towns to consider setting aside large remnant patches in urban landscapes to preserve the fauna biodiversity that depends on these forest patches or thickets. As such, given the importance of Kitwe woodland as a habitat for some bird species, its continued protection by the JGI in collaboration with the government of Tanzania is commended. Further surveys on other fauna, including ecosystem services providers such as pollinators (e.g., butterflies and bees), are also recommended.

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Appendix 1. List of species recorded from Kitwe woodland.

Key (adopted from Stanford & Msuya 1995).

c common; likely to be seen or heard every day.

fc fairly common; likely to be seen or heard more than 50% of days in the field, but not every day.

uc uncommon; likely to be seen or heard 20-50% of days in the field.

r rare; sighted only once or a few times, or likely to be seasonal in abundance.

pm Palearctic migrant.

Family and species	Species relative abundance	Detection probability	Comment
Phasianidae			
Hildebrandt's Francolin (<i>Pternistis hildebrandti</i>)	0.25	3.23	r
Red-necked Spurfowl (<i>Pternistis afer</i>)	0.25	3.23	r
Accipitridae			
Palm-nut Vulture (<i>Gypohierax angolensis</i>)	2.35	22.58	fc
African Marsh Harrier (<i>Circus ranivorus</i>)	0.12	3.23	r
Black Kite (<i>Milvus migrans</i>)	2.72	3.23	r
African Fish Eagle (<i>Haliaeetus vocifer</i>)	0.12	3.23	r
Columbidae			
Laughing Dove (<i>Spilopelia senegalensis</i>)	0.12	3.23	r
Emerald-spotted Wood Dove (<i>Turtur chalcospilos</i>)	0.62	6.45	r
Blue-spotted Wood Dove (<i>Turtur afer</i>)	0.37	6.45	r
Cuculidae			
White-browed Coucal (<i>Centropus superciliosus</i>)	0.49	9.68	uc
Jacobin Cuckoo (<i>Clamator jacobinus</i>)	0.12	3.23	r
Klaas' Cuckoo (<i>Chrysococcyx klaas</i>)	0.12	3.23	r
Caprimulgidae			
Nightjar (<i>Caprimulgus</i> sp.)	0.12	3.23	r
Apodidae			
African Palm Swift (<i>Cypsiurus parvus</i>)	2.10	16.13	uc
Coliidae			
Speckled Mousebird (<i>Coilus striatus</i>)	1.60	6.45	uc
Trogonidae			
Narina's Trogon (<i>Apaloderma narina</i>)	0.25	3.23	r
Alcedinidae			
African Pygmy Kingfisher (<i>Ispidina picta</i>)	0.12	3.23	r

Family and species	Species relative abundance	Detection probability	Comment
Giant Kingfisher (<i>Megaceryle maxima</i>)	0.25	3.23	r
Pied Kingfisher (<i>Ceryle rudis</i>)	0.12	3.23	r
Meropidae			
Little Bee-eater (<i>Merops pusillus</i>)	1.36	6.45	r
Eurasian Bee-eater (<i>Merops apiaster</i>)	1.85	12.9	uc, pm
Indicatoridae			
Greater Honeyguide (<i>Indicator indicator</i>)	0.12	3.23	r
Picidae			
Cardinal Woodpecker (<i>Dendropicos fuscescens</i>)	0.25	6.45	r
Platysteiridae			
Chinspot Batis (<i>Batis molitor</i>)	1.73	16.13	uc
Malaconotidae			
Orange-breasted Bush-shrike (<i>Chlorophoneus sulfureopectus</i>)	0.74	16.13	uc
Black-crowned Tchagra (<i>Tchagra senegalus</i>)	1.36	25.81	fc
Black-backed Puffback (<i>Dryoscopus cubla</i>)	3.46	45.16	c
Tropical Boubou (<i>Laniarius major</i>)	2.20	32.36	c
Campephagidae			
Black Cuckoo-shrike (<i>Campephaga flava</i>)	0.12	3.23	r
Laniidae			
Red-backed Shrike (<i>Lanius collurio</i>)	0.25	3.23	r, pm
Isabelline Shrike (<i>Lanius isabellinus</i>)	0.12	3.23	r, pm
Oriolidae			
Black-headed Oriole (<i>Oriolus larvatus</i>)	0.25	6.45	r
Monarchidae			
African Paradise Flycatcher (<i>Terpsiphone viridis</i>)	2.22	25.81	fc
Corvidae			
Pied Crow (<i>Corvus albus</i>)	7.16	25.81	fc
Pycnonotidae			
Common Bulbul (<i>Pycnonotus barbatus</i>)	7.53	61.29	c
Yellow-throated Leaflove (<i>Atimastillas flavicollis</i>)	0.37	6.45	r
Hirundinidae			
Barn Swallow (<i>Hirundo rustica</i>)	0.12	3.23	r, pm
Lesser Striped Swallow (<i>Cecropis abyssinica</i>)	0.25	3.23	r
Macrospenidae			
Red-faced Crombec (<i>Sylvietta whytii</i>)	0.86	19.25	fc
Phylloscopidae			
Willow Warbler (<i>Phylloscopus trochilus</i>)	0.25	6.45	r, pm
Cisticolidae			
Tawny-flanked Prinia (<i>Prinia subflava</i>)	5.93	48.39	c

Family and species	Species relative abundance	Detection probability	Comment
Grey-backed Camaroptera (<i>Camaroptera brevicaudata</i>)	2.10	29.03	fc
Green-capped Eremomela (<i>Eremomela scotops</i>)	0.49	3.23	r
Timaliidae			
Arrow-marked Babbler (<i>Turdoides jardineii</i>)	1.48	6.45	uc
Sylviidae			
Garden (Warbler <i>Sylvia borin</i>)	0.12	3.23	r, pm
Sturnidae			
Violet-backed Starling (<i>Cinnyricinclus leucogaster</i>)	3.70	3.23	uc
Muscicapidae			
White-browed Scrub Robin (<i>Cercotrichas leucophrys</i>)	1.98	35.48	c
Pale Flycatcher (<i>Bradornis pallidus</i>)	0.37	9.68	r
Spotted Flycatcher (<i>Muscicapa striata</i>)	0.25	6.45	r, pm
White-browed Robin Chat (<i>Cossypha heuglini</i>)	0.62	6.45	r
Red-capped Robin Chat (<i>Cossypha natalensis</i>)	2.72	25.81	fc
Collared Palm Thrush (<i>Cichladusa arquata</i>)	0.74	6.45	r
Nectariniidae			
Scarlet-chested Sunbird (<i>Chalcomitra senegalensis</i>)	1.73	12.90	uc
Passeridae			
House Sparrow (<i>Passer domesticus</i>)	0.37	3.23	r
Northern Grey-headed Sparrow (<i>Passer griseus</i>)	0.49	6.45	r
Ploceidae			
Holub's Golden Weaver (<i>Ploceus xanthops</i>)	3.58	22.58	fc
Lesser Masked Weaver (<i>Ploceus intermedius</i>)	1.60	3.23	r
Village Weaver (<i>Ploceus cucullatus</i>)	1.11	12.90	uc
Estrildidae			
Green-winged Pytilia (<i>Pytilia melba</i>)	3.09	19.35	fc
Red-billed Firefinch (<i>Lagonosticta senegala</i>)	1.60	12.90	uc
African Firefinch (<i>Lagonosticta rubricata</i>)	0.37	3.23	r
Jameson's Firefinch (<i>Lagonosticta rhodopareia</i>)	0.25	3.23	r
Red-cheeked Cordon-bleu (<i>Uraeginthus bengalus</i>)	1.60	12.9	uc
Bronze Mannikin (<i>Lonchura cucullate</i>)	16.42	35.45	c
Motacillidae			
African Pied Wagtail (<i>Motacilla aguimp</i>)	0.62	9.68	r
Fringillidae			
Western Citril (<i>Crithagra frontalis</i>)	0.25	3.23	r
Yellow-fronted Canary (<i>Crithagra mozambicus</i>)	0.25	3.23	r
Brimstone Canary (<i>Crithagra sulphurata</i>)	0.25	3.23	r
Emberizidae			
Cabanis's Bunting (<i>Emberiza cabanisi</i>)	0.25	6.45	r

SEASONAL VARIATION IN RANGING PATTERNS AND ACTIVITY BUDGET OF YELLOW BABOONS IN THE ISSA VALLEY, TANZANIA

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ABSTRACT

Understanding how primates adjust their behaviour in response to environmental variability is critical for predicting their resilience to climate change. The yellow baboon (*Papio cynocephalus*) is a highly adaptable omnivore, yet its fine-scale behavioural responses to seasonal and short-term weather variation remain understudied in western Tanzania. This study examined how seasonality, temperature, and rainfall influence daily travel distance and activity budgets of yellow baboons in the Issa Valley. A total of 14,028 behavioural observations were collected using scan and focal sampling between 06:00 am to 06:30 pm, while weather data were recorded at 30-minute intervals using an automated HOBO station. Daily travel distance was analyzed using a Gamma GLM, revealing a strong and significant effect of seasonality ($\beta = 0.93875$, $SE = 0.07594$, $t = 12.36$, $p < 0.001$). Predicted travel distances varied across sub-seasons: early dry (2.56 ± 0.21 km/day), late dry (2.97 ± 0.25 km/day), early wet (3.19 ± 0.23 km/day), and late wet (2.31 ± 0.17 km/day). Baboons travelled 16% farther in the late dry season and 24% farther in the early wet season relative to the early dry baseline, reflecting resource scarcity followed by early vegetation renewal. Activity budgets, analyzed using multinomial logistic regression, showed that rainfall had the strongest behavioural influence. It significantly reduced copulation, foraging, grooming, play, and other activities, while increasing resting behaviour. Temperature alone had minimal effects, but its interaction with rainfall significantly moderated these impacts. Mean temperatures were consistent ($\sim 19.9^\circ\text{C}$), whereas rainfall more than doubled in the wet season. These findings highlight rainfall-driven seasonality as the primary determinant of baboon movement and behaviour, with important implications under future climate variability.

Keywords: Activity Budget, Yellow Baboons, Issa Valley, Season, and Tanzania.

INTRODUCTION

Baboons (*Papio spp.*) are one of the most adaptable and widespread genera globally, inhabiting diverse habitats across various climates and latitudes. Their ecological success is largely due to their omnivorous and flexible diet, complex social organization, and high behavioural plasticity, allowing them to thrive in both terrestrial and arboreal environments (Bronikowski & Altmann, 1996). These traits enable yellow baboons to cope with a range of environmental conditions, including human-altered landscapes, making them a critical species for understanding primate responses to environmental changes (Mason *et al.*, 2022).

Research on a wide range of primate species (Kosheleff & Anderson, 2009; Manyama, 2025a, 2025b) has documented variations in activity budgets in response to environmental factors such as temperature and rainfall. For instance, in South Africa, during periods of low food availability, chacma baboons (*Papio ursinus*) adopt time-minimizing strategies but switch to energy-maximizing strategies when food is scarce (Dzingwena *et al.*, 2025). This shift involves expanding their diet to include higher-return foods and increasing their foraging time (Bronikowski & Altmann, 1996; Johnson *et al.*, 2015a).

Changes in rainfall timing, intensity, and temperature regimes can reduce the predictability and availability of essential resources, particularly water and food resources (Manyama, 2025c), which directly influence their daily travel effort and other activities. Seasonal variations driven by these climatic changes may exacerbate resource scarcity, forcing baboons such as yellow baboons, chacma baboons, and other species to adjust their movement, foraging strategies, and overall activity budgets (Dzingweni *et al.*, 2025; Post, 1981; van Doorn *et al.*, 2010). Despite living in protected habitats in western Tanzania, the Issa Valley yellow baboon populations remain vulnerable to these environmental stressors, yet there is limited understanding of how fine-scale seasonal and weather variability affects daily behavioural patterns and ranging behaviour.

Addressing this gap is crucial for predicting how baboons will respond to shifting ecological conditions and for guiding effective conservation strategies under future climate scenarios. Understanding how environmental change reshapes the contexts in which baboons make behavioural trade-offs is particularly important, as these climatic pressures ultimately structure their daily activity budgets and long-term energy allocation. While baboons demonstrate notable behavioural flexibility, tolerating wide thermal variation and adjusting their activity through thermoregulatory behaviours, this resilience functions within physiological limits. The study investigated the influence of seasonality on the yellow baboon activity budget in the Issa valley, western Tanzania. Specifically, we assessed travel speed and distance as well as how temperature and rainfall influence overall activity budgets.

We hypothesized that (1) seasonal variation would significantly affect the daily travel

distances of yellow baboons in the Issa Valley, with individuals travelling faster and covering longer distances during the early wet and late dry seasons compared to the late wet and early dry seasons; and (2) temperature and rainfall would significantly influence baboon activity budgets, such that higher temperatures and increased rainfall would be associated with reduced foraging and travelling time, alongside corresponding increases in resting and social behaviours.

MATERIALS AND METHODS

Description of the study Area

The Issa Valley, located in western Tanzania, lies in the northern part of the Greater Mahale Ecosystem (GME) and is part of the Tongwe East Forest Reserve (Fig. 1). The vegetation in Issa Valley consists of deep valleys with evergreen riparian forests, steep slopes, and broad flat plateaus mainly covered by miombo woodland. It also features rocky outcrops and grasslands that are flooded seasonally (Drummond-Clarke *et al.*, 2022).

The Issa Valley supports a wide range of medium and large mammalian species (Piel *et al.*, 2020) as well as more cryptic, small mammals (D'Ammando *et al.*, 2022). Besides yellow baboons, Issa hosts seven primate species, including red-tailed monkeys (*Cercopithecus ascanius*), greater (*Otolemur crassicaudatus*) and lesser galagos (*Galago senegalensis*), vervet monkeys (*Chlorocebus aethiops*), red colobus monkeys (*Piliocolobus tephrosceles*), and chimpanzees (*Pan troglodytes schweinfurthii*). Additionally, and large predators are regularly observed; leopard, lions, hyena and side-stripe jackal as well as large ungulate like buffalo and eland.

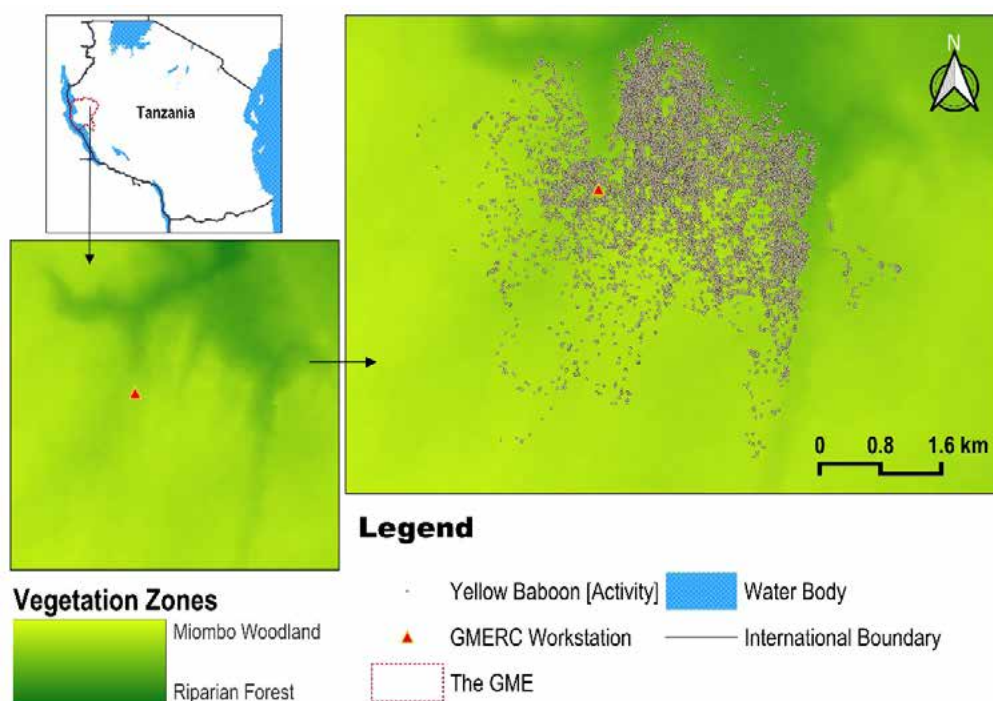


Figure 1. A map showing the study site (Issa Valley) and the home range of the yellow baboons (8.681 km²).

Study subjects

The study involved the focal yellow baboon troop, comprising 27 individuals. This group has been habituated since 2011 and has been followed regularly since 2012 (Johnson *et al.*, 2015b). Commonly referred to as the “Camp Troop”, it occupied a home range of approximately 4 km², which showed minimal seasonal variation. Over the years, the troop size had oscillated between 27 and 9 individuals, and by 2022, the home range of yellow baboons had expanded slightly to 8.681 km².

Data collection

We employed direct observation sampling methods: scan and focal follows (Altmann *et al.*, 1974). For focal follows, we randomly selected an individual and recorded data every 5 minutes, including activity (feeding, resting, travelling, and social), vegetation type, and GPS coordinates using Open Data Kit (ODK) and Garmin GPS, respectively, from 06:00 am to 06:30 pm. Weather data recorded on Onset

H8 Pro series logger (HOBO RX3000 Station) deployed at the base station (in the centre of the baboon home range) in woodland vegetation. The HOBO recorded ambient temperature every 30 mins and rainfall every 0.2 mm, a S-RGB-M002. There are distinct seasons at Issa: a rainy season (November-April) and a dry season (May-October), with the dry months defined as having <100 mm of rainfall (Johnson *et al.*, 2015b).

Data analysis

We conducted all statistical analyses in R version 4.3.2 (R Core Team, 2023). To assess how climatic variables influence yellow baboon behaviour, we applied a multinomial logistic regression model, as the response variable; activity type was categorical, including behaviours such as foraging, resting, travelling, and grooming. Predictor variables consisted of mean hourly temperature (°C), total daily rainfall (mm), and season (dry vs. wet). Daily travel distance was analysed using a Gamma Generalized Linear Model (GLM).

Travel distance was calculated by summing GPS step lengths between consecutive location fixes within each observation period and then averaging these values per day. All model assumptions were assessed by examining multicollinearity among predictors and through the use of diagnostic plots to ensure the robustness of the analyses.

RESULTS

view of seasonal variation in observation effort

A total of 14,028 behavioural observations were recorded during the study period (Table 1). Observation effort and data availability were higher in the wet season (67.3%) than in the dry season (32.7%), likely reflecting longer observation periods or increased baboon visibility and activity during wetter months. Mean temperatures were relatively similar between seasons (approximately 19.9°C). In contrast, mean rainfall was more than twice as high in the wet season (0.4832 mm) as in the dry season (0.2672 mm), confirming clear climatic differences between the two seasons.

Table 1. Summary of behavioural observation effort based on season

Season	Observations	Mean Temperature (°C)	Mean Rainfall (mm)	Percentage (%)
Dry	4587	19.8870	0.2672	32.7
Wet	9441	19.9868	0.4832	67.3

The influence of seasonal change on the daily travel distance of yellow baboons in Issa Valley

Under the Shapiro test of normality ($p < 0.05$) and Q-Q plot (Fig. 2), daily travel distances were not normally distributed over the dataset. This suggested the use of the Gamma GLM analysis due to its full distributional assumptions and variance structure on skewed continuous data.

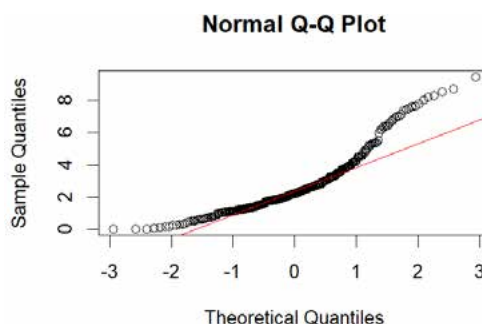


Figure 2. The Normal Q-Q plot shows that the daily travel distances were not normally distributed, and are not suitable for parametric statistical tests.

The Gamma GLM analysis showed that seasonal change had a significant effect ($\beta = 0.93875$, $SE = 0.07594$, $t = 12.362$, $p < 0.001$) on the daily travel distance of yellow baboons in Issa Valley. This suggests that seasonality influences baboons' daily travel rate (Fig. 3). The back-transformed predictions from the Gamma GLM revealed clear seasonal variation in baboon daily travel distance (Fig. 4). Using the early dry season as the reference (2.56 ± 0.2116 km/day), travel distances increased in both the late dry and early wet seasons, reaching approximately (2.97 ± 0.2544 km/day) and (3.19 ± 0.2266 km/day), respectively. These estimates indicate that baboons travelled about 16% farther in the late dry season and 24% farther in the early wet season compared to the early dry period. In contrast, travel distances declined during the late wet season (2.31 ± 0.1708 km/day), representing a slight reduction relative to the reference season.

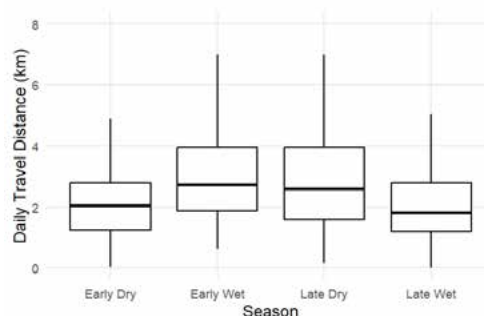


Figure 3. The box plot shows that the daily travel distances of yellow baboons were influenced by seasonality in Issa Valley.

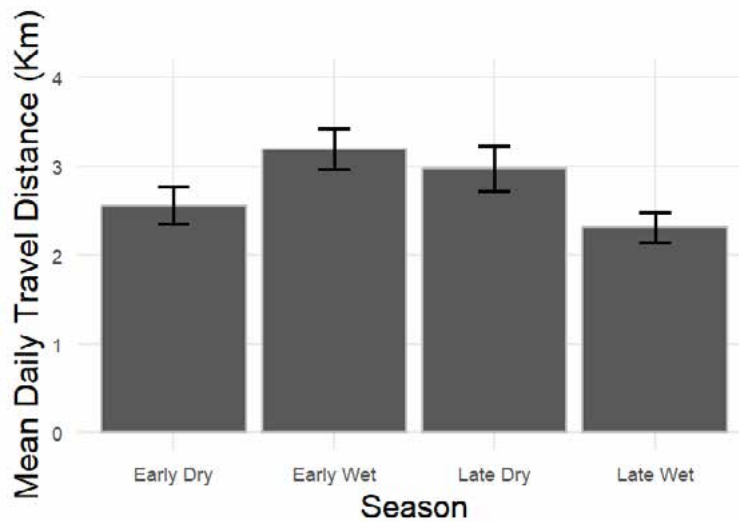


Figure 4. The bar graph with SE shows the predicted mean daily travel distance covered by yellow baboons in Issa Valley.

The influence of temperature, rainfall, and their interaction variation on the activity budget of the yellow baboon in Issa Valley

The rainfall had the strongest influence on baboon activities, significantly reducing

behaviours such as copulation, foraging, grooming, and playing, and increasing the resting time of the yellow baboon, while temperature alone had a minimal effect on any activity (Figure 5).

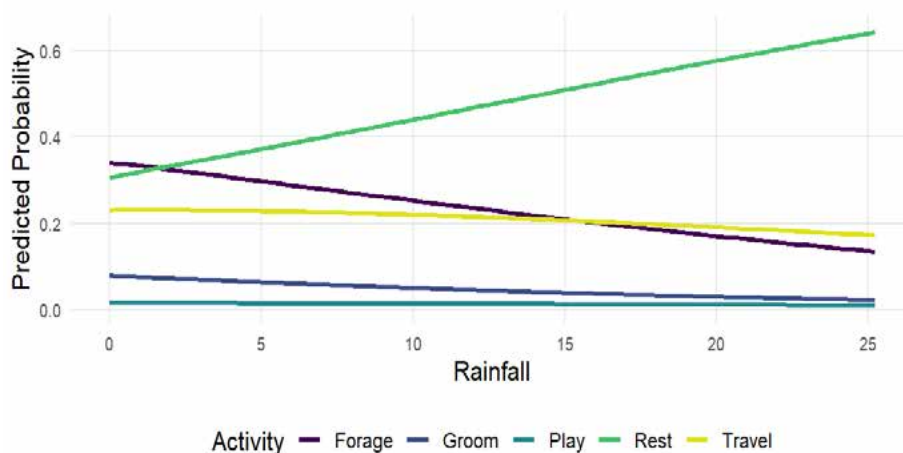


Figure 5. The histogram shows the prediction of the rainfall effect on yellow baboon activities in Issa Valley.

The interaction between temperature and rainfall was significant for most activities, indicating that warmer conditions slightly mitigated the suppressive effects of rainfall (Table 2).

Table 2. The influence of temperature and rainfall on yellow baboon activities

Activity	Predictor	Coefficient	P-value	Interpretation
Copulate	Rainfall	-1.4342	0.0015	Heavy rain significantly reduced the likelihood of copulation.
	Interaction Effect of Temperature and Rainfall	0.1057	0.0009	The negative effect of rain on copulation was slightly mitigated under warmer conditions.
Forage	Rainfall	-0.8293	0.0429	Rain significantly reduced the likelihood of foraging.
	Interaction Effect of Temperature and Rainfall	0.0737	0.0173	The negative effect of rain on foraging was slightly mitigated at higher temperatures.
Groom	Rainfall	-0.9915	0.0192	Rain significantly reduced grooming behaviour.
	Interaction Effect of Temperature and Rainfall	0.0812	0.0097	The negative effect of rain on grooming was slightly lessened under warmer conditions.
Other	Rainfall	-5.4239	0.0147	Heavy rain significantly suppressed other activities.
	Interaction Effect of Temperature and Rainfall	0.2362	0.0007	Warmer temperatures slightly reduced the negative effect of rain on other activities.
Play	Rainfall	-0.9837	0.0334	Rain significantly reduced the likelihood of play.
	Interaction Effect of Temperature and Rainfall	0.0825	0.0116	The suppressive effect of rain on play was slightly alleviated at higher temperatures.
Rest	Interaction Effect of Temperature and Rainfall	0.0747	0.0157	Warmer conditions slightly mitigated the negative effect of rain on resting.
Travel	Interaction Effect of Temperature and Rainfall	0.0725	0.0194	The negative effect of rain on travelling was slightly alleviated at higher temperatures.

DISCUSSION

The seasonal variation in observation effort

Higher observation effort during the wet season may reflect improved visibility due to increased baboon activity, extended daytime movement, and greater accessibility of observation sites, consistent with research showing that primate detectability varies with ecological conditions (Manyama, 2025b; P Stoltz & Saayman, n.d.). These environmental contrasts provide an essential framework for interpreting how yellow baboons adjust their ranging behaviour

and activity budgets in response to both predictable seasonal shifts and short-term weather fluctuations (Bronikowski & Altmann, 1996).

The influence of seasonal change on the daily travel distance of yellow baboons in Issa Valley

Seasonality significantly influenced daily travel distance in Issa Valley yellow baboons, and the seasonal movement pattern was ecologically meaningful. Baboons travelled the least in the early dry season, increased

their travel distances in the late dry and early wet seasons, and reduced movement again during the late wet season. These patterns are consistent with the behavioural ecology of baboons inhabiting seasonal habitats, where energy expenditure is tightly coupled with resource distribution (Bronikowski & Altmann, 1996; Manyama, 2025c; Stelzner, 1988). Ecologically, this pattern reflects shifting resource landscapes across the annual cycle. Movement peaks from the late dry to the early wet season likely reflect a transition from resource scarcity to early resource renewal. During the late dry season, water and food sources become increasingly limited, forcing baboons to travel farther in search of reliable patches (Bronikowski & Altmann, 1996; Post, 1981). During the early dry season, vegetation and water resources, although declining, remain sufficiently predictable. Baboons may exploit remaining high-quality patches with minimal need for long-range movement. Reduced travel distances at this time suggest energy conservation strategies that align with ecological models of foraging, which predict that animals minimize movement when resource density is acceptable and predictable (Bronikowski & Altmann, 1996; Johnson *et al.*, 2015b; Kashetsky *et al.*, 2021; Stelzner, 1988).

The late dry season brought a notable increase in travel distance. This reflects intensified resource scarcity as temporary water sources disappear remain to depend on permanent drainage rivers, fruiting trees decline, and herbaceous vegetation dries out. Baboons are forced to expand their foraging radius to locate scattered, often low-quality food patches. Similar patterns have been observed in chacma baboons in southern Africa, where dry-season resource scarcity drives increased day-range length (Dzingwena *et al.*, 2025). These findings reveal that Issa baboons confront energetic challenges common to primates in semi-arid settings.

The highest travel distances occurred in the early wet season. This may appear counterintuitive given the onset of resource

renewal, but newly sprouting vegetation and emerging insects occur in spatially patchy distributions, requiring baboons to cover larger areas to exploit fresh, high-quality foods (Manyama, 2025c). The first rains often bring short-lived flushes of green vegetation that are ephemeral and scattered. Increased travel, therefore, reflects active searching and selective foraging strategies. Increased ranging during this period has also been reported in vervet monkeys, reinforcing that early wet-season productivity can inconsistently increase movement (Young *et al.*, 2019). The decline in travel distance during the late wet season supports the hypothesis that resource abundance enhances foraging efficiency (Bronikowski & Altmann, 1996; Johnson *et al.*, 2015b; Manyama, 2025c). The landscape becomes greener, water is ubiquitous, and many plant species reach peak productivity. Under these circumstances, baboons can meet nutritional needs within smaller foraging areas. Reduced travel minimizes energetic costs, aligns with the optimal foraging theory, and has been observed in multiple primate populations.

The influence of temperature, rainfall, and their interaction variation on the activity budget of the yellow baboon in Issa Valley

Rainfall suppresses activity in baboons for several interconnected ecological and physiological reasons (van Doorn *et al.*, 2010). Wet fur in yellow baboons reduces insulation and increases heat loss, creating a thermal constraint that discourages energetically costly activities, especially at moderate or cool temperatures (Hill, 2006). Visibility is likewise reduced during rainfall, impairing predator detection, also, they are at risk of ecto- and endo-parasites invasion (Bronikowski & Altmann, 1996), as well foraging efficiency declines because rain obscures ground-based foods (Altmann' And & Muruth12, 1988).

The observed decrease in copulation, grooming, and play during rainfall further demonstrates that precipitation disrupts not only foraging but also social cohesion

and mating opportunities. Over time, such short-term behavioural reductions may have cumulative consequences for group stability, social bonding, and reproduction, especially if climate change increases the frequency or intensity of rainfall extremes (Gesquiere *et al.*, 2008).

Temperature alone showed limited influence on baboon behaviour, reinforcing evidence that these primates tolerate a broad range of thermal conditions. Baboons rely heavily on behavioural thermoregulation, seeking shade, sunning, altering posture, and adjusting the timing of activity, to maintain thermal comfort despite daily fluctuations (Hill, 2006; Stelzner, 1988). However, the interaction between temperature and rainfall revealed important context-dependent responses. Warmer temperatures appeared to buffer some of the negative effects of rain: grooming and play were more feasible because warmth reduced the risk of chilling; foraging resumed sooner since wet conditions were less physiologically stressful; and movement became less energetically costly compared with cold, wet conditions.

CONCLUSION

These season patterns carry important implications for understanding how baboons will cope with future environmental change. Although their behavioural flexibility allows them to tolerate wide thermal variation and to mitigate some of the immediate impacts of rainfall, this resilience has limits. The buffering effect of warm temperatures suggests that baboons can sustain travel, social and foraging activities under mild weather disturbances, but more severe combinations such as prolonged rainfall paired with low temperatures may significantly constrain essential behaviours tied to energy balance, social cohesion, and reproduction. As climate change intensifies weather unpredictability, increasing the frequency of extreme storms and cool-wet conditions, baboons may face heightened physiological stress and reduced opportunities for feeding, bonding, and mating. Consequently,

while temperature alone may not shape behaviour strongly, the interaction of multiple climatic variables will become increasingly critical in determining baboon activity patterns and overall fitness, highlighting the need for continued monitoring of primate behavioural ecology under shifting climatic regimes.

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SPATIAL VARIATION OF THE INDIAN HOUSE CROW (*Corvus Splendens Splendens*) BREEDING BIOLOGY, EFFECTIVENESS AND SUSTAINABILITY OF CONTROL MEASURES - A REVIEW

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ABSTRACT

The Indian House Crow (IHC; *Corvus splendens splendens*) is one of the most successful members of the genus *Corvus*. Due to its high adaptability, the species has expanded its range across all continents except Antarctica, causing significant social, ecological, and economic impacts. Various methods have been used to control, manage, or eradicate the species, but most have proven ineffective. Poisoning results in high non-target mortality and biodiversity loss. Trapping is hampered by the crows' intelligence and ability to learn to avoid traps, while shooting requires government permits and is complicated by the species' close association with human settlements. Biological control is often considered a promising alternative. However, the global variation in the breeding biology of the IHC remains poorly understood. This review aims to examine global differences in the breeding biology of the IHC and their implications for biological control. A desk literature review was conducted, and 75 relevant studies were identified from Google Scholar, EBSCOhost, and the IHC 500-Database. Findings reveal regional differences in the breeding season, which mostly occur during the dry season. The nesting sites are determined by the structural features of trees, rather than their abundance. The average clutch size ranges from 3.5 to 4.5, and urban areas show higher breeding success compared to rural areas. There is limited information on extra-pair paternity in IHC. In conclusion, many aspects of breeding biology vary geographically, while others are insufficiently documented. Therefore, for effective biological control, management, and eradication of the IHC, strategies should be localised.

Keywords: Corvids, reproductive biology, spatial variation, nest success, and reproductive success

INTRODUCTION

The Indian House Crow (IHC; *Corvus splendens*), a member of the order Passeriformes, family Corvidae, and genus *Corvus*, which comprises over 40 species including ravens, jays, jackdaws, and magpies (Anjum *et al.*, 2021; Haring *et al.*, 2012; Weissensteiner *et al.*, 2020) 6,24]], "issued": {"date-parts": [{"2021"}]}}, {"id": "629", "uris": [{"http://zotero.org/users/14348568/items/CPQ8XW4M"}], "itemData": {"id": "629", "type": "article-journal"}, "abstract": "The first comprehensive overview of intra- and interspecific variation within the genus *Corvus* as well as first insights into the phylogenetic relationships of its species is presented. DNA sequences

of the mitochondrial control region were obtained from 34 of the 40 described species (including subspecies: 56 taxa, is native to the Indian subcontinent (Ryall, 2016). Due to its remarkable ability to colonize novel environments, the species is now established across much of the world, except for Antarctica (Ndimuligo *et al.*, 2022). Renowned for its aggressiveness and competitive dominance, the IHC has been implicated in biodiversity declines and associated socio-economic impacts globally (John & Kagembe, 2022; Kimario *et al.*, 2020).

The eradication, management, and control of the IHC remain major concerns in many countries, as its population continues to

increase at an unprecedented rate and its spread outpaces most invasive bird species. Commonly employed control measures, including poisoning, trapping, shooting, and nest destruction, are often prohibitively costly, labor-intensive, ineffective, or environmentally unsustainable (Archer, 2001; Nxele & Shivambu, 2018; Puttoo & Archer, 2004; Ryall, 2010; Suliman *et al.*, 2010) the Indian House Crow, *Corvus splendens*, has become a serious pest in many places from East London in the south to Djibouti in the north. House Crows have a major negative impact on indigenous biodiversity, depreciating native birds (especially newly hatched nestlings. Chemical control using agents such as Starlicide and α -chloralose, although frequently applied, not only kills crows but also results in high non-target mortality, thereby threatening avian biodiversity (Puttoo & Archer, 2004; Suliman *et al.*, 2010) Shri Lanka, Burma Malaysia and southern China. These species have spread over the past 100 years into inland and coastal areas of the Indian Ocean islands where, in the absence of natural predators, their population dramatically increased to become a pest of considerable importance. In the local context, there is scanty information on its introduction to Mauritius and it is quite possible that Mauritius has been exposed to these birds ever since ships sailed between southern Asia and Mauritius in the early 17th century, the bird being cared for on board by sailors and which flew to land on arrival of ships. In the past, different control strategies have been implemented so as to reduce the population of crows, namely, cage trapping, shooting and feeding bait treated with poisonous insecticide but none gave any promising result. Finally, the Ministry of Agriculture, Food Technology and Natural Resources resorted to the assistance of a local firm and an expert in crow to undertake a control programme using a new chemical known as Starlicide which has proven effective against several other pest avian species in some European and African countries. The control programme lasted for twelve weeks and was carried out in close collaboration with the technical staff of National Parks

and Conservation Service. The chemical Starlicide has proved to be effective and could be safely used to control these birds in the future.”,”DOI”:https://www.researchgate.net/publication/237457289_CONTROL_AND_OR_ERADICATION_OF_INDIAN_CROWS_Corvus_splendens_IN_MAURITIUS,”journalAbbreviation”:*AMAS*,”page”:*299-309*,”source”:*ResearchGate*,”title”:*Control and/ or Eradication of Indian Crows (Corvus splendens).*

Biological control is increasingly regarded as a more sustainable alternative, offering cost-effectiveness alongside ecological and environmental compatibility. However, successful implementation of biological strategies requires a clear understanding of the species’ breeding biology and its spatial variation across invaded regions. This review highlights the global variation in aspects of breeding biology (Fig. 1) of the IHC, providing insights to address critical knowledge gaps and to inform more effective, localized management and control strategies for this highly invasive species.

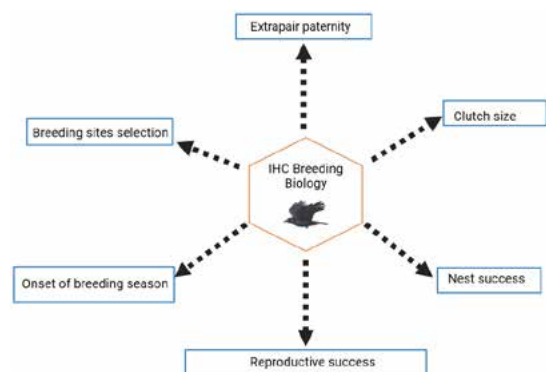


Figure 1: The aspects of IHC breeding biology (Author, 2025)

MATERIAL AND METHODS

This review included 75 relevant peer-reviewed journal articles that were identified through literature searches conducted in Google Scholar, EBSCOhost, and the Corvid Literature Database. Boolean operators were

applied to refine the search and maximize coverage. The primary search string combined species identifiers with breeding biology terms which include, (“*Indian House Crow*” OR “*Corvus splendens*” OR “*House Crow*” OR “*C. splendens*”) AND (“breeding biology” OR “reproductive biology” OR “breeding ecology” OR “breeding success” OR “clutch size” OR “nest success” OR “nesting” OR “fledgling success” OR “incubation period” OR “mating system” OR “extrapair paternity” OR “breeding behavior”). Additional queries were developed to capture studies addressing management and control strategies in relation to reproduction, including (“*Indian House Crow*” OR “*Corvus splendens*”) AND (“biological control” OR “pest control” OR “management” OR “eradication” OR “population control” OR “invasive species management”) AND (“breeding biology” OR “nesting ecology” OR “reproductive biology”). To provide a broader ecological context, supplementary searches on Corvids in general were conducted using terms such as (“*Corvids*” OR “*Corvus*” OR “*crows*”) AND (“breeding biology” OR “reproductive biology” OR “breeding ecology”) AND (“nesting success” OR “clutch size” OR “mating system”). This approach ensured comprehensive coverage of both species-specific and comparative literature relevant to the breeding biology of the IHC. The current review focused on identifying spatial variation in key breeding biology parameters of the IHC across its global range and the effectiveness and sustainability of the control measures. By synthesizing these patterns, the study aims to address existing knowledge gaps and provide critical insights to support more effective and context-specific control and management strategies for the study species.

RESULTS AND DISCUSSION

Aspects of breeding biology of the IHC Onset and ending of the breeding season

The timing of the breeding season in the IHC varies both locally and globally (Nahid *et al.*,

2019). These variations are largely influenced by climatic differences and seasonal shifts, particularly changes in temperature, as well as the availability of food resources and suitable nesting materials (Behrouzi-Rad, 2010; Grudinskaya *et al.*, 2022). Similar to other bird species, IHCs require sufficient food reserves prior to reproduction to enhance breeding success (Cuttriss *et al.*, 2015). *Thinornis rubricollis rubricollis*, which, in this part of its range, is a threatened obligate beach bird. Breeding sites had higher abundances, equivalent species richness, and different assemblages of invertebrate prey compared with absence sites. Assemblages at breeding sites were characterised by more isopods, and fewer beetles of the family Phycosecidae. Breeding habitat selection by plovers appears to be associated with selection for sites with more food, and any process that degrades food resources at a site (e.g. kelp harvesting or marine pollution events).

In Pakistan, the breeding season of the IHC varies across regions, occurring between August and September in Khyber Pakhtunkhwa, June and September in Mansehra, and May and September in the Pothwar region (Ali *et al.*, 2007; Awais *et al.*, 2015; Zamir *et al.*, 2022). In India, particularly in the northern part of the country, breeding extends from March to August (Allan & Davies, 2005). Likewise, in Junagadh, Gujarat, the season lasts from June to September (Radadia, 2013), while in some parts of India, the species breeds year-round (Roshnath & Sinu, 2021). On Kharg Island, Iran, breeding occurs from May to September (Behrouzi-Rad, 2010), with the peak across much of the Indian subcontinent generally falling between April and June (Ramakrishna *et al.*, 2012). In contrast, in Norway, the breeding season spans almost the entire year, from January to August (Nahid *et al.*, 2019). In Durban, South Africa, IHCs breed from October to January, most intensively between October and November (Allan & Davies, 2005); a similar pattern is observed in Mauritius (Feare & Mungroo, 1990). In Kenya, breeding begins in September and peaks from

November to December (Chongomwa, 2011; Ryall, 1992), closely mirroring the pattern in Dar es Salaam, Tanzania, where breeding occurs from September to January (Fry *et al.*, 2000; Kimario *et al.*, 2020). These similarities may reflect shared climatic conditions and overlapping seasonal cycles. On Zanzibar Island, Tanzania, breeding extends from October to March (Archer, 2001).

The observed pattern indicates that the dry season is the preferred breeding period for IHC across most regions. This trend may be attributed to the greater availability of nesting materials, such as dry sticks, twigs, and branches, which constitute the primary nest-building resources (Behrouzi-Rad, 2010), as well as the favorable sunny weather conditions that enhance egg incubation and nestlings' survival. Based on these findings, we recommend that IHC population control efforts be intensified during the dry season, when breeding activity peaks and individuals are actively searching for nesting materials and food. Similar recommendations were made by (Kimario *et al.*, 2020), who further emphasized the role of the gonadosomatic index in determining the onset of breeding, thereby improving the timing and effectiveness of IHC population control, management, and eradication strategies.

Breeding/Nesting Site Selection

Birds are generally selective in their choice of nesting and breeding sites, basing their preferences not on tree abundance but on specific structural characteristics (Han *et al.*, 2019). Nest-site selection plays a critical role in reproductive success and offspring survival (Fogarty *et al.*, 2017). The IHC typically prefers relatively tall, evergreen trees with large canopy volumes (≥ 3 m), high branch density, and greater trunk diameter (Kaur, 2020; Soh *et al.*, 2002; Sohi & Kler, 2017). The IHC typically prefers relatively tall, evergreen trees with large canopy volumes (≥ 3 m), high branch density, and greater trunk diameter (Kaur, 2020; Soh *et al.*, 2002; Sohi & Kler, 2017). The IHC typically prefers relatively tall, evergreen trees with large canopy volumes (≥ 3 m), high branch density, and greater trunk diameter (Kaur, 2020; Soh *et al.*, 2002; Sohi & Kler, 2017).

crow (*Corvus splendens*). Such features likely provide enhanced protection from predators and improved opportunities for locating food during the breeding season. In addition, IHCs show a tendency to nest in areas with high anthropogenic food availability (Behrouzi-Rad, 2010; John, 2024; Roshnath & Sinu, 2021). Colonial nesting has also been observed in some populations (Begum *et al.*, 2011). Both the preferred tree species (Table 1) and the nesting materials used exhibit spatial variation.

It has been observed that the IHC shows strong nesting preferences for tree species commonly found within its native range in the Indian subcontinent (Dharani, 2019). Across most invaded regions, six tree species are consistently favored: *Azadirachta indica*, *Peltophorumpterocarpum*, *Terminalia mantaly*, *Polyalthia longifolia*, *Pithecellobium dulce*, and *Mangifera indica*. These preferences have been documented in Durban (Allan & Davies, 2005), Mauritius (Feare & Mungroo, 1990), Mombasa (Chongomwa, 2011), Dar es Salaam (Shimba & Jonah, 2017), Singapore (Soh *et al.*, 2002) possibly reducing its breeding success, was the focus of our study. We compared the nest sites ($n = 30$, Pakistan (Awais *et al.*, 2015), Punjab (Kaur & Khera, 2020), and Kolkata, India (Dutta & Raut, 2015). This evidence suggests that while various attributes of nesting trees may influence nest-site selection, IHC exhibit a consistent affinity toward particular species. Nonetheless, in areas where preferred species are scarce, the crows readily utilize less-preferred trees, and in cases of further limitation, artificial poles and buildings serve as nesting alternatives (Benmazouz *et al.*, 2021; Noreen & Sultan, 2022) yet few taxonomic groups are remarkably successful at adapting to urban environments. We systematically surveyed the global literature on the effects of urbanization on species of family Corvidae (crows, choughs, jackdaws, jays, magpies, nutcrackers, ravens, rooks, treepies. Management strategies for reducing IHC populations should therefore incorporate habitat modification, particularly crown manipulation through pruning to reduce canopy volume,

which has been shown to discourage nesting and consequently lower crow abundance (Yong *et al.*, 2024) Africa, and Southeast Asia. Existing crow management measures include food limitation, habitat modification, and direct population control. The latter method provides a more immediate solution, particularly in nesting locations where crow attacks occur more frequently. Crow ladder traps are widely adopted to capture crows, but a thorough examination of the environmental conditions that optimize trapping efficacy is lacking. We assessed factors affecting crow trap efficacy in Singapore to better advise future deployment strategies of crow traps. We obtained data from 170 crow trapping operations including the identities of the contractors conducting the trapping operations, operation start date, crow density, intensity of bird feeding, various land use cover proportions, and an index to quantify the spatial-temporal proximity to previous trapping operations. We used a spatial gamma generalized linear mixed model (GLMM).

In Pakistan, Indian House Crow (IHC) nests are primarily constructed using twigs, along with discarded metallic wires, plastics, and plant materials such as grass, leaves, tendrils, rhizomes, wood, tree branches, feathers, and roots (Awais *et al.*, 2015; Noreen & Sultan, 2022). In India, similar patterns have been reported, with nests incorporating twigs, grasses, feathers, leaves, metal wires, thick plastic fragments, fine cloth, coir, and fibers (Ali *et al.*, 2011; Jadhav *et al.*, 2018). Remarkably, in Western Bengal, more than 90% of nesting material was composed of metals (Das & Kapoor, 2024) India. Among seven identified materials, six were anthropogenic, with metal waste comprising 90.90%. This finding highlights

the alarming occurrence of metal pollution and its potential impact on avian populations. The excessive use of anthropogenic materials in nest construction raises concerns about the negative consequences of metal and plastic pollution on birds. The study emphasizes the need for comprehensive waste management strategies, public awareness campaigns, and responsible urban planning to mitigate these harmful effects and ensure the wellbeing of urban bird communities.”,”container-title”:”Munis Entomology and Zoology Journal”,”DOI”:”https://www.munisentzool.org/”,”journalAbbreviation”:”Munis EntomologyandZoologyJournal”,”page”:”733-737”,”source”:”ResearchGate”,”title”:”A House Crow (*Corvus splendens*, a trend that poses significant risks to other avifauna due to the hazardous and unstable nature of such materials. In Dhaka, Bangladesh, however, nests are generally dominated by twigs, supplemented with other secondary materials (Begum *et al.*, 2011).

The IHCs can serve as bio-indicators of urban pollution, as the frequent incorporation of anthropogenic materials into their nests reflects the presence of solid waste in cities. This not only highlights environmental health risks for humans but also poses hazards to other bird species. Consequently, urban sanitation should be prioritized as a management measure to reduce the availability of nesting materials such as plastics, wires, and cloth. Moreover, a lack of nesting materials due to urban sanitation can lead to poor-quality nest construction, which can lead to reproductive failure (Awais *et al.*, 2015), thereby contributing to population control of IHCs.

Table 1: Preferred IHC tree species in different parts of the world

Tree species	Location	References
<i>Ficus bengalensis</i> , <i>Ficus religiosa</i> , <i>Ficus hispida</i> , <i>Ficus infectoria</i> , <i>Artocarpus heterophyllus</i> , <i>Mangifera indica</i> , <i>Spondis dulcis</i> , <i>Sygygium cumini</i> , <i>Sygygium samarangense</i> , <i>Psidium guajava</i> , <i>Eukalyptus alba</i> , <i>Madhuca longifolia</i> , <i>Mimusops elengi</i> , <i>Achras sapota</i> , <i>Annona squamosa</i> , <i>Polyalthia longifolia</i> , <i>Tamarandus indica</i> , <i>Caesalpinia pulcherrima</i> , <i>Delonix regia</i> , <i>Acacia catechu</i> , <i>Albizia lebbek</i> , <i>Butea monosperma</i> , <i>Dalbergia sissoo</i> , <i>Cassia fistula</i> , <i>Cassia tora</i> , <i>Sterculia foetida</i> , <i>Kleinhovia hospital</i> , <i>Magnolia grandiflora</i> , <i>Michelia champaca</i> , <i>Moringa pterygosperma</i> , <i>Aegele marmelos</i> , <i>Alstonia scholaris</i> , <i>Casurina equisetifolia</i> , <i>Terminalia arjuna</i> , <i>Tectona grandis</i> , <i>Azadirachta Indica</i> , <i>Ziziphus mauritiana</i> , <i>Diospyros glutinosa</i> , <i>Dillenia indica</i> , <i>Sterculia foetida</i> , <i>Anthocephalus cadamba</i> , <i>Euphorbia longan</i> , <i>Barringtonia acutangula</i> , <i>Hopea odorata</i> , <i>Lagerstroemia speciose</i> , <i>Bombax ceiba</i> and <i>Nyctanthes arbor-tristis</i>	Kolkata, India	(Dutta & Raut, 2015)
<i>Delonix regia</i> , <i>Thespesia populnea</i> , and <i>Azadirachta indica</i>	Southern India	(Ezhilarasi <i>et al.</i> , 2021)
<i>Acacia nilotica</i> , <i>Ficus religiosa</i> and <i>Mangifera indica</i>	Maharashtra State, India	(Jadhav <i>et al.</i> , 2018)
<i>Adhatoda vasica</i>	Northern Gujarat, India	(Gajera <i>et al.</i> , 2009).
<i>Dalbergia sissoo</i> , <i>Eucalyptus oblique</i> , <i>Syzygium cumini</i> , <i>Melia azedarach</i> , <i>Populus deltoids</i> , <i>Ficus religiosa</i> , <i>Saraca asoca</i> and <i>Azadirachta indica</i>	Punjab, India	(Kaur <i>et al.</i> , 2015)
<i>Palmae phoenix</i> , <i>Tamarix aphylla</i> , and <i>Eucalyptus spp</i>	Israel	(Yosef <i>et al.</i> , 2019)
<i>Artocarpus heterophyllus</i> , <i>Alstonia scholaris</i> , <i>Casuarina sp</i> , <i>Polyalthia longifolia</i> and <i>Tectona grandis</i>	Bangladesh	(Jahan <i>et al.</i> , 2018).
<i>Melia azedarach</i> , <i>Pinus roxburghii</i> , <i>Platanus orientalis</i> , <i>Eucalyptus spp</i> , <i>Populus euphratica</i> , <i>Populus nigra</i> and <i>Morus nigra</i>	Pakistan	(Anjum <i>et al.</i> , 2021; Awais <i>et al.</i> , 2015; Noreen & Sultan, 2022).
<i>Peltophorum pterocarpum</i> and <i>Samanea saman</i>	Singapore	(Soh <i>et al.</i> , 2002)possibly reducing its breeding success, was the focus of our study. We compared the nest sites (n ^ 30.
<i>Mangifera indica</i> and <i>Azadirachta indica</i>	Mombasa, Kenya	(Chongomwa, 2011).
<i>Mangifera indica</i> , <i>Araucaria sp.</i> , <i>Pinus sp.</i> , <i>Casuarina equisetifolia</i> , <i>Persea Americana</i> , <i>Artocarpus heterophyllus</i> , <i>Syzygium cumini</i> , <i>Ficus sp.</i> , <i>Tibouchina sp.</i> , <i>Albizia adianthifolia</i> , <i>Podocarpus henkelii</i> and <i>Trichilia dreagena</i>	Durban, South Africa	(Allan & Davies, 2005).

Tree species	Location	References
<i>Senna siamea</i> , <i>Azadirachta indica</i> , <i>Polyalthia longifolia</i> , <i>Terminalia mantaly</i> , <i>Muntingia calabura</i> , <i>Cordia sebestena</i> , <i>Peltophorum pterocarpum</i> , <i>Pithecellobium dulce</i> , <i>Leucaena leucocephala</i> , <i>Carica papaya</i> , <i>Delonix regia</i> , <i>Mangifera indica</i> , <i>Trichilia emetica</i> , <i>Cananga odorata</i> , <i>Cocos nucifera</i> , <i>Eucalyptus spp</i> , <i>Casuarina cunninghamiana</i> , <i>Ficus thonningii</i> , <i>Millingtonia hortensis</i> , <i>Roystonea regia</i> , <i>Terminalia superba</i> , <i>Acacia longifolia</i> , <i>Ceiba pentandra</i> , <i>Ficus benjamina</i> , <i>Hura crepitans</i> , <i>Thevetia peruviana</i> , <i>Senna spectabilis</i> , <i>Albizia lebbeck</i> , <i>Tamarindus indica</i> , <i>Terminalia catappa</i> , <i>Annona muricata</i> , <i>Citrus limon</i> , <i>Dichrostachys cinerea</i> , <i>Enterolobium contortisiliquum</i> , <i>Melia azedarach</i> , <i>Persea Americana</i> , <i>Psidium guajava</i> , <i>Pteleopsis myrtifolia</i> , <i>Sesbania sesban</i> , <i>Spathodea campanulata</i> , <i>Syzigium cummnii</i> , <i>Tabernaemontana divaricate</i> , <i>Mangifera indica</i> and <i>Pithecellobium dulce</i>	Dar es Salaam, Tanzania	(John, 2024; Shimba & Jonah, 2017)
<i>Ficus bengalensis</i> , <i>Eucalyptus spp</i> and <i>Ziziphus aucherii</i>	Kharg Island, Persian Gulf	(Behrouzi-Rad, 2010)
<i>Mangifera indica</i> and <i>Ficus religiosa</i>	Parsa, Nepal	(Ranjan & Kushwaha, 2013)

Extrapair paternity

Extrapair paternity (EPP) refers to a mating system in which a female mates with one or more males outside her social pair, often resulting in extrapair fertilisation (Akçay & Roughgarden, 2007). EPP is widespread, occurring in over 70% of socially monogamous birds, particularly passerines (Bolopo *et al.*, 2020; Moreno *et al.*, 2015; Overeem *et al.*, 2014), but is comparatively rare in non-passerines, especially raptors (Wan *et al.*, 2013). Several hypotheses have been proposed to explain its adaptive significance, including fertility assurance, acquisition of “good genes,” genetic compatibility, and enhanced offspring genetic diversity (Wan *et al.*, 2013). Variation in EPP both within and across species is attributed to factors such as breeding synchrony, population density, life-history traits, genetic diversity (inbreeding avoidance), pathogen pressures, and geographical location (Brouwer *et al.*, 2017; Valcu *et al.*, 2021) where offspring are sired by a male other than the social male, varies enormously both within and

among species. Trying to explain this variation has proved difficult because the majority of the interspecific variation is phylogenetically based. Ideally, variation in EPP should be investigated in closely related species, but clades with sufficient variation are rare. We present a comprehensive multifactorial test to explain variation in EPP among individuals in 20 populations of nine species over 89 years from a single bird family (Maluridae). However, current knowledge remains limited: phylogenetic history explains only about 50% of the observed variation (Brouwer *et al.*, 2017), and evidence indicates that EPP frequency can also vary with latitude and altitude (Bonier *et al.*, 2013; Brouwer & Griffith, 2019; Valcu *et al.*, 2021) ecological traits, and population demography. The siring of offspring by an extrapair male, or extrapair paternity (EPP).

Indian House Crows (IHCs), like most members of the *Corvus* genus, are socially monogamous, with pair bonds that typically persist across

successive breeding seasons (Awais *et al.*, 2015; Kaur & Khera, 2020). Although studies specifically addressing extrapair paternity (EPP) in IHCs are scarce, evidence from related corvids, including American crows, Steller's jays, carrion crows, and ravens, demonstrates that EPP does occur within the family (Knief *et al.*, 2020; Overeem *et al.*, 2014; Townsend, 2009) the number of mating opportunities with a conspecific can be limited. As a consequence, individuals may mate with a heterospecific (social). Given the reproductive biology of IHCs, the potential for EPP is high. The time lag between insemination and fertilisation, which may span several weeks, creates an opportunity for sperm competition among multiple males (Kimario *et al.*, 2020; Wan *et al.*, 2013). Investigating the presence and spatial variation of EPP in IHCs is therefore essential. Such insights could clarify how socially monogamous species overcome demographic and genetic bottlenecks, facilitating establishment in novel environments, while also informing strategies for population control and management of this invasive species.

Clutch size

Clutch size i.e. the number of eggs laid per nest, shows spatial variation across different regions invaded by the IHC. (Patankar *et al.*, 2021) in order to review literature on traits that influence responses to urbanization. We review 226 papers that were published between 1979 and 2020, and aggregate information on five major groups of traits that have been widely studied: ecological traits, life history, physiology, behavior and genetic traits. Some robust findings on trait changes in individual species as well as bird communities emerge. A lack of specific food and shelter resources has led to the urban bird community being dominated by generalist species, while specialist species show decline. Urbanized birds differ in the behavioral traits, showing an increase in song frequency and amplitude, and bolder behavior, as compared to rural populations of the same species. Differential food resources and predatory pressure results in changes in life

history traits, including prolonged breeding duration, and increases in clutch and brood size to compensate for lower survival. Other species-specific changes include changes in hormonal state, body state, and genetic differences from rural populations. We identify gaps in research, with a paucity of studies in tropical cities and a need for greater examination of traits that influence persistence and success in native vs. introduced populations.”,”container-title”:”Land”,”DOI”:”10.3390/land10020092”,”ISSN”:”2073-445X”,”issue”:”2”,”journalAbbreviation”:”Land”,”language”:”en”,”license”:”https://creativecommons.org/licenses/by/4.0/”,”page”:”92”,”source”:”DOI.org (Crossref reported difference in IHC clutch size between rural and urban environments, with urban populations exhibiting larger clutch sizes. This trend is likely an adaptive response to the elevated risks associated with urban settings, such as traffic-related mortality, whereby producing more eggs increases the probability of successful recruitment and compensates for losses caused by urbanisation pressures.

In Tamil Nadu, India, the mean clutch size of Indian House Crows (IHCs) was reported as 4.5 eggs (4–5 eggs), with eggs typically pale blue-green and marked with brown speckles and streaks (Ali *et al.*, 2011). Elsewhere in India, a mean clutch size of 4.41 was recorded (Sirsat, 2022), and across the Indian subcontinent, clutch size generally ranges between 4 and 5 eggs (Ramakrishna *et al.*, 2012). Outside India, the mean clutch size has been reported as 4.1 in both Dar es Salaam, Tanzania (Shimba & Jonah, 2017), and Merewent, South Africa (Allan & Davies, 2005). In Pakistan, the mean clutch size was 4.0 (1–6 eggs) in Mansehra (Awais *et al.*, 2015) and 3.8 in Islamabad-Rawalpindi (Ali *et al.*, 2007). Similar values were reported on Kharg Island, Persia, and in Mauritius, where the mean clutch size was 4 (Behrouzi-Rad, 2010; Puttoo & Archer, 2004), closely resembling the 3.9 reported in Mombasa, Kenya ((Ryall, 1992). In Zanzibar, the mean clutch size was slightly lower at 3.53 (Archer, 2001), while in Parsa District, Nepal,

it was recorded as 4.4 (4–5 eggs) (Ranjan & Kushwaha, 2013).

Based on reported studies, the clutch size of IHCs generally ranges between 3.5 and 4.5 eggs across most regions. Understanding this reproductive parameter is essential for developing effective strategies for the control and management of IHC populations. We suggest deploying the knowledge of clutch size to perform population modeling and forecasting, allowing managers to estimate potential population increase and evaluate the intensity of control efforts required to achieve population suppression.

Nest success

Nest success is defined as the proportion of eggs that successfully hatch relative to the total number of eggs laid (Armstrong *et al.*, 2002; Awais *et al.*, 2015) the proportion of clutches resulting in one or more fledglings, is a key indicator for assessing the effect of management on bird populations. However, the figures reported for New Zealand populations are usually “apparent nest success”, the number of successful nests divided by the total number found. Apparent nest success invariably overestimates the true success rate, and the degree of bias depends on the population and monitoring regime. Consequently, apparent nest success rates cannot be reliably compared. We used Stanley’s (2000. Geographic variation in IHC nest success is influenced by multiple factors, including tree species, crown cover, and habitat characteristics (Shimba & Jonah, 2017; Yosef *et al.*, 2019). Urban populations typically exhibit higher nest success than rural populations, likely due to the greater availability of food resources in urban environments.

In southern India, IHC nest success was reported at 50%, with open nests constructed primarily of twigs (Ezhilarasi *et al.*, 2021). In Pakistan, nest success was relatively low (46.7%) due to brood parasitism by the Indian Koel (*Eudynamis scolopacea*), which is known to significantly reduce reproductive

success (Ali *et al.*, 2007; Downs & Hart, 2020). In Mansehra, Pakistan, nest success was slightly higher at 55.1%, corresponding to an average of 2.2 hatchlings per nest (Awais *et al.*, 2015). In Dar es Salaam, Tanzania, nest success varied with canopy cover, with a mean value of 58.4% (Shimba & Jonah, 2017). Nest success serves as a key indicator of reproductive output, reflecting the number of new individuals added to the population each breeding season. Consequently, effective control and management strategies for IHC should consider local nest success rates to target interventions more efficiently.

Reproductive success

Breeding success in songbirds is defined as “full-season productivity,” or the number of young raised to independence within a breeding season (Streby *et al.*, 2014). For IHCs, breeding success has been observed to be higher in urban areas compared to rural areas (Benmazouz *et al.*, 2021) yet few taxonomic groups are remarkably successful at adapting to urban environments. We systematically surveyed the global literature on the effects of urbanization on species of family Corvidae (crows, choughs, jackdaws, jays, magpies, nutcrackers, ravens, rooks, treepies. Factors enhancing reproductive success in corvids include inherited traits such as laying large clutches, early breeding, efficient nest defense and food provisioning, and the ability to identify and secure territories with abundant food, minimal predation, and suitable nesting conditions (Benmazouz *et al.*, 2021; Loman, 1977; Smart *et al.*, 2024) yet few taxonomic groups are remarkably successful at adapting to urban environments. We systematically surveyed the global literature on the effects of urbanization on species of family Corvidae (crows, choughs, jackdaws, jays, magpies, nutcrackers, ravens, rooks, treepies. Conversely, breeding failure in many bird species is driven by nest predation, food scarcity, suboptimal habitat structure, disease and parasites, adverse weather, and poor nest construction (Awais *et al.*, 2015; Chamberlain *et al.*, 2009; Smart *et al.*, 2024).

Studies specifically quantifying IHC reproductive success are limited, with most focusing on nest success. For example, (Awais *et al.*, 2015) reported a mean of 1.44 fledglings per nest, corresponding to 69% reproductive success, while (Ranjan & Kushwaha, 2013) observed 81.82% reproductive success in Parsa District, Nepal. Accurate data on reproductive success are therefore essential for planning effective population control and management strategies for IHCs.

Control measures

Due to their significant negative socio-ecological and economic impacts, IHCs have been subjected to eradication and control programmes in several countries, including Tanzania, South Africa, Mauritius, Kenya, Yemen, and Singapore. The two primary methods commonly employed are physical or mechanical control (e.g., trapping and shooting) and chemical control (poisoning) (Puttoo & Archer, 2004; Suliman *et al.*, 2010) Shri Lanka, Burma Malaysia and southern China. These species have spread over the past 100 years into inland and coastal areas of the Indian Ocean islands where, in the absence of natural predators, their population dramatically increased to become a pest of considerable importance. In the local context, there is scanty information on its introduction to Mauritius and it is quite possible that Mauritius has been exposed to these birds ever since ships sailed between southern Asia and Mauritius in the early 17th century, the bird being cared for on board by sailors and which flew to land on arrival of ships. In the past, different control strategies have been implemented so as to reduce the population of crows, namely, cage trapping, shooting and feeding bait treated with poisonous insecticide but none gave any promising result. Finally, the Ministry of Agriculture, Food Technology and Natural Resources resorted to the assistance of a local firm and an expert in crow to undertake a control programme using a new chemical known as Starlicide which has proven effective against several other pest avian species in some European and African countries. The control programme

lasted for twelve weeks and was carried out in close collaboration with the technical staff of National Parks and Conservation Service. The chemical Starlicide has proved to be effective and could be safely used to control these birds in the future.”,”DOI”.”https://www.researchgate.net/publication/237457289_CONTROL_AND_OR_ERADICATION_OF_INDIAN_CROWS_Corvus_splendens_IN_MAURITIUS”,”journalAbbreviation”.”AMAS”,”page”.”299-309”,”source”.”ResearchGate”,”title”.”Control and/ or Eradication of Indian Crows (Corvus splendens. In Socotra, Yemen, physical control proved highly effective, with successful eradication achieved through shooting adult crows and removing nestlings (Suliman *et al.*, 2010) affecting more than 25 nations around the Indian Ocean, Arabian Peninsula, and Southeast Asia. It causes problems for development, public hygiene, biodiversity, tourism, and traffic. House crows arrived in 1995 on Socotra Island, Yemen Republic, coming from mainland Yemen by ship. Socotra Island is a UNESCO World Heritage Site, having a unique ecosystem with a larger number of endemic species. Ongoing management did not achieve eradication of the species for more than a decade; however, it kept numbers of the breeding population low. The last 13 birds were finally eradicated in April 2009, thanks to techniques and experienced personnel from abroad and support by the Small Grant Programme of the Global Environmental Facility. More vigilance in dealing with this species, as well as a better transfer of international knowledge to local managers, are required to tackle problems caused by invasive house crows in the affected regions.”,”container-title”.”Proceedings of the Vertebrate Pest Conference”,”DOI”.”10.5070/V424110552”,”ISSN”.”2641-273X”,”journalAbbreviation”.”vertebrate_pest_conference”,”language”.”en”,”page”.”257-262”,”source”.”DOI.org (Crossref. Similar success with shooting was reported in Australia (Ryall, 2010; Ryall & Ryall, 2002). In Mauritius, poisoning with Starlicide and α -chlordane substantially reduced IHC populations (Puttoo & Archer, 2004).

In East Africa, IHC control programmes have primarily relied on physical methods, including trapping and egg destruction during the breeding season, and occasionally on poisoning with Starlicide. These approaches achieved partial success, significantly reducing IHC populations in Djibouti, Mombasa, Malindi, and Zanzibar (Archer, 2001b). However, complete eradication was hindered by inadequate funding, poor coordination, lack of consistency, and limited collective effort in implementing control measures (Archer, 2001b). In Zanzibar, for instance, control efforts ultimately failed despite expenditures exceeding US\$1.5 million (Evans *et al.*, 2023) and associated with costs due to the damage caused and actions required to manage them. We synthesised global cost data to identify variation across regions, types of impact, and alien bird species. Costs amount to US\$3.6 billion, but this is likely a vast underestimate. Costs are low compared to other taxonomic groups assessed using the same methods; despite underreporting, alien birds are likely to be less damaging and easier to manage than many other alien taxa. Research to understand why this is the case could inform measures to reduce costs associated with biological invasions. Costs are biased towards high-income regions and damaging environmental impacts, particularly on islands. Most costs on islands result from actions to protect biodiversity and tend to be low and one-off (temporary). To date, no biological methods have been formally implemented for the control of IHCs. However, brood parasitism by the Great Spotted Cuckoo (*Clamator glandarius*) in Israel and the Indian Koel (*Eudynamys scolopaceus*) in India has been documented, positioning these species as natural enemies of IHCs (Downs & Hart, 2020; Jindal & Kumar, 2023). While such interactions highlight the potential for biological control, they also carry the risk of introducing new ecological problems.

Currently, IHC management relies largely on physical methods and poisoning.

Although these approaches have shown some effectiveness, they are costly, ecologically disruptive, and environmentally unsustainable. A deeper understanding of the species' biology is therefore essential to guide the development of long-term, ecologically sound, and environmentally friendly control strategies.

CONCLUSION

The present study highlights considerable spatial variation in the breeding biology of IHC both locally and internationally. The timing of the breeding season varies by location, ranging from seasonal patterns in some regions to year-round breeding in others. Nest-site selection is determined primarily by tree characteristics rather than abundance, and both preferred tree species and nesting materials differ geographically. Globally, the most favored tree species are those native to the IHC's original range. Clutch sizes also vary across regions but generally fall between 3.5 and 4.5 eggs. Nest success shows substantial geographic variation, with brood parasitism significantly reducing reproductive output in some areas. Despite these differences, data on reproductive success and extrapair paternity remain limited, underscoring the need for further research. Overall, variation in IHC breeding biology appears to be influenced by local climate, habitat characteristics, and urbanisation levels. These spatial differences emphasize the importance of implementing targeted, site-specific strategies for the control, management, and potential eradication of IHC populations worldwide.

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MAMMAL DIVERSITY AND DISTRIBUTION FOR ENHANCING VISITORS EXPERIENCE AND CONSERVATION IN RUAHA NATIONAL PARK

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ABSTRACT

Ruaha National Park (RUNAPA) is among the renowned national parks in southern Tanzania inhabited by charismatic and megafauna species. However, RUNAPA is among the less visited protected areas, perhaps due to less information on wildlife status, distribution and other attractions found therein. Using aerial census data from 1990 to 2020 and ground surveys data from October to December 2021 collected by the Tanzania Wildlife Research Institute (TAWIRI). We determined; the species richness, evenness and diversity of medium to large-sized mammals in RUNAPA; the spatial-temporal distribution of mammals in RUNAPA for the past 30 years; and spatially link mammal distribution to the existing park roads network. A total of 29 medium to large mammal species were directly observed during the road transect and encountered signs of 7 species during the walking transect. The 30 years (1990 – 2020) aerial data indicated a skewed distribution of browsers, grazers and mixed feeders to the northern and central part of the park during the 1990 to 2010, and extended to the southern part of the park during the 2010 – 2020 census period. The south-east and south-west of the park had wildlife species but do not have all weather roads, implying that they are inaccessible during the wet season. Medium to large mammals are evenly distributed in RUNAPA, however, the limited access to the south-east and south-west of the park could negatively impact the parks tourism potential and consequently the country's economy. Investing in road infrastructures would support sustainable tourism in places which had seasonal roads.

Keywords: Mammals, spatio-temporal distribution, tourism, Southern Tanzania.

INTRODUCTION

Mammals play a crucial role in ecosystems and food chains, their diversity and interactions with other species contribute to the ecological balance and ecosystem's health and stability (Arbieu *et al.*, 2018; López-Arévalo *et al.*, 2011). Their diverse role includes the provision of ecosystem services such as nutrient cycling, energy flow, the exertion of bottom-up and top-down processes and seed dispersal, pollination, and controlling pest populations (Korkmazhan & Dunn, 2022; Lacher *et al.*, 2019; Tochigi *et al.*, 2022); as well as hold

cultural significance and recreational value (Boeyens & Ryst, 2014; de Melo *et al.*, 2014); The diverse mammal populations, behaviors, life histories, and adaptations provide engaging learning experiences for visitors, particularly students and nature enthusiasts, contributing to their understanding and appreciation of nature (Arbieu *et al.*, 2018; Curtin, 2009). Understanding mammal diversity and distribution therefore not only provides insights into ecological processes but can also enhance and boost tourists' experience, prioritize conservation efforts, and allow for adaptive management strategies (Hayes & Schradin,

2017; Krebs, 2020; Pédarros *et al.*, 2020).

Mammal species' presence, abundance, and distributions in different ecosystems are essentially attributed to the availability of habitat, predators, forage, invasive species and water resources, landscape features, and anthropogenic pressure (Barnes & Douglas-Hamilton, 1982; Dickman, 2008; Lewis *et al.*, 2017; STEP, 2016). The ideal physical and non-physical environment that attracts the inhabitation and distribution of mammals influences the ecological functioning of any ecosystem at large (Kashaigili *et al.*, 2005). Any reduced forage (quantity and quality) and habitats for a particular species at space and temporal scales strongly influence the dynamics of the species distribution patterns (Arbieu *et al.*, 2018; Barnes & Douglas-Hamilton, 1982). Understanding the spatial and temporal distribution of mammals is therefore among the fundamental ecological attributes for appropriate protected area management being conservation, antipoaching patrols, or tourism (Hockings *et al.*, 2004; Lewis *et al.*, 2017; Rija & Kideghesho, 2020).

Ruaha National Park (RUNAPA) is among the renowned national parks in southern Tanzania for its biodiversity resources (Barnes & Douglas-Hamilton, 1982; Jasson & Kiwango, 2021; Stommel, Hofer, & East, 2016). It is inhabited by a number of charismatic and megafauna such as the big five (lion - *Panthera leo*; leopard - *Panthera pardus*, elephant - *Loxodonta Africana* and buffalo - *Syncerus caffer*) (Dickman, 2008; Stommel, Hofer, Grobbel, *et al.*, 2016) which are all considered the main attractor of tourists to most sub-Saharan African protected areas. Other key wildlife attractions include medium-sized and large mammals such as, Kudu (*Tragelaphus spp*), roan antelope (*Hippotragus equinus*), giraffe (*Giraffa camelopardalis*), impala (*Aepycerous melampus*), hippopotamus (*Hippopotamus amphibius*), zebra (*Equus burchellii*), wildebeest- (*Connochaetes taurinus*), puku (*Kobus vardonii*) and African wild dog (*Lycaon pictus*) (Dickman, 2008; TAWIRI, 2015).

Besides having most of the attractions similar to those found in the northern tourist circuits of the country, RUNAPA is among the less visited protected areas (Simbaya, 2022) (Caballero, 2014), perhaps due to less information on wildlife status, occupancy and other attractions found therein. Moreover, inadequately linked information on the distribution and diversity of mammals along game viewing circuits may contribute to fewer sightings and hence, less visitor's satisfaction and experience.

Several studies have been conducted in RUNAPA exploring its biodiversity potential; birds (Jasson & Kiwango, 2021), single species movements and habitat use (Roug *et al.*, 2020), and small mammals (Stanley *et al.*, 2015), however, distribution of medium to large mammals is less updated (Barnes & Douglas-Hamilton, 1982; Norton & Griffiths, 1975) given the increased deterioration of water quantity and quality (Epaphras *et al.*, 2008; Stommel *et al.*, 2016) and the extension of the Ihefu section (Zia *et al.*, 2011). This study seeks to contribute to the tourism information for the southern part of the country by exploring the distribution of medium to large mammals in RUNAPA and advice on an integrated conservation plan that aligns with visitor management strategies, ensuring conservation and positive visitor experiences coexist. Specifically, the study aims to; i) determine the species richness, evenness, and diversity of medium to large-sized mammals in different habitats of RUNAPA; ii) determine the spatio-temporal distribution of mammals in the RUNAPA for the past 30 years; and iii) develop spatial maps illustrating the distribution of mammal species to identify key areas for conservation and visitor experiences.

MATERIALS AND METHODS

Study area description

Located between southern Tanzania (7°30'S and 35°00'E), the Ruaha National Park (20,226 km², Fig.1) is the second largest National Park following the recently gazetted Nyerere National Park in 2019, having an area of 30,000

km² (formerly part of Selous Game Reserve). The park encompasses the Great Ruaha River that maintains wildlife into the park boundaries harboring a large number of hippopotamus (*Hippopotamus amphibius*), it also harbors East Africa's largest elephant population and well known for its iconic Greater and Lesser Kudu, (*Tragelaphus strepsiceros* and *Tragelaphus imberbis*) respectively (Dickman, 2008). The park is also renowned for a good number of endangered wild dogs (*Lycaon pictus*) and over 500 bird species including, the Ruaha hornbill (*Tockus ruahae*) and Ruaha white-headed black chat (*Myrmecocichla collaris*).

The Park lies on an ecotone of miombo woodland of Commiphora-Combretum woodland and Acacia (Barnes, 1983), whereby, the western is

covered by deciduous Brachystegia (miombo) woodlands and Drypetes gerrardii; on the eastern Commiphora-Combretum woodland and bushland, while the north-eastern corner is dominated by Acacia bushed grassland (Mtahiko, 2004). Rainfall is mainly from November to April receiving up to 1600mm per annum and the mean annual temperature varies from 18°C at higher altitudes to 28°C in lower ranges (Kashaigili *et al.*, 2005; Mtahiko, 2004). In this study, the survey area was divided into four (5) zones namely; Central (Chaugigima, Mdonya, Msembe, Kalenga Kelu, Ruaha river and Wio); Eastern (Lunda and Madogoro); northern (Mpululu and Mapenza); southern (Majimoto, Ulanga, Ng'iriama, Madundasi, Mkoji, Makwaru and Ilambitari) and Western (Jongomero and Isunkaviola).

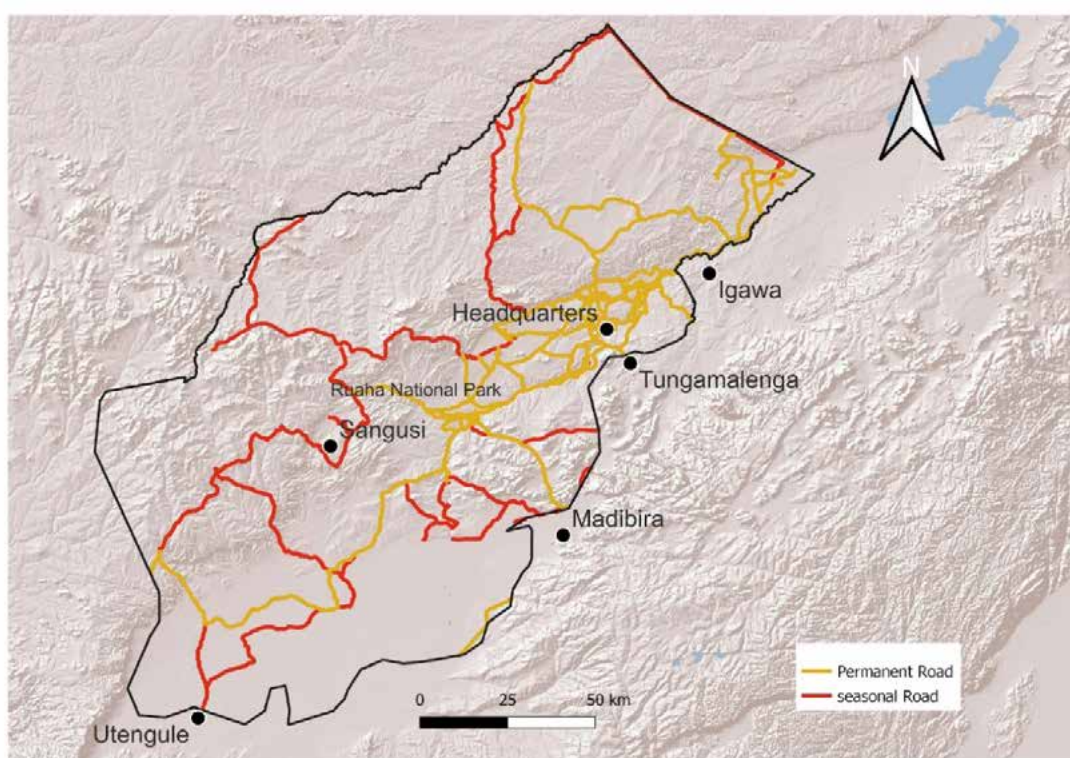


Figure 1: Ruaha National Park in the southern Tanzania

Study design

The study used the aerial census data collected by the Tanzania Wildlife Research Institute (TAWIRI) from 1990 to 2020 to determine the distribution of mammals in the park. These data

were further complimented by ground surveys (vehicle and walking transects) conducted by TAWIRI from October to December 2021 employing vehicle and walking transect surveys (Arbieu *et al.*, 2018; Gaidet-Drapier *et al.*, 2006).

Aerial surveys

The aerial census was conducted by TAWIRI (TAWIRI, 2015) following the systematic reconnaissance flight (SRF) (Norton-Griffiths, 1978), whereby transects were spaced 5km apart with variable orientations (North-South; East-West, and Northwest-Southeast) due to the nature of the terrain, ecological gradient and aiming at maximizing the number of samples or to allow for safe flying in difficult terrain.

Vehicle transects

A total of 5 road transects traversing the five protection zones in RUNAPA were selected for this survey. The central zone transects covered 75km; the eastern zone covered 35km; the northern zone covered 40km; the southern zone covered 60km; and the western zone covered 45km (Table S1). To maximize observations vehicle was driven at a speed of 20-30 km/hr with two observers on the right and left sides of the vehicle observing mammals within 250m from each side. Wildlife observations were done at a length of 5km and separated by a length of 2km, where no records were made (Figure 2).

Walking transects

The purpose of the walking transect was to explore the indirect mammal observations by the use of signs and tracks of species that probably were not captured directly using the road transect. These transects were conducted after the end of the 5km road transect observation to a distance of one kilometre each, a total of 51km were surveyed (Table S1). They run perpendicular from the main road transect alternating left or right of the road transect (Fig. 2).

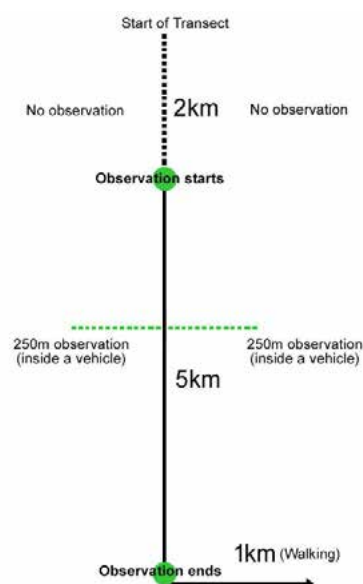


Figure 2: Layout of ground survey transect used for direct and indirect observations of the medium-sized to large mammals in Ruaha National Park.

Habitat was classified as open or dense woodland, bushland, thickets, shrubland, or grassland. The open habitat was referred to as any habitat where the maximum visual distance was at least 500 meters, while dense habitat referred to any habitat where the maximum visual distance was about 100 meters or less. Woodland consisted of woody vegetation taller than 10 meters, bushland between 6 and 10 meters, and shrubland less than 6 meters, while grassland consisted of only grass and herbaceous vegetation. All data were collected during the dry season which is also the peak time for tourism activities (high season) and on the game viewing and patrol circuits. Statistical analysis was carried out using different packages in R version 4.1.3 (R Core Team, 2022).

Data collection

Aerial surveys

We obtained aerial census data sets for the past 30 years from TAWIRI which included information on species name, group size, location (latitude and longitude), date, and time.

Vehicle and walking transect

For each sighting, the observers recorded the species name, group size, location (latitude and longitude), date, and time as well as the vegetation type. In addition, all medium to large mammal species signs observed during the walking transects were recorded.

Data analysis

Statistical analyses aimed to determine how the species abundance, richness, diversity, and evenness of medium to large-sized mammals vary in different habitats of RUNAPA and their interaction. Data from the ground counts of the observed mammals in RUNAPA was tested for normality using the Shapiro-Wilk test ($P \leq 0.05$) and log-transformed to reduce heteroscedasticity. Species abundance, richness (Menhinick's Index), diversity (Shannon index), and evenness of the observed mammals were determined for each zone and habitat type. One-way analysis of variance followed by multiple comparisons was performed at a 5% confidence level to compare means across and between different zones and habitats. To map

the distribution of medium to large mammals in RUNAPA, the medium to large mammals were categorized into browsers (giraffe and kudu (Greater & Lesser)), grazers (buffalo puku, hippopotamus, wildebeest and zebra) and mixed feeders (elephant, impala and roan antelope) (Estes & Otte, 2012). Distributions were mapped via QGIS 3.28 using both aerial and ground survey data. To identify key areas for conservation and visitor experiences, existing Park Road networks were mapped using QGIS 3.28, and the data collected during the ground survey were overlaid to assess the distribution along this network.

RESULTS

A total of thirty-seven medium to large mammal species were recorded using both direct and indirect direct observation. Out of the directly observed mammal species, the most abundant species were impala, giraffe, elephant, and Kudu (Lesser and Greater) (Table S2).

Species richness, diversity, and evenness of medium to large-sized mammals in different zones and habitats of RUNAPA

Across the five zones of the study area, the direct observations recorded relatively similar species richness and diversity (Figure 3 a & b, Table S3), species evenness differed significantly between the northern and eastern zones ($P = 0.016$) but did not differ between other zones (Fig 3 c, Table S3).

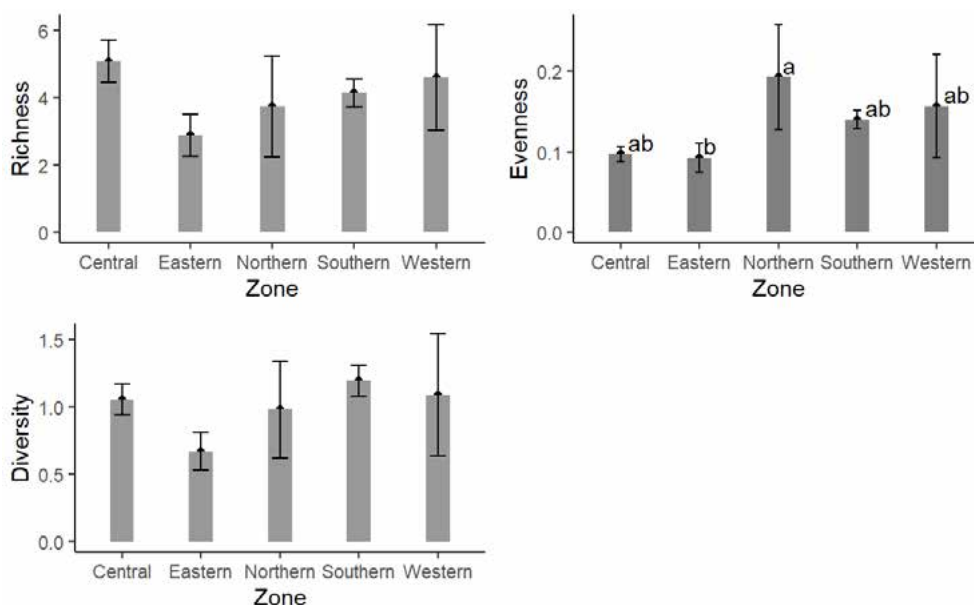


Figure.3: Species richness (a) diversity (b) and evenness (c) as directly observed in the five zones of Ruaha National Park.

Similarly, the indirect observations showed that the five zones had relatively similar species richness, diversity and evenness (Fig. 4 a, b, and c, Table S4).

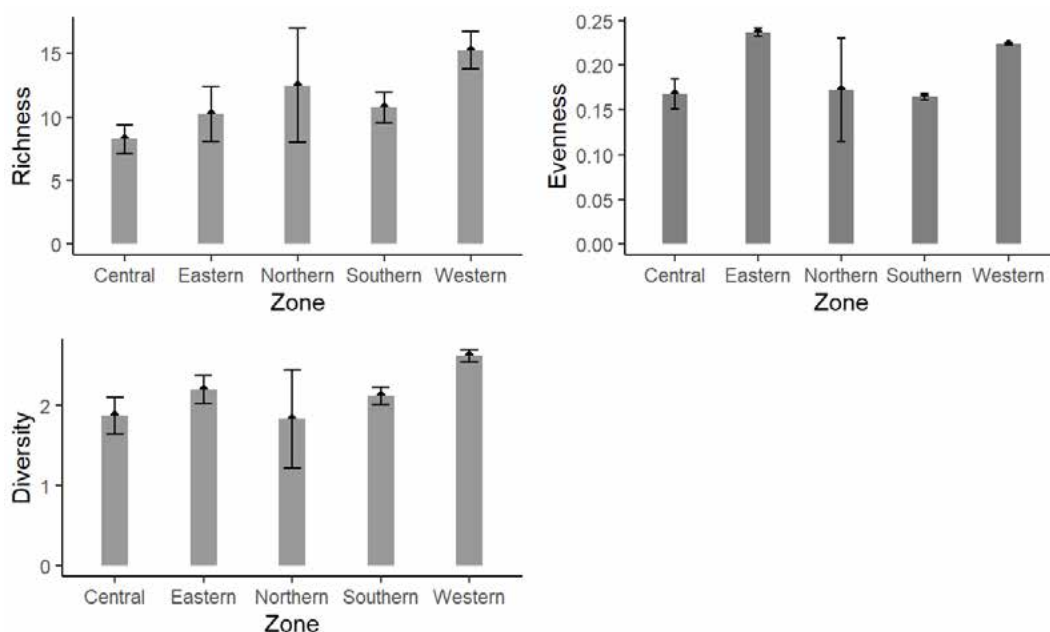


Figure . 4: Species richness (a), diversity (b) and evenness (b) as indirectly encountered in the five zones of Ruaha National Park.

Mammal species richness and diversity did not show a significant difference across the different habitats of RUNAPA, however species evenness significantly differed across habitat types ($P=0.046$) (Fig 5 a, b & c, Table S5).

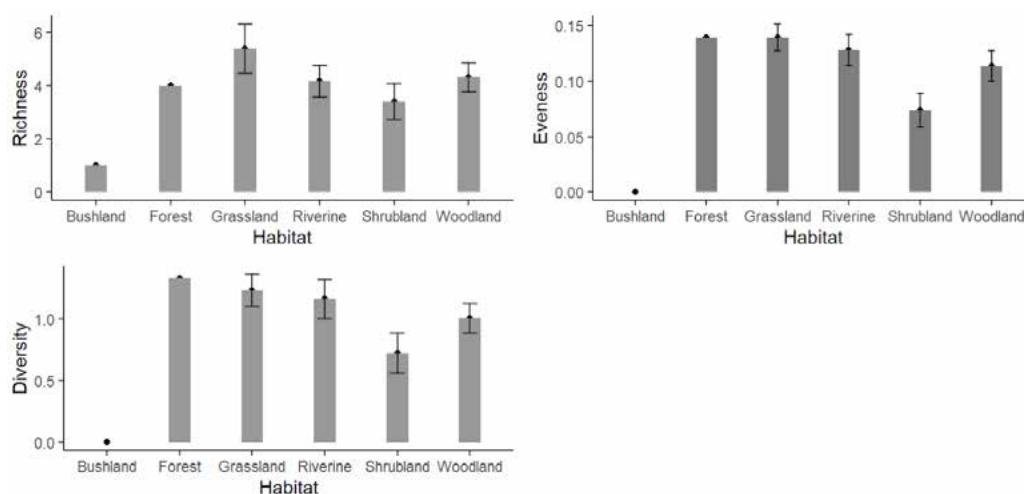


Figure 5: Species richness (a) diversity (b) and evenness (c) as directly observed in the different habitats of Ruaha National Park.

The indirect observation showed a significant difference in species richness, diversity and evenness across habitats (Fig. 6 a, b, & c, Table S6)

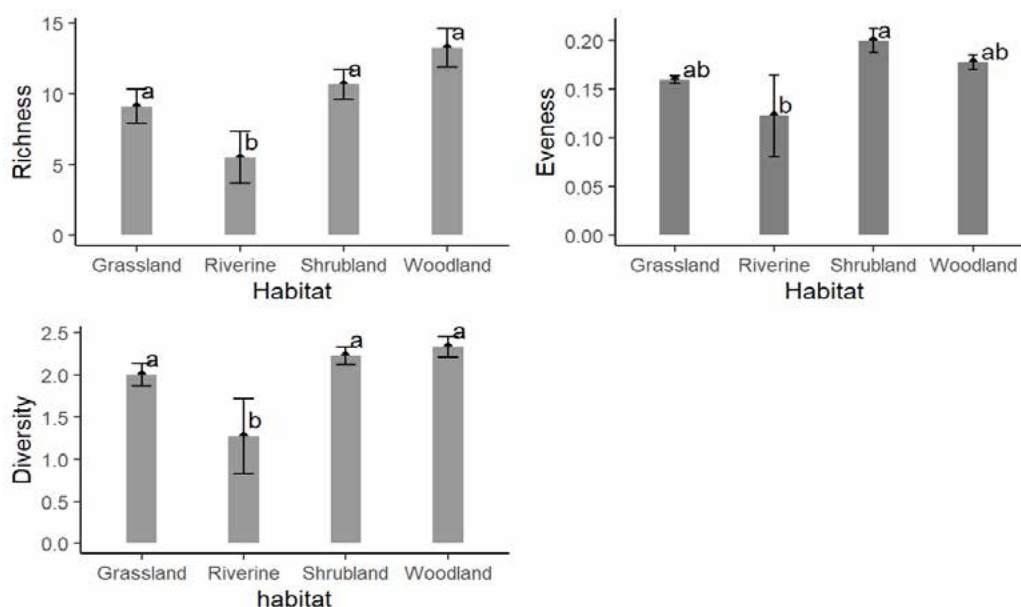


Figure. 6: Species richness (a), diversity (b) and evenness (c) as indirectly encountered in the different habitats of Ruaha National Park.

Spatiotemporal distribution of mammals in the RUNAPA for the past 30 years

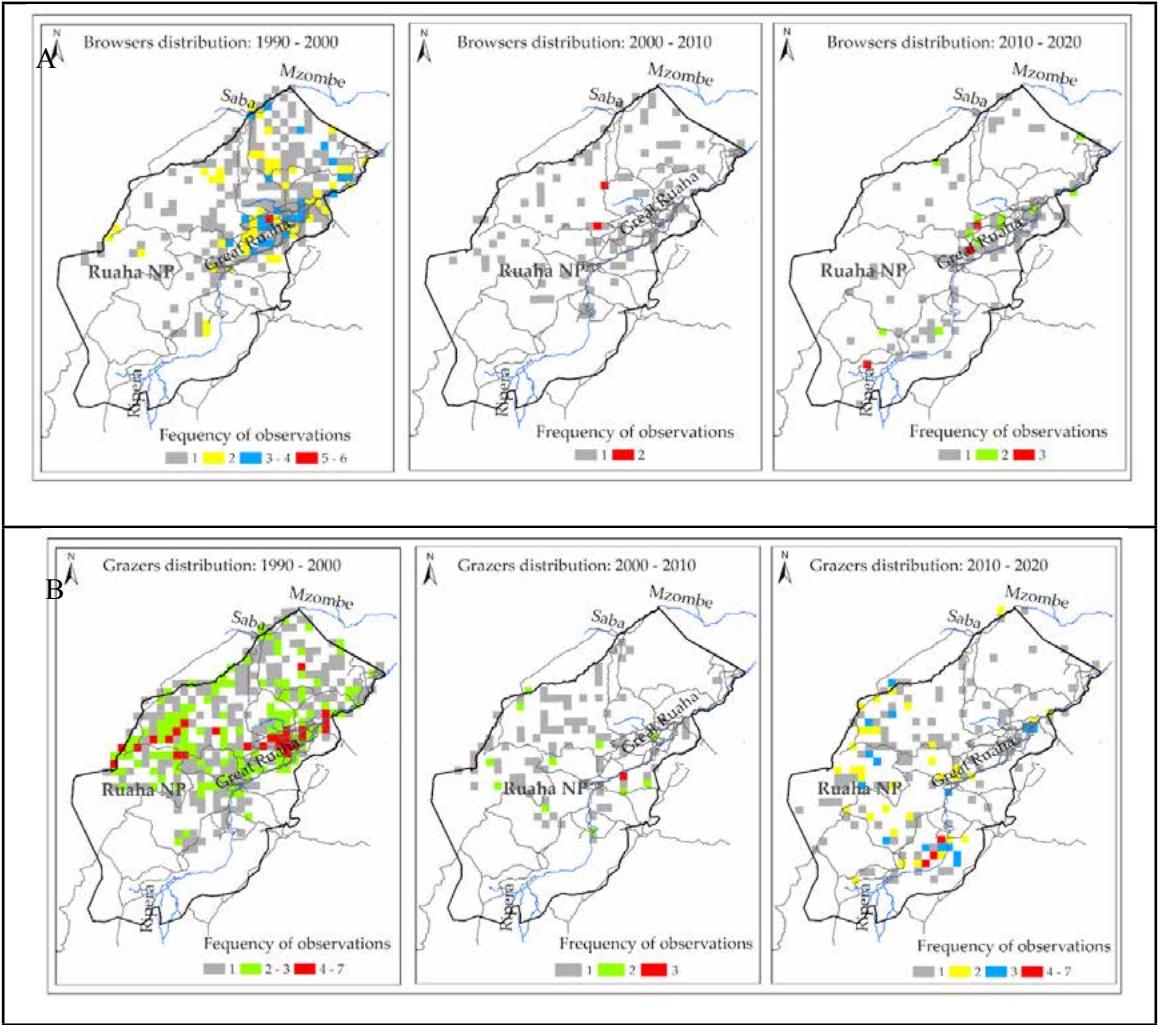
Based on the aerial surveys conducted in different time periods over 30 years (1990 –

2020), there were generally more observations of browsers, grazers, and mixed feeders in the northern and central parts of the park for the census conducted from 1990 to 2010. The distributions extended to the southern part

of the park during the 2010 – 2020 census period (Fig. 7). Observations of browsers decreased from 5 to 6 (1990 – 2000) to only 3 observations (2010 – 2020), (Figure 7 a). Grazers were consistently observed during the 1990 – 2000 and 2010 – 2020 census periods to a maximum of 7 observations with a slight decrease to 3 observations during the 2000 – 2010 census period (Fig. 7 b). Observations of mixed feeders have increased from 5 (1990 – 2000) and 2 observations during 2000 – 2010

to about 8 observations during the 2010 – 2020 census period (Fig. 7 c).

Moreover, the ground survey conducted during the dry season in 2021 indicated more observations of mixed feeders to a maximum of 30 observations followed by browsers with up to a maximum of 19 observations and grazers whose maximum observations were only 10 (Fig. 7 d).



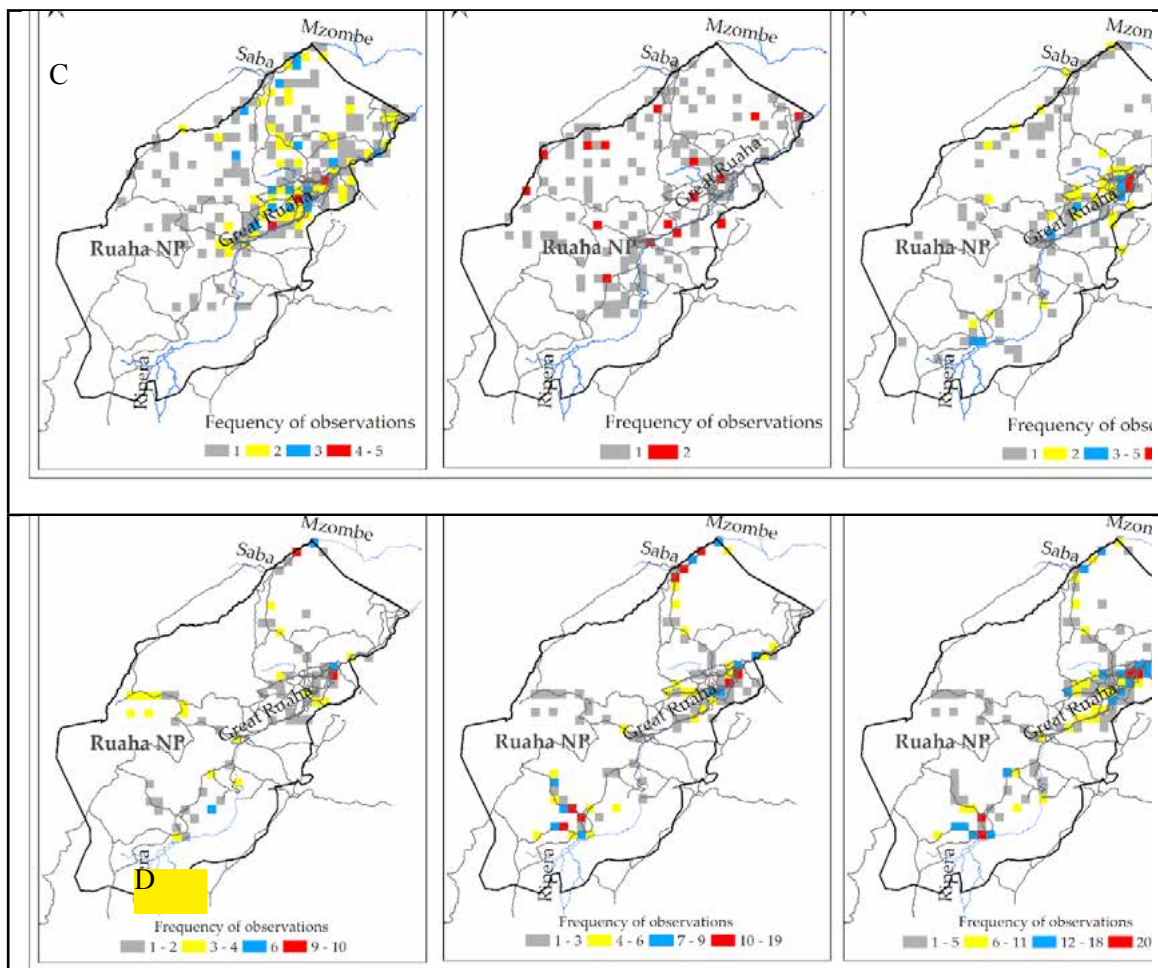


Figure 7: Trend of spatiotemporal distributions of browsers (a), grazers (b) and mixed feeders (c) as observed during the aerial census surveys for the period from 1990 to 2020, and the ground survey (d) conducted in 2021.

Spatially link mammal species distribution to the existing park road network to identify

Key areas for conservation and visitor experiences

Species were evenly distributed along most of the existing all-weather and seasonal road

network (Fig. 10 a), however, some parts of the south-east and south-west of the park have wildlife species but do not have all-weather roads, implying that they are not accessible during the wet season (Fig.10 b).

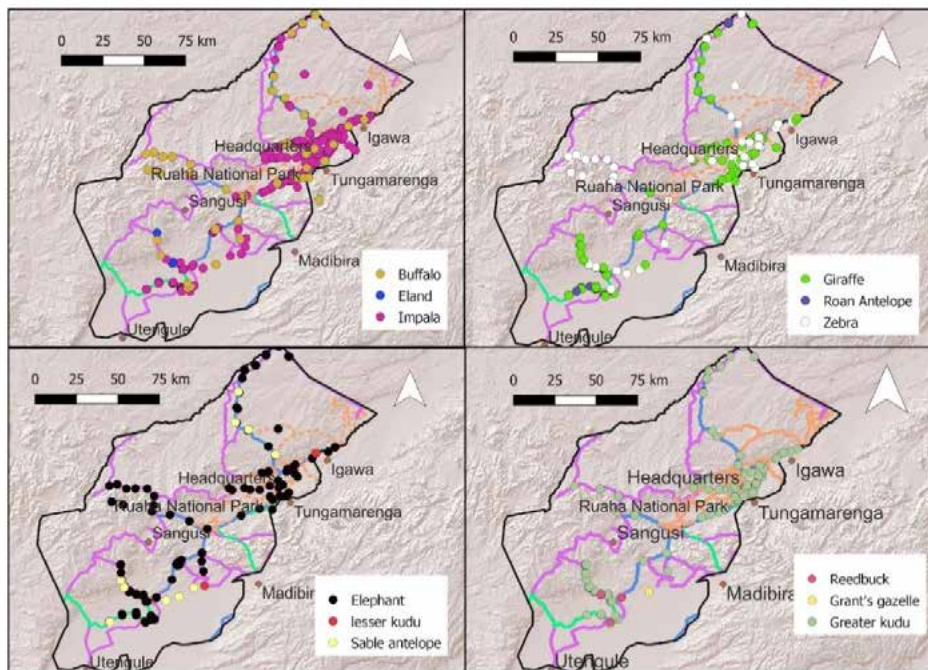


Figure 10: Species distribution along the existing all weather and seasonal (a) and all weather only (b) road networks of Ruaha National Park

DISCUSSION

The 29 wildlife species recorded in this study represent a major proportion of the species that can be found in RUNAPA, indicating a high potential of this park in terms of wildlife species diversity and therefore with a high potential for tourism activities as well as conservation of wildlife and their habitats (Kabuga, 2018; Mtahiko, 2004). Overall, species abundance and richness reveal that Impala (*Aepyceros melampus*) a mixed feeder is the most encountered species followed by the giraffe (browsers) and the greater kudu which browsers. Mixed feeders would inhabit most habitats because their stomach anatomy allows them to adopt on different food types (Hofmann, 1989) grazing during wet season and browsing in the dry season (Monro, 2017).

RUNAPA is dominated by acacia and miombo woodlands with few patches of grasslands in Usangu plains and Ihefu wetland (Roug *et al.*, 2020), this mosaic of habitat types is ideal for the above mentioned species implying that the

habitat is favoring both browsers and grazers habitat and feeding requirements, therefore able to exploit larger areas (Estes & Otte, 2012). Similarly, study covering the Ruaha-Rungwa ecosystem (Barnes & Douglas-Hamilton, 1982; Norton-Griffiths, 1975), on the density and distribution of wildlife showed the same dominance of Impala, lesser and greater kudu, buffalo, giraffe and elephants in RUNAPA.

Mammals distribution tend to be linked to specific habitats and adjacent to land use types (Starik *et al.*, 2020), hence it is important to understand the importance of such factors and processes while describing the observed distribution pattern. Both aerial survey and ground transect survey results clearly indicate that wildlife are substantially using the park, across the five zones and within the different habitat types. The increased distribution of observations in the southern part of the park during the 2010 – 2020 aerial census is attributed to the extension of the park in 2006 to include Ihefu wetlands.

Overlaying the ground survey observation on the existing roads network clearly demonstrate the need of improving the road network to areas with seasonal road networks. We found a mismatch between the existing all weather roads network and the distribution of mammals in RUNAPA; in particular, game viewing is more reliant on the road network and certain protected areas like the famous Serengeti National Park are more important for tourism simply because they are well invested in road networks (Eagles & Wade, 2006). Since tourists experience relies in observing a diverse number of species (Reynolds & Braithwaite, 2001), the concentration of roads at the central part of the park imply a high tourism pattern along this area leaving other parts of the park less visited as the seasonal roads are difficult to access during the dry season.

The relationship between mammals' distribution, road infrastructures and tourism potential documented in this study may be sufficiently general to translate to other protected areas with similar species richness though, the relative distribution of tourism infrastructures might vary across spatial scale. For example, the Park with rugged terrains with several hills, introducing the walking hiking safari would be suitable to capitalize game viewing in the areas which cannot be easily accessed using roads for example the Isunkaviola plateau.

We conclude that, RUNAPA is endowed with species diversity and richness and hence, suitable tourism destination. However, the limited access to the other parts of the park could negatively impact the parks tourism potential and consequently the country's economy. Furthermore, limited access means, low anti-poaching operations which is likely to accelerate illegal activities especially in the southern part which is recovering from human activities. We recommend that, funding from the multilateral and bilateral cooperation agencies and the government to prioritize in investing in infrastructures to support sustainable tourism in places which had seasonal roads.

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SUPPLEMENTARY INFORMATION

Table S1: Transects used during the 2021 ground survey in Ruaha National Park

SN	Zone	Transect ID	Vehicle transect (Km)	Walking transect (Km)
1	Central	Chaugigima	10	2
		Mdonya	30	6
		Msembe	15	3
		Kalenga Kelu	15	3
		Wio	5	1
		Total	75	15
2	Eastern	Lunda	25	5
		Madogoro	10	2
		Total	35	7
3	Northern (25Km)	Mpululu	20	4
		Mapenza	5	1
		Mdonya	15	3
		Total	40	8
4	Southern	Majimoto	5	1
		Ulanga	15	3
		Ng'irima	15	3
		Madundasi	15	3
		Makwaru	5	1
		Ilambitari	5	1
		Total	60	12
5	Western	Jongomero	10	2
		Isunkaviola	30	6
		Magangwe	5	1
		Total	45	9

Table S2. Species observed and signs encountered during the 2021 ground survey in Ruaha National Park

SN	Species	Direct observation	Indirect observation	Number of Encounters	Remarks for the indirect observations
1	Aardvark		11	11	Burrows and dungs
2	African civet		55	55	Dungs and tracks
3	Baboon	10		10	
4	Black backed jackal	3	9	12	Dung and tracks
5	Bufalo	10	57	67	Dung
6	Bushbuck	5	10	15	Dung
7	Bushpig	5	14	19	Dung and tracks
8	Dikdik	42	67	109	Dung and tracks
9	Duiker	13	54	67	Dung and tracks
10	Eland	1	28	29	Dung and tracks
11	Elephant	24	156	180	Dung and tracks
12	Giraffe	67	148	215	Dung and tracks
13	Grant's gazelle	3		3	
14	Greater Kudu	71	75	146	Dung and tracks
15	Hartebeest	2	17	19	Dung
16	Hippopotamus	4	8	12	Dung and tracks
17	Impala	402	126	528	Dung and tracks
18	Leopard		7	7	Dung and tracks
19	Lesser kudu	9	6	15	Dung
20	Lion	5	16	21	Dung and tracks
21	Olive baboon	1		1	
22	Red duiker		5	5	Dung
23	Reedbuck		4	4	Dung
24	Roan Antelope	4	2	6	Dung
25	Sable antelope	3	10	13	Dung
26	Serval cat		2	2	Dung and tracks
27	Sitatunga	1		1	
28	Southern Reedbuck	2	2	4	Dung
29	Spotted hyaena		48	48	Dung and tracks
30	Steinbok	1		1	
31	Topi	6		6	
32	Vervet monkey	28	5	33	Dung and tracks
33	Warthog	34	43	77	Dung and tracks
34	Waterbuck	13	4	17	Dung
36	Yellow baboon	9	17	26	Dung, pods and tracks
37	Zebra	28	41	69	Dung and tracks

Table S3: Descriptive statistics (meanSe), One-way ANOVA used to compare mean difference across and between zone at 5% confidence level directly observed in Ruaha National Park (Different letters across rows shows significant different otherwise non-significant difference)

Variable	Central	Eastern	Northern	Southern	Western	F-Value	P-Value
Richness	5.09	2.890.62	3.75	4.14	4.601.57	1.59	0.1850
Diversity	1.06	0.67	0.98	1.19	1.09	1.74	0.1490
Evenness	0.100.01	0.09		0.140.01	0.16AB	3.24	0.0164*

Table S4: Descriptive statistics (meanSe), One-way ANOVA used to compare mean difference across and between zone at 5% confidence level indirectly encountered in Ruaha National Park (Different letters across rows shows significant different otherwise non-significant difference)

Variable	Central	Eastern	Northern	Southern	Western	F-Value	P-Value
Richness		10.25	12.50	10.75	15.25	1.73	0.1660
Diversity	1.87	2.120	1.83	2.12	2.62	1.196	0.3300
Evenness		0.24	0.17	0.17	0.22	2.79	0.0400*

Table S5: Descriptive statistics (meanSe), One-way ANOVA used to compare mean difference across and between habitats at 5% confidence level directly observed in Ruaha National Park (Different letters across rows shows significant different otherwise non-significant difference)

Variable	Grassland	Riverine	Shrub-land	Woodland	Forest	Bush-land	F-Value	P-Value
Richness	5.39		3.40	4.31	4.00	1.00	0.983	0.4340
Diversity	1.23	1.16	0.72	1.01		0.00	1.85	0.1140
Evenness	0.14	0.13	0.08	0.12	0.14	0.00	2.38	0.0464*

Table S6: Descriptive statistics (meanSe), One-way ANOVA used to compare mean difference across and between habitats at 5% confidence level indirectly encountered in Ruaha National Park (Different letters across rows shows significant different otherwise non-significant difference)

Variable	Grassland	Riverine	Shrub-land	Woodland	F-Value	P-Value
Richness	9.11	5.50B	10.67AB	13.25	4.73	0.0070**
Diversity	1.99A		2.230.10	2.330.12	5.477	0.0033**
Evenness	0.17AB0.004	0.12B	0.20	0.190.01	4.36	0.0102*

LONG-TERM CHANGES IN DUNG BEETLE COMMUNITY STRUCTURE IN THE SAVANNA GRASSLAND OF THE SERENGETI ECOSYSTEM

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ABSTRACT

Dung beetles contribute to a range of important ecosystem processes, including secondary seed dispersal, nutrient recycling, and the partial suppression of faecal-oral transmitted parasites. Understanding the structure of dung beetle communities is therefore essential for interpreting how these functions operate within specific environments. However, growing evidence indicates that human-driven environmental change is reshaping insect communities, potentially altering the services they provide. In this study, we compared dung beetle survey data collected in 1991–1992 with equivalent data gathered in 2024 within the Serengeti Ecosystem, an area characterised by high densities of herbivore dung. Our results reveal substantial changes in community composition, relative abundance, and weight distribution across functional groups, alongside a decline in asymptotic species diversity. These shifts were associated with changes in rainfall patterns over time, highlighting broader ecological transformations occurring even within protected areas. The reduction in asymptotic species diversity likely reflects the loss of specialist or habitat-sensitive species, which are often the most vulnerable to environmental disturbance. Overall, our findings demonstrate that dung beetle community structure has undergone significant changes over the past three decades, indicating long-term ecological shifts in the Serengeti Ecosystem, with important implications for biodiversity conservation and the maintenance of ecosystem services.

Keywords: Ecological change, functional group(s), species diversity, species richness, relative abundance, biomass, rainfall patterns, herbivore faeces

INTRODUCTION

Coprophagous dung beetles (Scarabaeinae) are vital components of many terrestrial ecosystems, performing various ecological roles, including soil fertilisation, soil aeration, secondary seed dispersal, enhancing plant growth, nutrient cycling, and biological pest control, among others (da Silva & Cassenote, 2019; Doube, 2018; Salomão *et al.*, 2020). Dung beetles are also regarded as ecological indicator species and have been used to show environmental disturbances, as they are sensitive to and respond quickly to changes in landscape use and structure (Andresen, 2008; Noriega *et al.*, 2021; Panza *et al.*, 2025;

Stanbrook *et al.*, 2021). Furthermore, for the establishment and maintenance of diversity gradients, dung beetles have been used to study the importance of area, climate, and energy resources (Gebert *et al.*, 2020). Dung beetles use the ephemeral patches of dung from mammals as food and nesting resources; these energy resources influence dung beetle diversity, making them the most important taxon to study (Gebert *et al.*, 2020; Medina Hernández & Vaz-De-Mello, 2007).

Many dung beetle communities show strong seasonality within years (Andresen, 2005; Foster, 1993). In temperate regions, temperature is a seasonal determinant factor for dung beetle

communities, while in subtropical and tropical savannas, rainfall is significant as it determines the abundance and quality of their primary resources (manure) (Foster, 1993; Novais *et al.*, 2016; Pessôa *et al.*, 2021). Changes in rainfall patterns are thought to affect decomposition by altering the physiological traits, feeding behaviour, and interaction of decomposer arthropods (Sun *et al.*, 2023). For instance, reduced rainfall can markedly lower dung beetle abundance, which in turn diminishes dung decomposition rates. While the timing of rainfall has long been associated with the seasonal emergence of dung beetle populations, field-based evidence on how rainfall patterns impact richness, abundance, and ecological roles remains limited (Correa *et al.*, 2025; Sun *et al.*, 2023). Considering the variability of rainfall worldwide, further research is necessary to fully understand the links between rainfall and the detritus community.

Despite the known importance of dung beetles in supporting diverse ecosystems and a substantial decrease in insect diversity, particularly among dung beetles, during the twentieth century (Errouissi & Jay-Robert, 2019), there are few, if any, long-term studies conducted in a single locality on their community structure change (Escobar *et al.*, 2008; Howden & Howden, 2001). This could be because many terrestrial ecosystems have been significantly subjected to human habitat disturbance and land use over time (Gebert *et al.*, 2020; Howden & Howden, 2001), with a limitation of understanding of the dung beetles and other invertebrates' susceptibility to these anthropogenic pressures (Fuzessy *et al.*, 2021; Gebert *et al.*, 2020). The questions of how and why species communities fluctuate over spatial and temporal scales have long fascinated many ecologists (Escobar *et al.*, 2008). The comparison of these species' diversity across various spatial and temporal scales, as well as in different ecological and biogeographical contexts, tends to help ecologists understand the mechanisms that are

helping shape the community's structural and regulating functional dynamics (Ahuatzin *et al.*, 2023; Escobar *et al.*, 2008).

This study aims to compare the temporal variation in dung beetle community structure over extended periods in the savanna environment of East Africa, where rainfall and dung supply are highly variable. Using data collected on the dung beetle community between 1991 and 1992 in the Ndutu area within the Greater Serengeti Ecosystem by (Foster, 1993), compared to the ones collected in 2024 in the same sites, we aim to answer the following questions: (i) How has dung beetle species diversity and richness shifted over time? (ii) How has community composition changed over time, and does average monthly rainfall account for the observed patterns? (iii) How has the proportional abundance and weight distribution of the major functional group of dung beetles changed over time?

MATERIALS AND METHODS

Study area description

This study was conducted in the Ndutu area within the southern part of the Serengeti Ecosystem, in the northern Tanzania rangeland (3°00'45"S and 34°59'47"E; Fig. 1). This study area is along the border of Serengeti National Park (SNP) and Ngorongoro Conservation Authority Area (NCAA), specifically along and in the woodlands near Lake Ndutu and the adjacent grasslands. This location was selected primarily because it lies at the centre of the wildebeest's wet season range and encompasses diverse vegetation types in a compact area (Foster, 1993). As shown in Figure 1, six study sites (B, C, D, 4, 5, and 6) were established in the Ndutu area to represent contrasting vegetation types. While four sites (site 1, 2, 3 and A) were in the open treeless grassland.

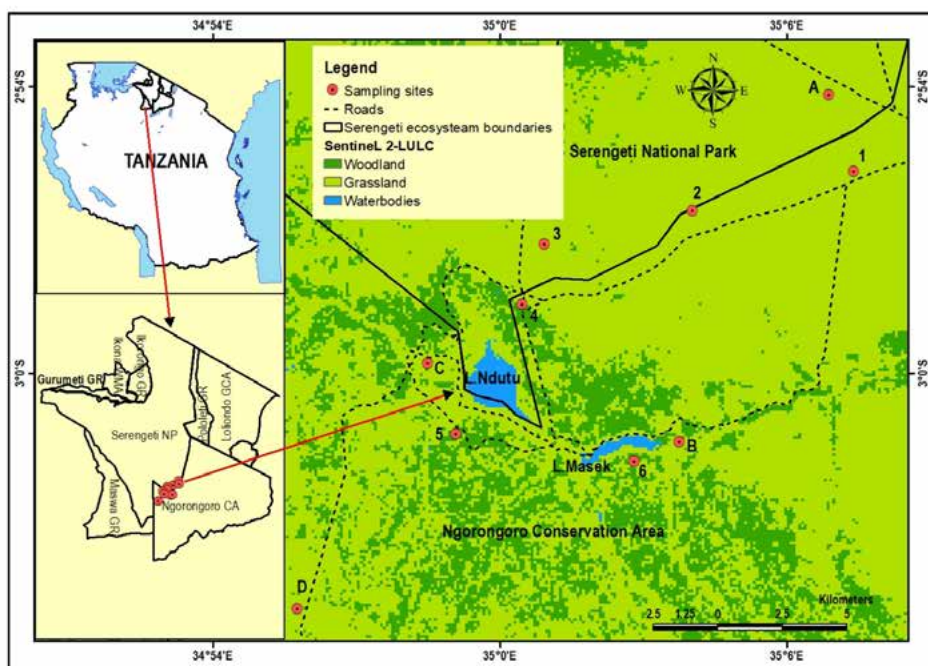


Figure 1: A map showing the study area with 10 sites within a broad vegetation type, near Lake Ndutu in the southern part of the Serengeti Ecosystem.

Study design and sampling of dung beetles

Using a detailed map from Foster (1993), aligned with Google Earth imagery, this study resampled the same locations originally surveyed. All ten sites sampled between 1990 and 1992 were revisited twice in 2024. At each site, a 500 m linear transect was established approximately 50 m from the habitat edge and oriented perpendicular to the nearest road. Five pitfall traps (5 L buckets, 21.5 cm diameter) were installed along each transect at 50 m intervals, with rims flush to the ground to allow beetles of all sizes to enter. Each trap was baited with 300 g of wildebeest dung suspended slightly above the soil surface. Traps were deployed in the morning (07:00–09:00) and collected after 24 hours. Since previous work indicated minimal differences between diurnal and nocturnal assemblages, this sampling approach was unlikely to bias species representation. Collected specimens were preserved in 70% ethanol and later processed at the TAWIRI entomology laboratory for cleaning, sorting, identification, and counting. Individuals were first grouped

into morphospecies and then identified to species level where possible, with unmatched taxa retained as unidentified morphospecies. Foster's original study, conducted between October 1990 and September 1992, sampled dung beetles across multiple sites using pitfall traps baited with dung from various herbivores, although wildebeest dung proved most effective. Sampling intensity varied seasonally, with traps deployed multiple times per month and checked after 24 hours. Sites typically contained six permanently installed traps spaced at least 20 m apart. Trap design included a buried bucket fitted with a funnel and bait suspended above, later modified to increase capacity and allow drainage. Sampling occurred in both morning and evening to capture diurnal and nocturnal species, and a reference collection was archived at the Hope Museum of Entomology, Oxford.

Functional groups

Dung beetles are classified into functional guilds based on how they use dung for feeding, breeding, and nesting. These include dwellers,

which live and feed within dung; tunnellers, which bury dung directly beneath the pat; rollers, which transport dung away before burial; and kleptoparasites, which exploit dung buried by other species. Because dung beetle communities are shaped by both environmental and biological factors, this study grouped species into three main functional categories based on dung relocation behaviour: rollers, tunnellers, and dwellers. To capture functional differences, rollers were further divided by body size (large vs. small), reflecting their dung-removal capacity, while tunnellers were classified by burial speed (fast vs. slow), which influences burial depth and competitive ability.

Long-term rainfall patterns in the Ndutu area

The Ndutu area has a bimodal rainfall pattern, with long rains from March to May and short

rains from November to December, and receives about 600 mm of annual rainfall (Fig.2A), classifying it as semi-arid. Rainfall data were obtained from monthly records near Lake Ndutu, showing that long-term patterns (1990–2024) are consistent with those observed during the sampling years (1991, 1992, and 2024). To reduce seasonal bias, analyses focused on two comparable wet months with similar rainfall conditions. December 1990 data were combined with January 1991 due to minimal seasonal differences. Only samples using wildebeest dung bait were included to ensure consistency. February and April (1991–1992) were selected for comparison as they matched rainfall conditions in February and March 2024. Overall, rainfall has increased over the past three decades (Fig. 2B).

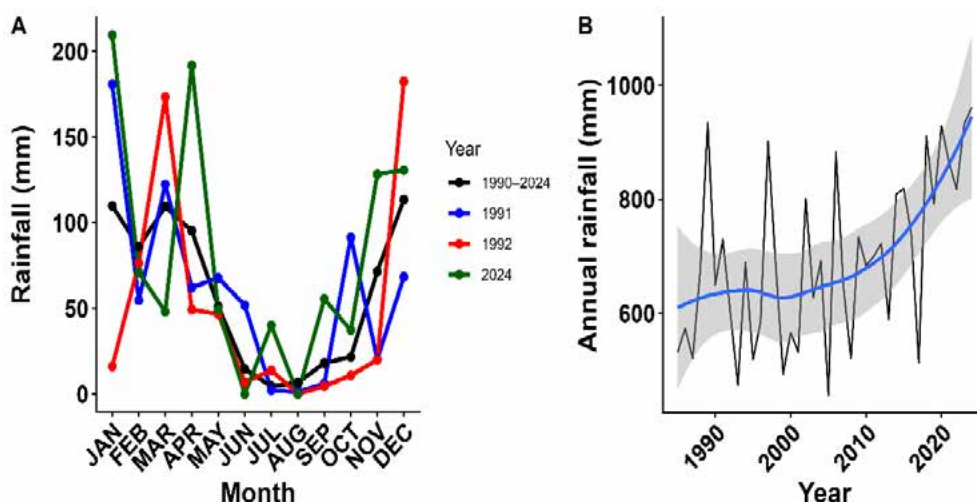


Figure 2: A) The average monthly rainfall (mm) for the Ndutu area in the southern part of the Serengeti Ecosystem from 1990 to 2024 B) Long-term annual rainfall trend in the Ndutu area from 1990 to 2024.

Statistical analysis

Dung beetle diversity and sampling completeness were assessed using the iNEXT package, which generated species accumulation curves based on Hill numbers for species richness ($q = 0$) and diversity ($q = 1$ and $q = 2$), using the number of individuals as

a proxy for sampling effort. Analyses included all species (identified and unidentified) and focused on two comparable sampling months. Sampling coverage was estimated to evaluate inventory completeness, and temporal changes in diversity were quantified using asymptotic Hill numbers. Because species richness is sensitive to rare taxa, multiple diversity orders

were interpreted to capture patterns of species presence, evenness, and dominance. Changes in community composition were analysed using multivariate approaches. Species counts per trap were standardised by sampling effort and converted into Jaccard dissimilarities, which were visualised using two-dimensional NMDS ordination. Temporal differences were tested using PERMANOVA with 999 permutations, and pairwise comparisons were conducted with Bonferroni correction. Rainfall (monthly and cumulative) was included as a covariate in separate models to assess environmental effects. Analyses were conducted both across all sampling months and restricted to two focal months to control for seasonal variation. Temporal shifts in functional group absolute abundances were analysed using negative binomial GLMMs, incorporating year and functional group as fixed effects and trap nested within sampling date as a random effect. Model selection was based on likelihood ratio tests and AIC, with post hoc comparisons conducted using Tukey-adjusted estimated marginal

means. Finally, Individual beetle weights were measured and compared across years using non-parametric Kruskal–Wallis tests followed by Dunn’s post hoc tests. All analyses were performed in R.

RESULTS

Species accumulation curves approached an asymptote across all sampling years, with inventory completeness exceeding 95%, indicating that sampling effort was sufficient to capture most dung beetle species during the focal months (Figure 3). This consistency supports the reliability of comparisons in functional group abundance, biomass, and asymptotic Shannon diversity among years. In total, 141,918 individuals were recorded across the two focal months from 1991 to 2024. The highest abundance occurred in 1992 (63,951 individuals; 45.06%), followed by 1991 (47,344; 33.36%), while 2024 recorded the lowest abundance (30,623; 21.58%) (Fig. 3).

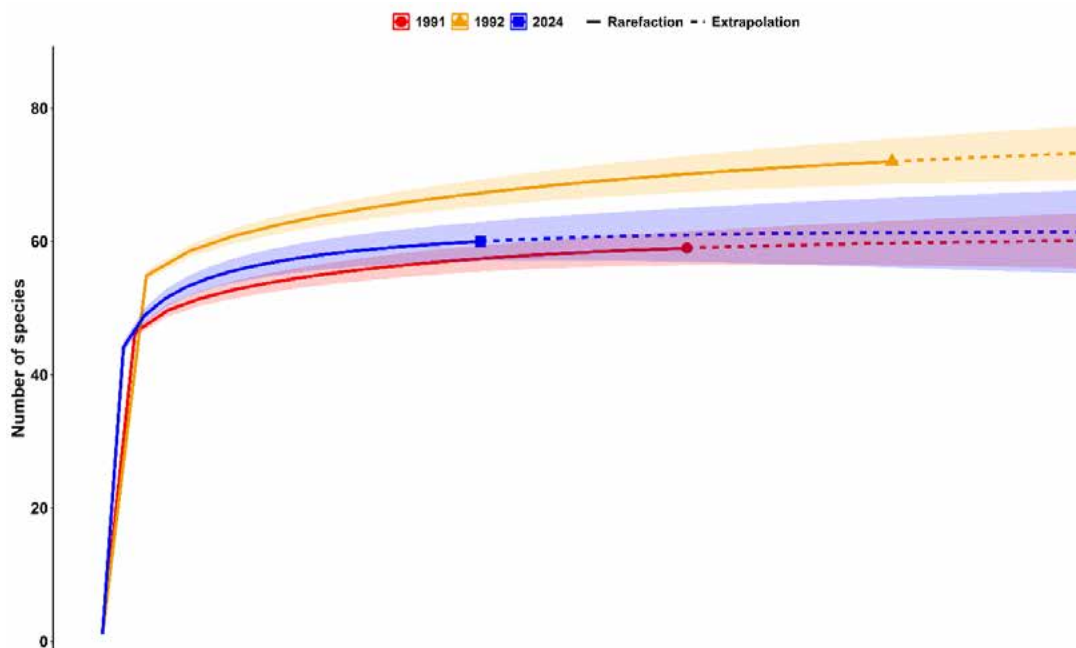


Figure 3: Species accumulation curve across sampling years of dung beetles sampled with pitfall traps using wildebeest dung bait in the Ndutu area in the Southern Serengeti Ecosystem. The shaded area is a 95% confidence interval.

Asymptotic Shannon diversity and species richness change over time

Shannon and species richness were highest in 1992 (72 species), compared with 59 species in 1991 and 60 species in 2024 (Fig. 3), with asymptotic richness showing the same pattern (Fig. 4A). Shannon and Simpson diversity

were also greatest in 1992, indicating higher evenness and lower dominance during this period (Fig. 4B and 4C, respectively). In contrast, diversity declined by 2024 compared to 1992, suggesting increased dominance by fewer species and simplification of community structure over time.

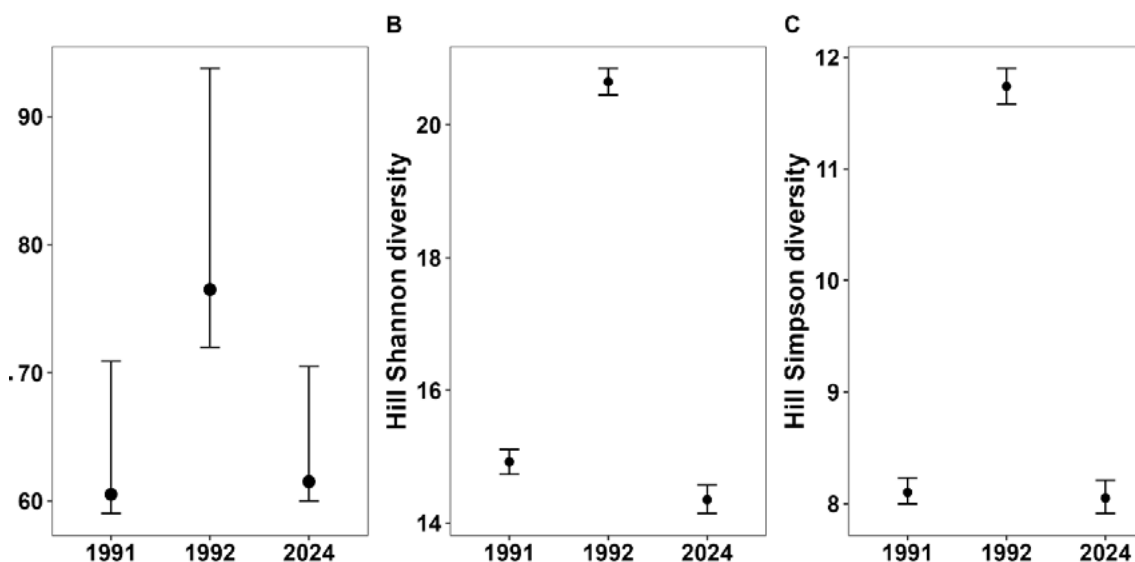


Figure 4: A) Asymptotic estimates of species richness, B) Hill-Shannon diversity across years, C) Hill-Simpson diversity across years. The vertical error bars indicate 95% confidence intervals.

Dung beetle community composition changes across sampling years with monthly and three-month cumulative rainfall effects (for all sampling months and two focal months)

NMDS based on Jaccard dissimilarity produced a moderate stress value (0.127), indicating that the two-dimensional ordination reliably represented differences in community composition among years across all sampling months. PERMANOVA results showed significant temporal variation in dung beetle community composition across the three sampling years ($F = 37.83$, $R^2 = 0.129$, $p = 0.001$; Fig. 5). Three-month cumulative rainfall

was also a significant predictor of community composition ($F = 36.678$, $R^2 = 0.067$, $p = 0.001$), although it explained less variation than year effects.

Pairwise PERMANOVA comparisons indicated significant differences between all sampling years after Bonferroni correction ($p = 0.003$ in all cases), including between 1991 and 2024 ($F = 30.921$, $R^2 = 0.087$, $p = 0.003$), 1992 and 2024 ($F = 23.681$, $R^2 = 0.077$, $p = 0.003$), and 1991 and 1992 ($F = 19.714$, $R^2 = 0.045$, $p = 0.003$). Overall, these results indicate strong temporal turnover in dung beetle community composition.

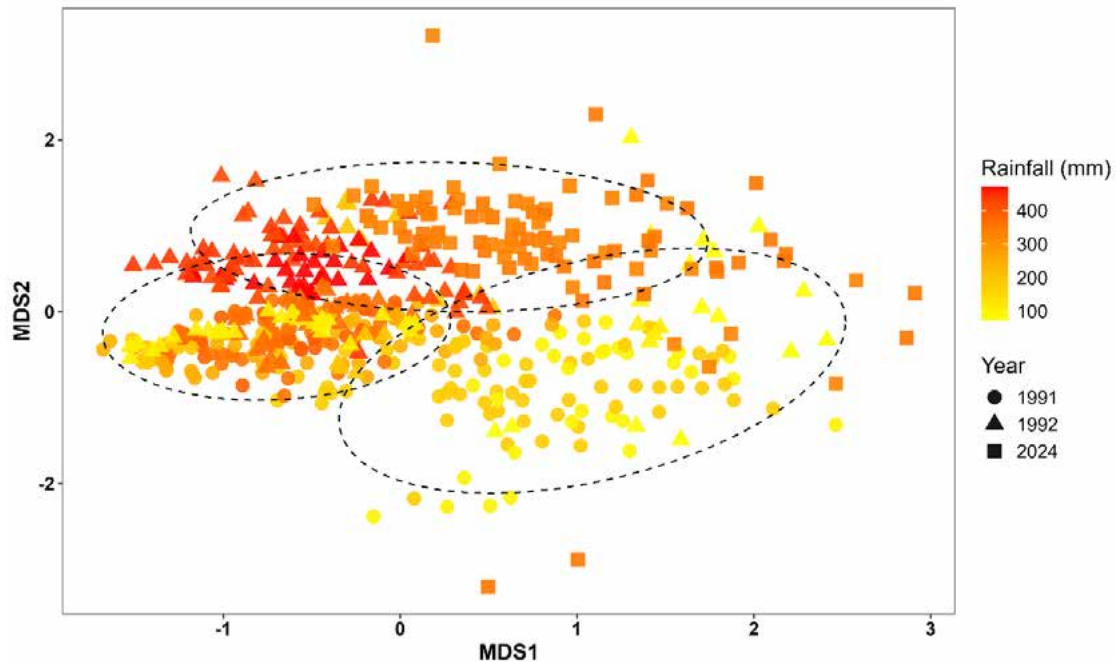


Figure 5: Changes in dung beetle species composition across sampling years (1991, 1992, and 2024) in the Ndutu area of the southern Serengeti Ecosystem, based on Jaccard dissimilarity. Each point represents a 24-hour dung beetle sample pooled across traps at each site and sampling month for each year. Colours indicate the cumulative 3-month rainfall corresponding to each sampling month.

Shifting of the absolute abundance of functional groups over time

The model including the interaction between year and functional group provided a significantly better fit than the additive model ($\chi^2 = 592.36$, $df = 8$, $p < 0.001$), indicating that temporal changes in dung beetle abundance differed among functional groups (Figure 6). Year effects were significant for 2024 compared with 1991 ($z = -3.22$, $p = 0.001$), but not between 1991 and 1992 ($z = -1.840$, $p = 0.066$). Post hoc analyses showed that dweller abundance was significantly higher in 2024 than in 1991 ($p = 0.003$), with no significant differences between 1992 and 2024 ($p = 0.096$)

or between 1991 and 1992 ($p = 0.158$). Fast-burying tunnellers increased from 1991 to 1992 ($p = 0.017$) but declined in 2024 relative to 1992 ($p = 0.004$), with no difference between 1991 and 2024 ($p = 0.405$). Large rollers showed a consistent and strong decline across all years, with significant differences between all pairwise comparisons (all $p < 0.001$). Slow-burying tunnellers increased over time, with significant differences between 1991 and 1992 ($p < 0.001$), 1991 and 2024 ($p < 0.001$), and 1992 and 2024 ($p = 0.010$). Small rollers also increased significantly in 2024 compared to both 1991 and 1992 ($p < 0.001$), with no difference between 1991 and 1992 ($p = 0.834$).

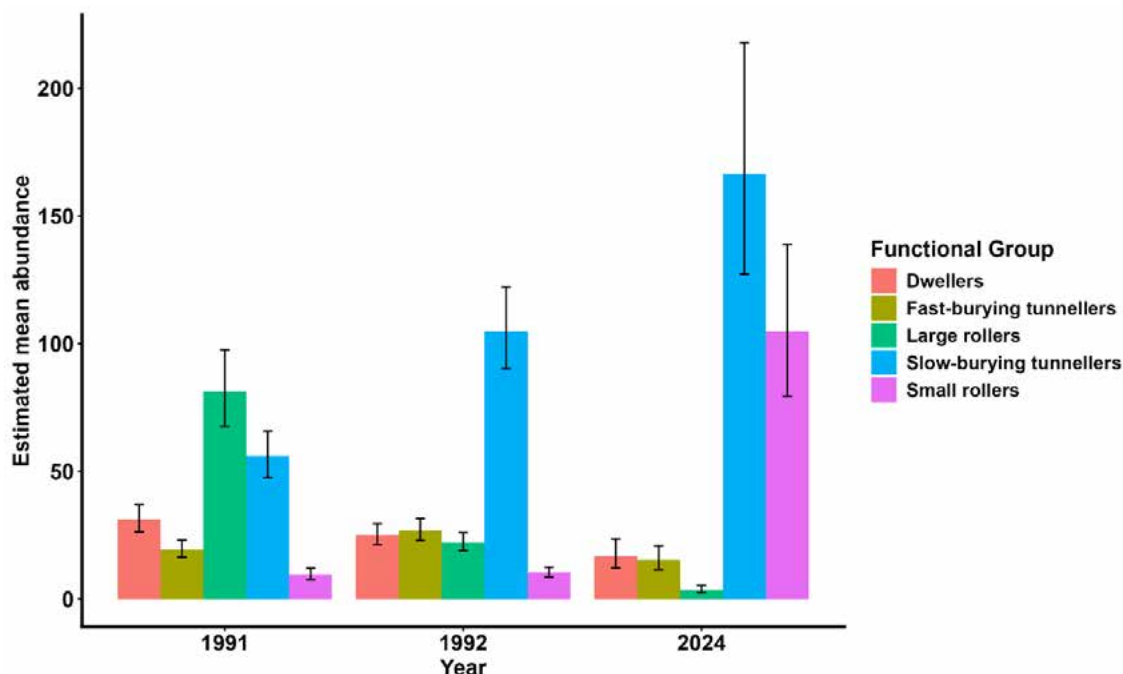


Figure 6: Estimated mean absolute abundance per trap per sampling date with 95% confidence intervals. Functional groups showed contrasting temporal trends consistent with the significant year $\times$ functional group interaction. Large rollers and fast-burying tunnellers exhibited the strongest declines over time, whereas small rollers and slow-burying tunnellers either increased or remained relatively stable across years.

Total weight distribution

The distribution of individual dung beetle body weight across three sampling years (1991, 1992, and 2024) showed a clear shift toward smaller-bodied individuals over time ($\chi^2 = 7871.7$, $df = 2$, $p < 0.001$; Figure 7). Specifically, the upper tail of the distribution decreased, indicating a decline in large-bodied dung beetles, while

the relative frequency of smaller-bodied individuals increased over time, with pairwise highly significant differences between years (1991 vs 1992, 1991 vs 2024, and 1992 vs 2024, all $p < 0.001$). This pattern suggests a decline in total weight contributed by large species, accompanied by a relative increase in smaller species across the years.

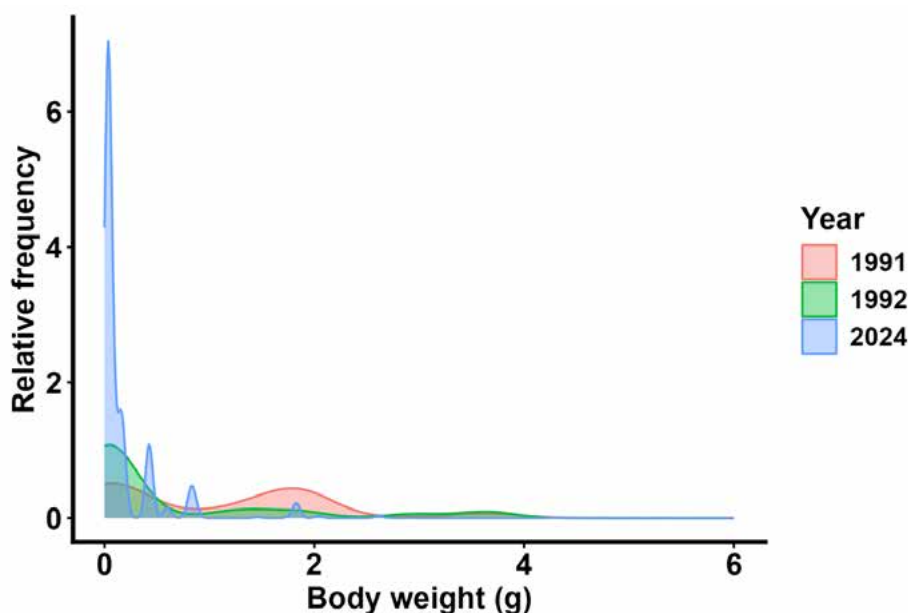


Figure 7: Density distribution of individual dung beetle body weights across the three sampling years (1991, 1992, and 2024), with each curve showing the relative frequency of weights per year. Differences among years were assessed using the Kruskal–Wallis test, followed by pairwise Dunn tests with Benjamini–Hochberg correction.

DISCUSSION

Long-term changes in the dung beetle community

Overall, this long-term study demonstrates substantial changes in dung beetle community structure in the Serengeti Ecosystem, including declining diversity, strong shifts in species composition linked to rainfall variability, marked changes in functional group composition, and reduced biomass, collectively indicating altered ecosystem functioning. Species diversity declined significantly over the study period, with reductions in asymptotic Shannon diversity suggesting long-term ecological change. This likely reflects the loss of specialist and habitat-sensitive species (Cazalis, 2022), which are most vulnerable to environmental stress, as evidenced by the absence of ~17.5% of species recorded in 1991–1992 in 2024. Species with narrow ecological niches or limited dispersal are particularly prone to local extinction under environmental change (Brink, 2025), while reduced evenness and increased dominance may also contribute to declining diversity (Escobar *et al.*, 2008).

Despite similar seasonal sampling windows, variation in rainfall across years was an important driver of community turnover and declining diversity. Rainfall strongly influences resource availability, habitat structure, and organismal biology (Cristaldo *et al.*, 2025), particularly in arid and semi-arid systems where both rainfall amount and timing shape ecological interactions (Fischer *et al.*, 2022; Liberal *et al.*, 2011). Previous studies also highlight precipitation as a key determinant of dung beetle dynamics in seasonal ecosystems (Beiroz *et al.*, 2018; Canziani & González-Vainer, 2024; Ndozi *et al.*, 2025), while long-term changes in richness and composition may result from climate variability and land-use interactions (Escobar *et al.*, 2008).

Functional composition and biomass also changed significantly over time, with declines in large-bodied rollers and tunnellers and increases in small rollers, consistent with reduced total biomass. Similar shifts have been linked to environmental disturbance favoring small, generalist species over large-bodied taxa (Escobar *et al.*, 2008; Quintero & Halfpeter, 2009). Increasing anthropogenic disturbance

in the Serengeti may be contributing to habitat degradation and altered community structure (Veldhuis *et al.*, 2019), consistent with evidence that disturbance reduces dung beetle diversity (Estupiñan-Mojica *et al.*, 2022) and other taxa richness in the region (Sinclair *et al.*, 2015).

Possible effects of wildebeest on the long-term changes of dung beetle communities

The wildebeest population shows no significant change in abundance over the past three decades, with stable numbers since the early 1990s (Figure 8). This suggests that wildebeest

population size is unlikely to have driven the observed changes in dung beetle communities. In contrast, long-term rainfall in the Ndutu area has increased significantly from the 1990s to 2024 (Figure 2B). This rise in precipitation may have altered vegetation structure and indirectly influenced wildebeest habitat use, including the timing of arrival, duration of stay, and departure of migratory herds in the Ndutu region (Foster, 1993). Such environmental changes could affect dung availability and interspecific competition among dung beetles, potentially contributing to the observed decline in large rollers and tunnellers.

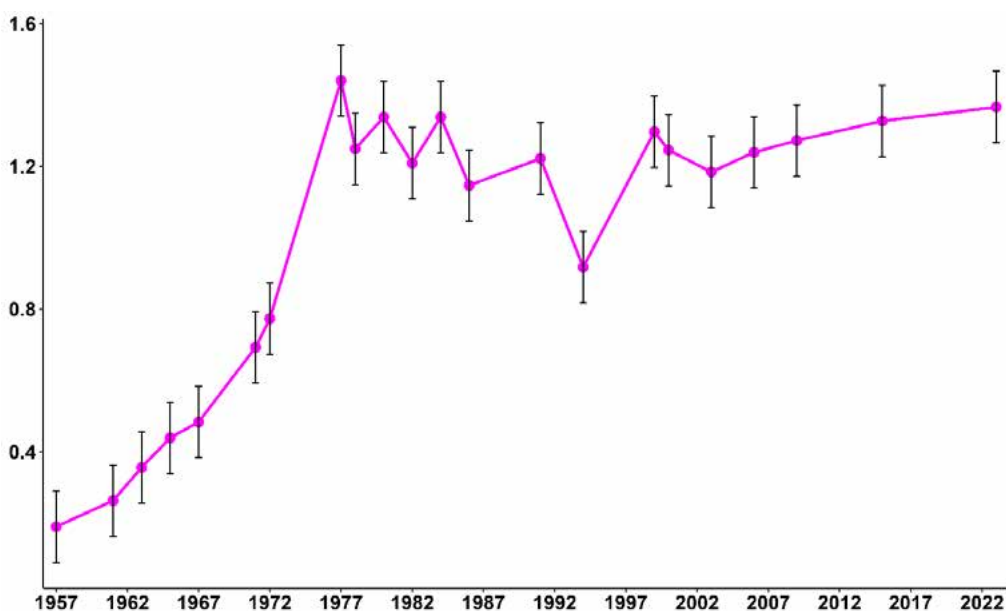


Figure 8: Long-term trend in migratory wildebeest estimated abundance in the Serengeti Ecosystem over the years (from 1957 to 2015). This data highlights wildebeest population changes over time.

CONCLUSION

Therefore, our results emphasise the need to integrate insect biodiversity into conservation planning. Long-term ecological monitoring, including invertebrate groups such as dung beetles, is crucial for detecting early indicators of ecosystem degradation. Conservation approaches should also prioritise sustaining healthy herbivore populations and reducing human pressures around protected area boundaries to maintain dung beetle diversity.

Overall, this study adds to increasing evidence supporting a wider taxonomic focus in biodiversity assessments and highlights the value of long-term comparisons for understanding ecological change.

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IMPACT OF HABITAT DISTURBANCE ON THE ABUNDANCE OF BLACK-AND-WHITE COLOBUS MONKEYS IN RAU FOREST, MOSHI, KILIMANJARO, TANZANIA

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ABSTRACT

This study was conducted at Rau Forest Reserve, Moshi-Tanzania on May 2025. The study aims to assess how habitat disturbance influences the abundance and behaviour of black-and-white colobus monkeys (*Colobus guereza*). Scan and focal sampling were used to record overall activities of the species and also line transect method to record the abundance of black-and-white colobus monkeys at Rau Forest Reserve. Undisturbed zones had high abundance of the colobus monkeys (87), following by moderate disturbance (77) whereas, the disturbed zone had least of the individuals of black-and-white colobus monkeys (61) statistically there was no significant difference on the mean abundance of these species where ($F=1.051$, $df=2$, $p=0.361$). Also, the activity patterns done at Rau Forest were resting, feeding, vigilance, grooming and moving in which the habitat disturbance affected their behavioral pattern, and statistically there was no significant difference on variation of these activity pattern where ($F= 4.78$, $df=4$, $P= 0.58$). Generally, Colobus monkeys were more abundant in undisturbed areas, indicating a preference of these species for quality habitat and low human interference. The variation between disturbed and undisturbed activity patterns suggests behavioral adaptation to environmental stress. The study therefore recommends that Tanzania Forest Services (TFS) should put more emphasis on conservation efforts in minimizing habitat disturbances by formulating a customized management plan for Rau Forest, enacting strict laws and provision of community conservation education. Also, regular monitoring should be conducted to safeguard the forest and ensure the long-term survival of Colobus populations in the area.

Key words: Black-and-White Colobus Monkeys, Disturbance, Rau Forest

INTRODUCTION

The mantled guereza, or black and white colobus monkey, is among the most recognizably important primates in Sub-Saharan Africa. These old-world monkeys, which belong to the family Cercopithecidae, feature bold, contrasting black and white fur, long tails, and intricate social behaviours. The coloration has, until recent years, been the main distinguishing feature for classification within the genus (Butynski *et al.*, 2025; HULL, 1978). From an ecological perspective, their folivorous diet makes them important for forest ecosystem health because of their role in seed dispersal and in fostering plant recovery (Harris, 2006).

Globally, there are five recognized species of black and white colobus; *Colobus guereza*, *C. angolensis*, *C. polykomos*, *C. satanas*, and *C. vellerosus* (Behie, 2015). *Colobus guereza* is sub-speciated into seven subspecies by (Groves, 2007) as *C. g. caudatas*, *C. g. dodingae*, *C. g. guereza*, *C. g. kikuyuensis*, *C. g. matschiei*, *C. g. accidentalis*, and *C. g. percivali*. The subspecies classification of *C. guereza* remains unresolved (Grubb, 2006). This species account provisionally follows the arrangement of Chapman & Chapman, (2000), Grubb, (2006) and Fashing, (2007) in recognizing eight subspecies. Sadly, the colobus monkeys have been relentlessly pursued for their unique pelts, and as bushmeat, which in combination

with loss has placed increasing pressure on their populations (Dunham, 2017; Red & Jong, 2020). The hunting of primates for their meat is predominantly practiced in Central and Western Africa, while it is less common in Eastern and Southern Africa (Anderson, 2021; Harrison *et al.*, 2016; Levi *et al.*, 2011; Stoner *et al.*, 2007). Increased hunting pressure often coincides with logging activities due to the influx of hunters and the demand for meat that supports the logging crews (Rosin, 2014; Sirén *et al.*, 2006; Wright, 2003). Colobus monkeys and most Gorilla populations in Africa tend to rely on foliage during the lean seasons but may be driven to total habitat devastation. Analysis of survey data indicates that human hunting power has a more severe impact than selective logging or low intensity bush-fallow farming (McLester *et al.*, 2019). The majority of primate populations face continuous active and passive anthropogenic activities (Mittermeier *et al.*, 2012). Many primate species are classified as endangered or vulnerable, underscoring the need to grasp better the primary threats to their survival. Unfortunately, deriving generalizations about how primate populations respond to logging has proven to be difficult (Chapman *et al.*, 2007).

In the Kilimanjaro region of northern Tanzania, the Mount Kilimanjaro guereza *C. g. caudatus* is known to be native primate to northeast Tanzania. It can be found on both Mount Kilimanjaro and Mount Meru, along with smaller groups in various forest reserves on the lower south and east slopes of Mount Kilimanjaro. These reserves include Rau, Kahe I, Kahe II, and Kileo Forest Reserve (Butynski & De Jong, 2015). Rau Forest is one of the last remaining pieces of lowland forest and provides important habitat for many wildlife species, including Colobus guereza. This forest is part of the Eastern Arc biodiversity hotspot, recognized for its many unique species and ecological importance (Burgess *et al.*, 2007). However, human activities such as illegal logging, agricultural expansion, and settlement growth are quickly damaging this habitat. Additionally, natural threats like wildfires and

unpredictable climate changes from global warming are worsening habitat loss (IPCC, 2021; Plumptre *et al.*, 2016).

The black and white colobus monkeys have a limited distribution, found mainly in equatorial Africa (Than *et al.*, 2009). Their population in Africa is primarily confined to the western side, extending from Nigeria and Cameroon to Ethiopia, and the eastern side, including Kenya, Uganda, and Tanzania (Groves, 2007). In Tanzania, black and white colobus monkeys are mostly located in the northern regions, including the montane forests of Kilimanjaro National Park, Chome Nature Reserve and Shengena forest in the South Pare -Same district; these areas have a good population of black and white colobus monkeys which might encountered habitat disturbances from human pressure and Mount Meru in the Arusha National Park (Rodgers *et al.*, 1980), as well as in Rau Forest Reserve. A viable population of black and white colobus monkeys exists in protected areas, but habitat loss and fragmentation threaten their survival (Rodgers *et al.*, 1980).

The black and white colobus monkey is a totally arboreal species, dependent on intact forest cover for locomotion, foraging, and social behaviour. Fragmentation of habitat breaks such natural activity and compels the monkeys to adapt to poor-quality habitats, which can reduce their birth rates and increase mortality (Fashing *et al.*, 2011; Marsh & Chapman, 2013). This disruption may lead to population decline, loss of genetic diversity, and even local extirpation in highly disturbed areas.

Habitat loss poses the highest threat to biodiversity worldwide (Debinski & Holt, 2000; Fahrig, 2003; Jr *et al.*, 2018; Wilcove *et al.*, 1998). A higher threat to species with small and narrow ranges (Hogue & Breon, 2022). An instance of this kind of species is black and white colobus monkeys because they are restricted in secluded forests of riverine, lowland, montane, and coastal areas (Fashing, 2007; Rodgers *et*

al., 1980). There is extensive evidence that previously forest habitats were linked but with disturbances from human beings and climate change the forests are fragmented (Rodgers *et al.*, 1980; Wade *et al.*, 2013).

Most of the primates like black and white colobus monkey populations are threatened with habitat degradation (deforestation), though not all of them respond positively to that disturbance (Fashing *et al.*, 2012; Petros, 2018). This is a major threat to white and black colobus monkeys as they inhabit the forest which are degraded (Fashing *et al.*, 2012). Timber logging and land clearing for agriculture impact the population of colobus monkeys, resulting in fragmented distribution (Rodgers *et al.*, 1980). Deforestation impacts Colobus Monkey but there also exist hunting activities. Although they are ecologically relevant, little site-specific data exist regarding the impact of habitat disturbance by black and white colobus on population numbers in Tanzania, specifically in dynamically evolving forests like Rau. Most literature available aims at broad East African or other primate species trends. Localized study is thus acutely in need to inform focused conservation programs and sustainable forest management techniques (Struhsaker, 2005).

Rau Forest, located in Moshi District, Kilimanjaro, Tanzania, is a biologically diverse remnant forest with a high diversity of plant and animal species, including the black and white colobus monkey (*Colobus guereza*). The species is of ecological important as a folivorous primate, with roles in seed dispersal and plant diversity conservation in forests (Marsh & Chapman, 2013). In addition, their attractive appearance and arboreal lifestyle render them an attractive species for ecotourism, which contributes to both national revenue and local living in terms of nature tourism (Burgess *et al.*, 2007; Red & Jong, 2020). Despite its economic and ecological worth, Rau Forest is increasingly threatened by a combination of human activities and natural disturbances. Activities including illegal forest clearing,

agricultural expansion, charcoal collection, and urbanization are fragmenting the landscape and altering the structural condition of the forest (Ahrends *et al.*, 2010; Plumptre *et al.*, 2016). These activities reduce canopy cover, and the presentation of the food sources is critical to the survival of colobus monkeys. In addition, intrinsic factors including climate variability, forest fires, and alien vegetation species further weaken habitats (Struhsaker, 2005). Colobus guereza is an exclusively arboreal animal that depends considerably on intact forest canopy cover for locomotion, feeding, and predator evasion. Disturbance and fragmentation of the habitat can lead to loss of habitat, restrict movement, and increase susceptibility to risks such as predation or man-conflict (Fashing *et al.*, 2011; Marsh & Chapman, 2013). Such changes can disrupt social group harmony, lower reproductive success, and ultimately lead to population declines (Struhsaker, 2005). While such threats are clear, there still exists a large gap in localized, site-specific research of the impact of habitat disturbances on the population and behavior of black and white colobus monkeys in Rau Forest. There are broad studies or those of other sites and species. This shortage of empirical data hinders the conservationists, policymakers, and forest managers in making intervention based on the realities of this ecosystem (Fashing *et al.*, 2012; Oates *et al.*, 2000).

The objective of this study was to; Assess the impact of habitat disturbance on the abundance of black-and-white colobus monkeys at Rau Forest. Specifically, the study was conducted to meet the following specific objectives (i) determined different types of habitat disturbances in Rau Forest (ii) analyzed the abundance of black and white colobus monkeys in different types of habitat disturbances and (iii) determined the activity pattern of black and white colobus monkeys in disturbed versus undisturbed areas of Rau Forest.

We hypothesized that H1: Rau Forest

is subjected to multiple types of habitat disturbances, including but not limited to logging, agricultural encroachment, grazing, and firewood collection, with varying frequency and spatial distribution. H2: The abundance of black and white colobus monkeys is significantly lower in highly disturbed habitats compared to moderate or undisturbed areas within Rau Forest. H3: Black and white colobus monkeys exhibit different activity patterns such as increased vigilance and reduced feeding or resting time in disturbed areas compared to undisturbed areas of Rau Forest.

METHODOLOGY

The study area

Rau Forest Reserve (Fig. 1) is situated in Moshi urban district in Kilimanjaro region. Rau Forest Reserve was established by the Government of Tanzania in 1957 and is one among of the urban forest in Tanzania located 3km Southeast of Moshi town between latitude, 3° 23' S and longitude, 37° 22' E. It covers an area of 25 km² receiving an average rainfall of 870mm and mean temperature of 24°C (Eustace *et al.*, 2015). The forest is characterized by a mix of closed canopy and disturbed forests which are home

to various wildlife species including black and white colobus monkeys. Rau Forest Reserve is also known as catchment forests among the important watershed areas in Tanzania and it is surrounded by six villages which are Msaranga, Mabogini, Mandaka Mnono, Njoro, Mji mpya and Kaloleni. The soil in this region consists of fluvisols and nutrient rich of gleysoils which are above alluvial sand and loam of volcanic origin. The minimum temperature is 21° C in July and maximum temperature in 26° C in February with average rainfall of 870mm/year (Rodgers *et al.*, 1980).

The vegetation in the study area can be classified into 3 categories (Rodgers *et al.*, 1980) as follows: - Natural ground water forest, this forest consists of tree species like *Lavoa swynnertonii*, *Albizia schimperiana*, *Ficus sycomorus*, *Newtonia buchananii*, *Tabernaemontana ventricosa*, *Macaranga kilimandscharica*, *Oxystigma msoo*, *Cordyla africana*, *Rauvolvia caffra*, *Syzygium guineense* and *Trichilia emitica*. Another vegetation is swamp forest. This forest consist of tree species like, *Ficus sycomorus*, *Acacia xanthophloea*, *Phoenix reclinata* and *Milcia excels*. The last vegetation found in Rau Forest Reserve is woodland forests which consists of tree species like *Acacia xanthophloea*, *Cedera odorata*, *C. mexicana*, *Lagerstroemia speciosa*, *Eucalyptus spp.*, *Cassia spp.*, *Tectona grandis* and *Mahogany spp.*

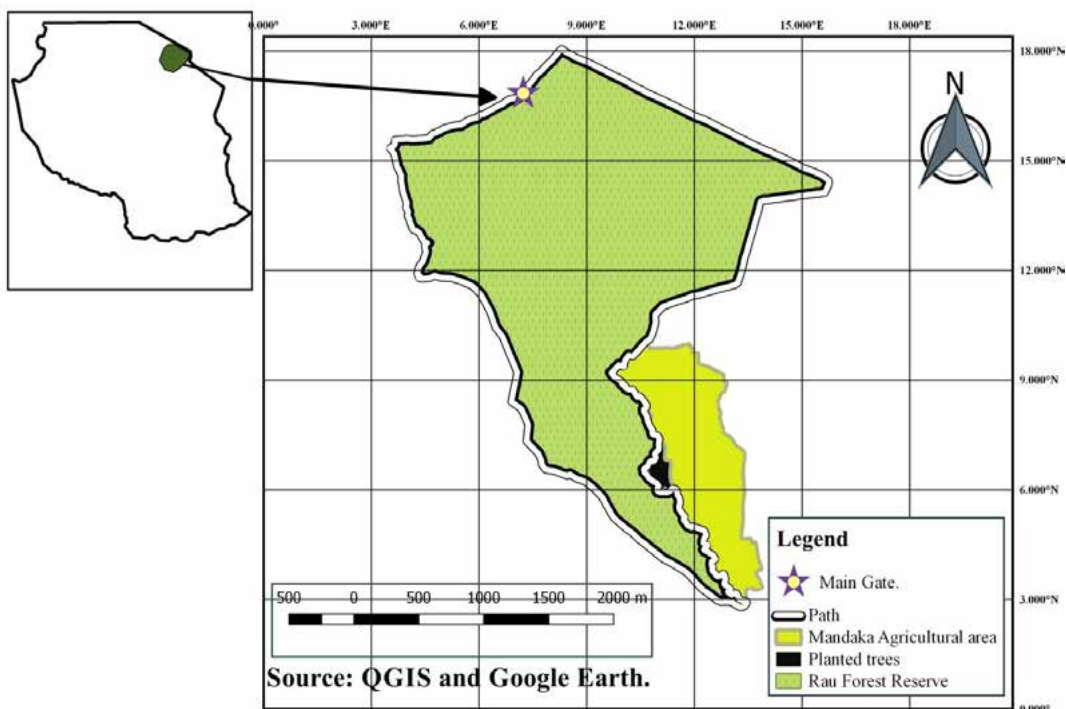


Figure 1: Map of Rau Forest reserve

Research approach and design

Quantitative research with numerical data by counting the colobus monkeys and measuring the likely disturbance levels occurring at Rau Forest Reserve. The study used a cross-sectional study design since the data were collected once at single point within a short period of time

Sampling technique

The study used stratified random sampling that divided Rau Forest Reserve into three main strata, the disturbed, moderate disturbed and undisturbed areas.

Sampling design

Different transects were plotted in different zones in the forest to observe and count colobus

monkeys seen or heard within a fixed distance (Fig.2). The established zones presented the low, moderate and highly disturbance parts of the forest, and each zone consisted of two transects with a total of six transects in all parts of the forest. Whereas, in each transect covered 1km length and width of 30m and the distance from one transect to another was 200m (Rovero *et al.*, 2006). In these transects different signs were recorded such as logging, footpaths, and human presences. All the walk transects were done early in the morning (at 6:30AM) and in the evening (at 4:30PM), and at each sighting point, perpendicular distance from the observer, time of sighting, number of individuals and the coordinates were recorded using GPS (eg Gamin 65).

Data collection

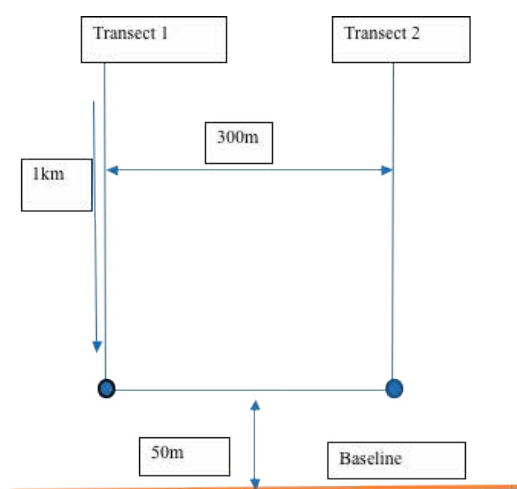


Figure 2: Transect layout and plots

Direct observation

The colobus monkeys were spotted and directly counted through direct observation. Also, binoculars were used where the species were a bit far from the observer, and were recorded in different types of habitats, and gather the data by assessing the groups of monkeys and the habitat condition in their natural environment. This was done especially in the morning or late afternoon period per day.

Line transect method

Different transects were set up in different parts of the forest representing low, moderate and high disturbance. It was a walk transect done at specific times usually early in the morning and in the evening then black and white colobus monkeys were counted when seen or heard within the fixed distance.

Focal and scan sampling

Focal observation was done for 5 minutes in every 15 minutes interval and a single individual, especially adult individual, was chosen opportunistically and recorded the

activity done within every 5 seconds. Group activity was recorded every 15 minutes by the use of scan sampling to determine the activity which was done by at least 50% of the group members.

Data analysis

In this study descriptive statistics were used to analyze data collected from the field by summarizing the abundance of colobus monkeys and the levels of disturbance in different zones using Microsoft software (Excel 2022) which is a versatile and user-friendly tool for organizing data, performing statistical calculation and forming visual presentations such as charts and graphs. Statistical Package for Social Science version 21 (SPSS 21) was used to analyze overall activity patterns of colobus monkeys, and the ratio number of records were transformed into percentages. On the other hand, ANOVA test was very helpful in the statistical analysis to determine whether there is significant difference in the mean abundance of the monkeys across low, moderate and high disturbance zones. ANOVA test was opted as a powerful and the most common statistical test to compare means of more than two groups.

RESULTS

Types of habitat disturbance

The assessment of habitat disturbances in Rau Forest Reserve revealed five primary types of habitats disturbances: agriculture, firewood collection, logging, natural factors and other activities (Tourism activities, beekeeping, human presences in the forest). These disturbances varied in their intensity levels, indicating varying levels of pressure on the forest ecosystem (Fig. 3). The results indicate that agriculture and other human activities are the most intense forms of disturbance, while firewood collection was moderate but also natural factors and logging were at lower intensity.

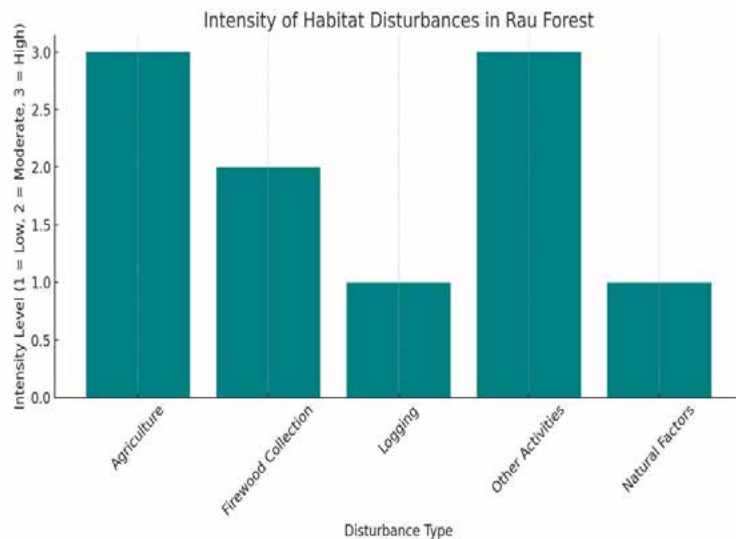


Figure 3: Intensity of habitat disturbance in Rau Forest

Black-and-White Colobus Monkeys Abundance

A total of 225 colobus monkeys were observed at all of the disturbance levels, (Disturbed, moderate disturbed and undisturbed). In which out of these 61 individuals were recorded in disturbed areas (27%), 77 individuals in moderate disturbed (34%) and 87 individuals in undisturbed areas (39%). The highest number of colobus monkeys was recorded in undisturbed areas, followed closely by moderate disturbed areas, while the lowest was disturbed areas (Fig. 4). This trend suggests that possible preferences of black-and-white colobus monkeys are in areas with minimal human disturbances and better habitat quality. Statistically, there was no significant difference in colobus monkeys' abundance across the three disturbance categories ($F=1.051$, $df=2$, $p=0.361$).

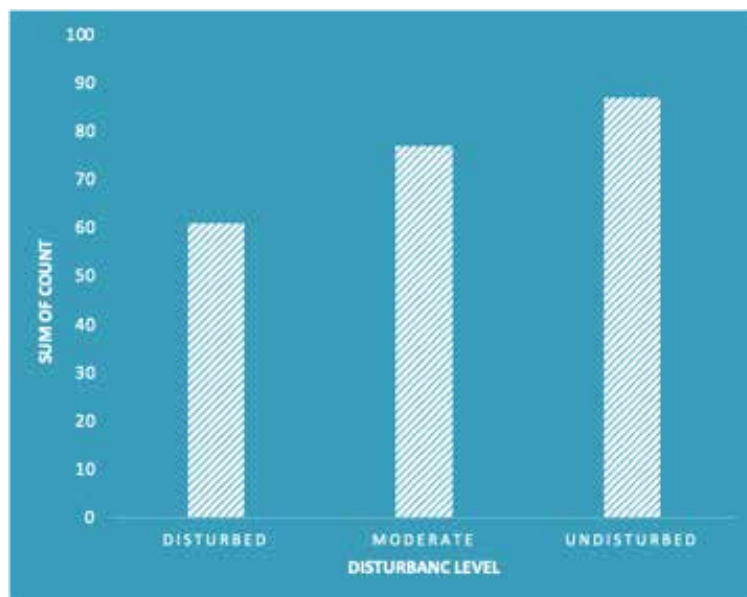


Figure 4: Showing Distribution of colobus monkey's sightings across different disturbance level in Rau Forest

Activity Pattern

A total of 141 observations were collected from both focal and scan sampling from one group with group size of 4 to 16 individuals. Black-and-white colobus monkeys allocated their time across different activity patterns in disturbed and undisturbed areas of Rau Forest. The main behaviour observed were feeding, resting, moving, grooming and vigilance (Fig. 5). In disturbed areas, Colobus monkeys spent most of their time feeding (39%), followed by resting (28%), Vigilance (17%), moving (11%) and grooming (5%). On the other hand,

in undisturbed areas, Colobus monkeys spent most of their time resting (42%), followed by feeding (23%), moving (14%), grooming (13%) and vigilance (8%) was lower in undisturbed areas compared to disturbed areas indicating increased alertness likely due to human presence or habitat disruption. These results suggest that habitat disturbance significantly affects colobus monkeys 'daily activity budgets shifting energy from rest and grooming toward vigilance and foraging. But the difference between these activities was not significant ($F = 4.78$, $df=4$, $P = 0.58$).

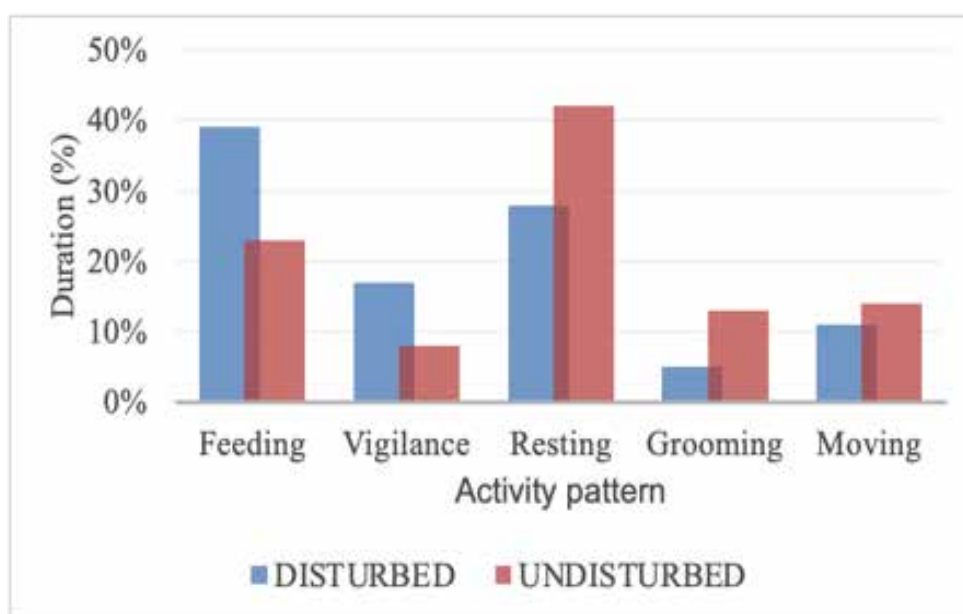


Figure 5: The effect of habitat disturbance on activity pattern

DISCUSSION

Types of habitat disturbance

According to the results it shows that anthropogenic pressures are more dominant than natural factors which show least disturbance. Agriculture and other human activities are the most highly intense disturbance, suggesting that they are primarily ecological threats at Rau Forest, leading to habitat fragmentation and degradation. Similar studies have shown

trends in reports of other tropical forests, where land conversion for agriculture and unsustainable resources extraction are primary drivers of loss of biodiversity and fragmentation (Keenan *et al.*, 2015; Newmark, 2008). Firewood collection showed moderate disturbance while natural factors and logging had lower disturbance intensity suggesting that forest's ecological balance is less threatened by natural factors such as windstorms, floods or seasonal droughts compared to anthropogenic disturbances.

Black-and-White Colobus Monkeys Abundance

Black-and-white colobus monkeys' abundance was high in undisturbed areas this may be influenced by high availability of food in undisturbed area compared to the disturbed areas as colobus monkeys are primarily folivorous, feeding on young leaves in selected tree species (Xia *et al.*, 1994). Another reason which probably made difference between the abundance was weather condition, black-and-white colobus monkeys tend to hide themselves when there is rainfall or cloud condition (Wikberg *et al.*, 2020). Disturbed habitat had low abundance since it typically had low vegetation quality and structural integrity needed to support such species as it is often characterized by logging, firewood collection, or agricultural encroachment (Fashing, 2001). Moderate disturbed was not rich as undisturbed areas but it still supports reasonable number of individuals this could be due to spatial retention of canopy or proximity to less disturbed patches that allow some level of movement and resource access.

Activity Pattern

The results indicate a clear shift in behavioral time allocation between disturbed and undisturbed habitats, with observable trends in feeding, vigilance, resting, grooming, and moving behaviors. These patterns suggest behavioral adaptations in response to anthropogenic pressure, consistent with existing literature on primate behavioral ecology. In disturbed areas colobus monkeys spent more time on Resting, vigilance and feeding. Comparing to other studies, Increased vigilance is often a behavioral response to environmental stressors such as habitat fragmentation and human presence (Beale & Monaghan, 2004; Treves *et al.*, 2001). Also, feeding occupied a higher proportion of time in disturbed areas. This may reflect reduced food quality or availability, forcing monkeys to increase foraging effort to meet nutritional needs (Chapman & Chapman, 2000; Forest *et al.*, 2017). Habitat disturbance may result

in the degradation or loss of preferred food trees, compelling primates to spend more time searching for less optimal resources (Kibaja *et al.*, 2023).

On the other hand, resting experienced high behavioral activity pattern compared to the other activity pattern in undisturbed areas this may be due to as argued by other authors, high resting might be due to behavioral thermoregulation with colobus resting under shaded tree canopies during afternoon and in sunny canopies in the morning and evening. Also, much time in resting might be needed to assist food digestion as colobus monkeys feed on cellulose-rich food materials requiring long gut passage times (Eustace *et al.*, 2015). Also, by comparing it to other studies suggests that resting behaviour was significantly more common in undisturbed habitats. In stable environments with fewer external stressors, animals can afford to allocate more time to energy-conserving activities like resting (Schlabritz-loutsevitch *et al.*, 2007). Grooming as a critical social behavior in primates, also decreased in disturbed sites. Reduced grooming may reflect a shift in priorities from social bonding to environmental scanning and foraging (Petros, 2018; Wijten *et al.*, 2012).

CONCLUSION AND RECOMMENDATION

This study revealed that habitat disturbance affects the abundance of black-and-white colobus monkeys in Rau Forest Reserve. Their abundance was highest in undisturbed areas and lowest in disturbed zones, indicating that habitat quality play's important role in determining the abundance of this species. Furthermore, observed behavioral differences between disturbed and undisturbed areas highlight the species' sensitivity to environmental changes. Overall, the study concludes that colobus monkey populations and behaviour are adversely affected by habitat disturbance. The integrity of this species' habitat, which is increasingly threatened by human activity, is closely related to its survival in Rau Forest.

RECOMMENDATIONS

The study conducted provides the following recommendation; Tanzania forest services (TFS) should enhance forest protection in undisturbed zones by giving high conservation priority with stronger enforcement against illegal activities such as logging and firewood collection also should involve local community in forest stewardship through conservation education.

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FACTORS INFLUENCING COMMUNITY AWARENESS OF THE ENDANGERED *Rungwecebus kipunji*

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ABSTRACT

Understanding the factors that influence community awareness is crucial for designing conservation approaches that effectively involve local populations. This research studied what determines the level of awareness about the *Rungwecebus kipunji* among residents of villages near the Mount Rungwe Forest Nature Reserve (MRFNR) in Tanzania. Data were gathered from four villages situated at varying distances from the reserve boundary: Iloilo, Kibisi, Syukula, and Lupoto. A household survey was conducted with 160 participants (40 from each village) using a structured questionnaire consisting of closed-ended questions. Binary logistic regression was applied to identify key factors affecting awareness, while a generalized linear mixed model (GLMM) was used to analyze predictors influencing perceptions of the Kipunji as a pest. Results indicated that proximity to the forest (<3 km) significantly increased awareness ($p = 0.016$). Additionally, ethnicity was significant ($p = 0.044$), with members of the Nyakyusa group nearly three times more likely to be aware of the endangered Kipunji compared to the Kinga group. Age also had a notable effect, where respondents aged 18-25 exhibited greater awareness than other age groups ($p = 0.013$). Regarding the perception of the Kipunji as a pest, education level ($p < 0.001$) and distance from the village ($p < 0.001$) were influential factors. Therefore, these results highlight the roles of geographical location, cultural background, and age demographics in shaping conservation awareness and attitudes.

Keywords: community awareness, conservation education, endangered species, Mount Rungwe Forest, *Rungwecebus kipunji*

INTRODUCTION

The urgency of conserving biodiversity and protecting endangered species continues to attract global attention (Jones *et al.*, 2005; Sodhi *et al.*, 2010). Within Tanzania's unique ecological systems, *Rungwecebus kipunji*, an endemic primate found in the Southern Highlands, exemplifies both the country's rich biological heritage and the conservation challenges it faces (Davenport *et al.*, 2008). Increasing anthropogenic pressures on natural habitats (Brooks *et al.*, 2010) underscore the need for integrated conservation strategies that combine scientific interventions with local community participation and awareness creation (Ehardt & Butynski, 2006). Successful protection of endangered species such as

Rungwecebus kipunji depends largely on the knowledge, attitudes, and involvement of neighbouring communities (Davenport & Jones, 2005). Identifying the social and ecological determinants that shape community awareness is therefore essential for the formulation of effective and inclusive conservation initiatives (Mbise *et al.*, 2021). The present study examined the socio-demographic and spatial factors influencing the awareness of *Rungwecebus kipunji* among residents of villages adjacent to the Mount Rungwe Forest Nature Reserve (MRFNR) in Tanzania.

Commonly known as the Kipunji, *Rungwecebus kipunji* is a critically endangered monkey species restricted to the MRFNR (Davenport & Jones, 2005; Ehardt & Butynski,

2006). Since its discovery in 2003, the species has been distinguished by its unusual morphology and characteristic vocal patterns (Ehardt & Butynski, 2006; Jones *et al.*, 2005). The Kipunji population faces a rapid decline primarily due to forest degradation, illegal hunting, and inadequate conservation enforcement (de Marques *et al.*, 2016). Consequently, the species has become a focal concern for conservation practitioners and researchers (Ehardt & Butynski, 2006). Active participation of local communities enhances the success and sustainability of conservation programmes targeting endangered species (Ehardt & Butynski, 2006; Hill, 2002). Raising awareness among people living near conservation areas promotes environmental stewardship and encourages behaviours that support biodiversity preservation (Hill, 2002; Mbise & Sosiya, 2023). However, increasing human activities within and around MRFNR continue to endanger Kipunji populations and degrade the reserve's forest habitats (Ehardt & Butynski, 2006; Jones *et al.*, 2005). Habitat destruction and fragmentation remain the most pressing challenges affecting both the species and the ecological integrity of the forest ecosystem (T. Davenport & Jones, 2005; T. R. Davenport *et al.*, 2008; Ehardt & Butynski, 2006).

This study therefore aimed to evaluate the extent of community awareness regarding the conservation status of *Rungwecebus kipunji* among residents of villages neighbouring the Mount Rungwe Forest Nature Reserve. By identifying existing knowledge gaps and community perceptions, the findings provide

a foundation for the development of targeted conservation education programmes. The outcomes are expected to enhance participatory conservation strategies and contribute to broader efforts in conserving both the Kipunji and the ecosystems that sustain it.

METHODS

Study area

The Mount Rungwe Forest Nature Reserve (MRFNR) lies within Rungwe District in the Mbeya Region of southwestern Tanzania, extending approximately between latitudes 9°3'–9°12'S and longitudes 33°33'–33°45'E (TFS, 2017). The surrounding communities depend largely on subsistence agriculture and small-scale livestock production as their main livelihood sources. The reserve is bordered by densely cultivated farmlands and village settlements, reflecting strong human influence in the landscape. Climatically, Mount Rungwe is characterised by a tropical montane environment, receiving substantial rainfall mainly between November and April. During the drier months, occasional mist and light showers are common. Annual precipitation varies considerably with altitude—averaging about 900 mm in the lowlands, 2700 mm in the uplands, and reaching up to 3000 mm along the southeastern slopes. Temperature conditions also fluctuate widely, from lows of around –6°C in the high-altitude zones to highs of approximately 29°C in the lower areas (TFS, 2017).

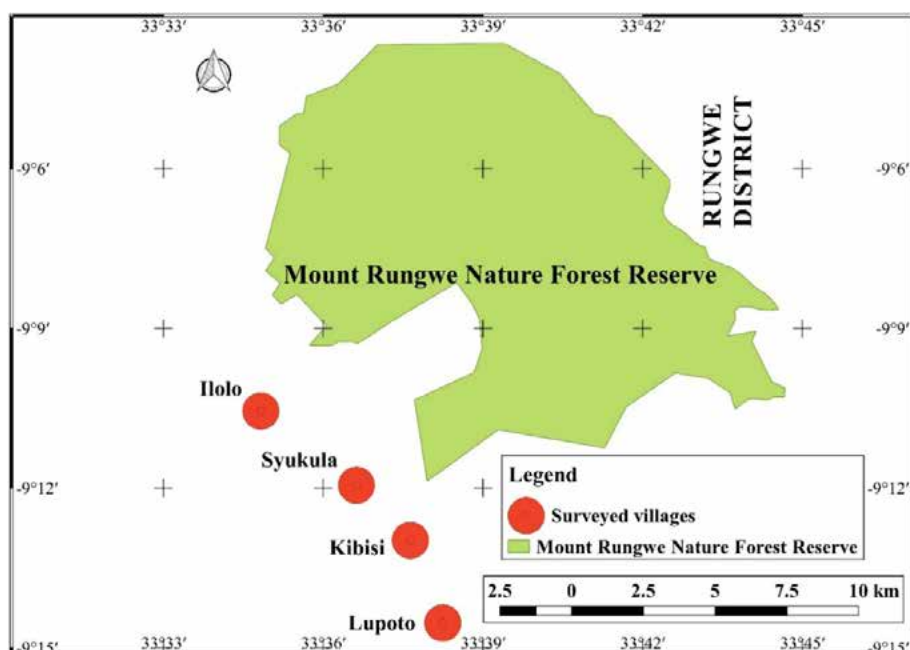


Figure 1: The surveyed villages Ilo, Kibisi, Syukula, and Lupoto are positioned along a distance gradient extending outward from the boundary of the Mount Rungwe Forest Nature Reserve (MRFNR), as illustrated by the red dots on the map.

Data collection and analysis

This study employed a questionnaire-based survey to evaluate community awareness of the Kipunji in the Mount Rungwe Forest Nature Reserve (MRFNR). Data were gathered from 160 households using a structured questionnaire consisting of closed-ended items designed to capture respondents' knowledge and perceptions of the species. The field data were systematically coded and analysed using the Statistical Package for the Social Sciences (SPSS) version 27.0. Descriptive statistics were first applied to summarise the key characteristics of the data. To identify variables influencing awareness, a binary logistic regression analysis was conducted based on the question: *"Do you know about the Kipunji or have you ever seen or heard of it?"* (responses: yes or no). Furthermore, a Generalised Linear Mixed Model (GLMM) was used to explore the predictors shaping perceptions of the statement: *"Do you consider the Kipunji a pest?"* (responses: yes, no, or not sure). In this model, the respondent's village was treated as the random effect, while age group, gender,

education level, main economic activity, and distance of the village from the forest reserve were included as fixed factors.

RESULTS

Demographic and Socioeconomic Characteristics of Respondents

The study achieved an even distribution of participants across all surveyed villages, ensuring broad representation of local views within the research area. In terms of ethnicity, the Nyakyusa accounted for the majority of respondents (56.9%, $n = 91$), followed by the Kinga group (43.1%, $n = 69$), mirroring the ethnic structure of the surrounding communities. The respondents were drawn from various age brackets, with no single group dominating, thereby allowing for a balanced assessment of awareness and perceptions of the Kipunji across generations. Gender participation was nearly equal, comprising 50.6% males ($n = 81$) and 49.4% females ($n = 79$), reflecting inclusive community engagement and underscoring the importance of integrating both men and

women in conservation education initiatives. In terms of education, most participants had completed primary education (45%, $n = 72$) or attained higher levels (26.3%, $n = 42$), whereas 28.7% ($n = 46$) had no formal schooling. This distribution highlights the need for conservation messages to be adapted to varying literacy levels. Livelihood analysis indicated that the local economy is primarily agrarian, with farming being the dominant occupation (70.6%, $n = 113$), complemented by livestock keeping (29.4%, $n = 47$). These findings are consistent with the subsistence-based economic structure characteristic of rural communities in the Mount Rungwe area.

Factors influencing local awareness of the endangered *Rungwecebus kipunji*

Findings from the logistic regression analysis revealed key factors influencing local awareness of the endangered *Rungwecebus kipunji*. The most significant predictors were the village's distance from the Mount Rungwe Forest Nature Reserve, ethnic background, and age group of respondents. As shown in Table 1, proximity to the forest reserve had a positive and statistically significant effect on awareness levels ($p = 0.016$), suggesting that residents living nearer to the reserve are

more likely to be informed about the Kipunji and ongoing conservation initiatives. This pattern likely reflects greater exposure to conservation activities and interactions with forest resources. Ethnicity was also found to be a significant determinant of awareness ($p = 0.044$) (Table 1). Participants from the Nyakyusa community were almost three times more likely to recognise or have knowledge of the Kipunji compared with their counterparts from the Kinga ethnic group. This result indicates that cultural background and social identity may shape how communities perceive and engage with conservation issues. Age emerged as another significant variable, where respondents aged 26–35 years demonstrated notably higher levels of awareness than those aged above 56 years ($p = 0.013$) (Table 1). This generational difference implies that younger adults may be more responsive to conservation information, possibly due to better access to education or environmental campaigns. Other factors, including gender, education attainment, and main economic activity, did not show statistically significant associations with awareness levels. Nevertheless, these variables remain relevant for understanding the broader socioeconomic dynamics that influence conservation participation (Table 1).

Table 1: The logistic regression results highlight village distance from the forest reserve, ethnic group, and age category as determinants of community awareness in *Rungwecebus kipunji*.

		B	S.E.	Wald	P value
Village distance		0.507	0.211	5.788	0.016*
Tribe	Reference: Kinga	1.039	0.517	4.045	0.044*
Age	Reference: >56				
	18-25	2.553	1.022	6.238	0.013*
	26-35	1.497	0.837	3.201	0.074
	36-45	0.677	0.786	0.741	0.389
	46-55	0.196	0.798	0.060	0.807
Gender	Reference: Female	-0.800	0.444	3.246	0.072
Education level	Reference: Secondary and above				
	No formal education	0.832	0.767	1.176	0.278

	Primary school	-0.013	0.643	0.001	0.984
Economic activity	Reference: Livestock keeping	-0.705	0.668	1.115	0.291

Perception of *Rungwecebus kipunji* as a Pest

Results from the Generalized Linear Mixed Model (GLMM) analysis of the question “*Do you regard the Kipunji as a pest?*” (response options: yes, no, or I don’t know) indicated that education level ($p < 0.001$) and distance of the village from the forest reserve ($p < 0.001$) were the main factors influencing respondents’ perceptions (Table 2). Other variables, including age, gender, and main economic activity, did not exhibit statistically significant relationships with this perception. Respondents with primary education were found to have a higher likelihood

of identifying the Kipunji as a pest compared to those with secondary or tertiary education, suggesting that awareness may not increase uniformly with formal education attainment. This outcome highlights the importance of context-specific environmental education that aligns with the knowledge and experiences of local communities. Distance from the Mount Rungwe Forest Nature Reserve was also a strong predictor. Individuals residing nearer to the reserve were more familiar with the species and, consequently, more likely to form perceptions—positive or negative—about its interactions with human activities (Table 2).

Table 2: Output of the Generalised Linear Mixed Model (GLMM) assessing the multinomial responses to the question “*Do you regard the Kipunji as a pest?*” (yes, no, or I don’t know), with respondents’ villages treated as random intercepts. Significant predictors are marked with asterisks.

		Coefficient	Std. Error	t	P value
Age	Reference: >56				
		1.805E-7	0.0003	0.001	1.000
		-9.504E-9	0.0002	0.001	1.000
		4.415E-8	0.0002	0.001	1.000
		3.055E-9	0.0002	0.001	1.000
Gender	Reference:Female	-9.907E-8	0.0001	-0.001	0.999
Education level	Reference:Secondary and above				
		1.333	0.0002	5951.570	>0.001***
		-5.833E-8	0.0002	0.001	1.000
Economic activity	Reference:Livestock keeping	9.674E-8	0.0003	0.001	1.000
Village distance		0.333	5.0126E-5	6649.882	>0.001***

DISCUSSION

This research examined multiple facets of public knowledge about the Kipunji monkey, encompassing indigenous ecological insights, views on protective measures, and opinions

regarding wildlife governance approaches. Through this exploration, the investigation sought to clarify interactions between protective programs and nearby residents, pinpointing obstacles and possibilities for improved partnership in species safeguarding.

The outcomes extend past the preservation of one primate, delivering broader lessons on biodiversity maintenance and eco-friendly progress. As Tanzania balances growth priorities with ecological integrity, grasping local viewpoints and involvement in protective actions remains essential for enduring environmental health and community welfare (Mabele, 2018; NEMC, 2022).

Worldwide, pinpointing elements that shape public knowledge is vital for crafting customized protective plans that successfully involve residents (Ehardt & Butynski, 2006; Hill, 2002; Mbise *et al.*, 2021; Nekaris *et al.*, 2018). Accounting for location-based, societal, and age-related influences allows protective programs to boost understanding and backing for safeguarding threatened primates like Kipunji (Jones *et al.*, 2005; Nekaris *et al.*, 2018). Such goals may involve improving habitats via educational outreach on species, ongoing tracking, and integrating regulatory measures (Davenport *et al.*, 2008; Ehardt & Butynski, 2006).

Nearness to safeguarded zones markedly affects public knowledge (Ehardt & Butynski, 2006; Htay *et al.*, 2022). Results indicated diverse knowledge degrees across settlements near Mount Rungwe Forest Nature Reserve. Settlements adjacent to the reserve typically displayed elevated knowledge, reflecting the impact of educational protective activities there. In contrast, more distant settlements showed reduced knowledge, pointing to the requirement for focused extension programs in those locations. Moreover, societal elements shape participation in protective activities (Nekaris *et al.*, 2018). Varied societal customs and traditions among groups can alter acceptance of protective communications (Bennett, 2016; Hill, 2002). Age-based variations in protective outlooks further emphasize the value of customized extension activities (Htay *et al.*, 2022; Mbise & Sosiya, 2023). Seniors, drawing from extensive life experiences and wisdom, often exhibit higher knowledge compared to youth (Mbise

& Sosiya, 2023). This may stem from their involvement in pursuits like agriculture, which increases contact with details on surrounding ecosystems and wildlife. Additionally, the investigation stresses education's role in knowledge degrees. Although structured schooling contributed, alternative aspects—such as group involvement and regional expertise—also supported knowledge in those with minimal formal education. Thus, adapting protective learning initiatives to suit varied schooling levels can increase their success.

CONCLUSION

This investigation offers key understandings into factors influencing public knowledge of the threatened *Rungwecebus kipunji* across settlements bordering Mount Rungwe Forest Nature Reserve. The results illuminate primary indicators of knowledge, namely settlement remoteness from the reserve, tribal affiliation, and age group. Remoteness from the settlement proved pivotal in determining knowledge extents, where nearer locations correlated strongly with heightened knowledge. This implies residents closer to the reserve encounter more frequent protective activities and details on forest inhabitants. Additionally, tribal affiliation stood out as a notable indicator, with Nyakyusa members demonstrating superior knowledge over Kinga members, underscoring societal influences on protective knowledge. Age group likewise emerged as a critical indicator, where mid-age adults (26–35 years) showed superior knowledge compared to elders. This highlights generational variances in protective perspectives, with mid-age individuals possibly more open to ecological concerns and programs. Hence, these outcomes stress incorporating location-based, societal, and age-related elements when creating focused protective approaches to elevate public knowledge and endorsement for safeguarding *Rungwecebus kipunji* and its environment. Furthermore, examining effective knowledge programs in higher-awareness settlements could yield useful models for upcoming protective work elsewhere. The investigation

also advises detecting knowledge hurdles in settlements and rolling out tailored educational protective programs for regional groups to elevate comprehension and involvement with the reserve.

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ANALYSIS OF CHANGES IN ELEPHANT POPULATION IN TARANGIRE NATIONAL PARK, NORTHERN TANZANIA

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ABSTRACT

Elephant populations across Asia and Africa have historically declined due to ivory poaching and habitat loss, yet recent evidence suggests population stability or growth in some regions, including Tanzania, despite ongoing threats such as climate change and expanding human populations and its associated stressors. In Tarangire National Park (TNP), anecdotal reports from local communities and wildlife experts indicate increasing elephant numbers, but empirical population data have been limited since 2016. This study estimated the current elephant population in Tarangire National Park (TNP) and assessed its demographic structure and population trends using road count surveys for field data collection, combined with a Bayesian modeling approach, during the wet and dry seasons of 2022 and 2024. The results indicated that the elephant population was predominantly composed of mature individuals ($n = 162$), followed by juveniles ($n = 94$) and infants ($n = 89$). A one-way ANOVA revealed a statistically significant difference among age classes ($F = 3.277$, $p = 0.040$). Subsequent Tukey HSD post-hoc tests showed that the significant difference occurred specifically between the mature and infant age classes, while juvenile and infant classes did not differ significantly, reflecting comparable numbers. Within the mature age class, females outnumbered males (70% vs. 30%), indicating a female-biased, aging population with moderate recruitment and potential future reproductive challenges. Bayesian model analysis integrating the 2014 TAWIRI census density (1.2 elephants/km²) as a prior with 2022 and 2024 field survey data yielded a posterior mean population of 5,799 elephants, with a 95% Highest Density Interval (HDI) of 5,387–6,201 elephants. This updated estimate reflects increased confidence in population size, driven by the 2022 and 2024 field observations. These findings highlight both the successes and emerging challenges in TNP's elephant conservation. While current protection measures have supported population growth, the aging demographic structure underscores the need for continued monitoring, habitat protection, and management interventions to maintain population resilience and ensure long-term survival of this iconic species.

Key words: African elephant, Population dynamics, Tarangire National Park

INTRODUCTION

For centuries, Elephant population trends fluctuated across Asian and African countries, which are known to be the natural distributional

range for Asian, savannah, and forest Elephants (Breuer, 2016). Between the 16th to 19th centuries, the fluctuation in the Elephant population was more negative than positive. Research suggests that the latter was primarily

due to two factors, namely the ivory trade, and later, massive habitat loss (Mumby, 2018; Chase, 2016). Ivory trade exacerbated elephant killings, especially after the increase in ivory prices, in combination with the proliferation of automatic rifles in Asian and African countries. Statistics from the ivory trade kept from 1903 to 1911 revealed that a total of 256 tons of ivory were exported, demonstrating that 1,200 to 1,500 elephants were killed per year in Tanzania only (Baldus, 2001). On the other hand, massive habitat fragmentation limits the required home range of the elephants, which impacts their activity pattern and their entire life cycle such as foraging hours and reproduction (Aarde, 2006). In Uganda for example, between the 1920s and 1990s, the range of land available to elephants dropped from 70% to under 7% (Lee, 2006). An early example of the drastic consequences can be seen in an early study that reported the fall of elephant populations from 3-1.3 million in 1979 to 500,000-700,000 in 1987 (Burton, 2000). This number is, however, relatively higher compared to the recent figures documented in 2007, reporting an estimation of Africa's elephant population to be between 470,000 and 690,000, and by 2014, this estimate decreased by 144,000, with declines of about 8% every year (Ngcobo *et al.*, 2018). Recently researchers have, however, expanded the list of factors causing the decline in Elephant populations to climate change and increased Human-Elephant Conflicts (HEC) (Graham, 2009). Since elephants are among the world's largest mega-herbivores (Shaffer, 2019), climate change poses a major threat to foraging materials and water availability, and this not only hinders the elephant's reproduction cycle, and increases the mortality rates, but also influences elephant's mobility leading to increased HEC (Mumby, 2018). Recently human-wildlife conflicts have also proven to be a potential threat to the survival of elephants and therefore impact the number of both Asian and African elephants.

While the above documentations report an alarming decline in the Elephants population resulting from various factors, contemporary

research work believes otherwise and has documented a stable to increasing state of the elephant population across continents mainly due to increased anti-poaching measures accompanied by elimination of or rigorous control over the ivory trade (Douglas-Hamilton, 2008). A study in Nepal suggested an increase in the elephant populations (Smith, 1992), and since the 1990s countries such as Zimbabwe, Botswana, Namibia, and South Africa have documented stable or increasing Elephant populations. The latter is primarily a result of proper management and the involvement of local communities in decision-making and revenue sharing (Jachmann *et al.*, 1995). Similar trends in elephant population increase have been reported in East African countries like Kenya and Tanzania (Smith, 2009). From 2002 to 2006, Kenya experienced an increase in elephant populations from 22,036, to 23,353, and in Tanzania, elephant populations have risen from 55,000 to 136,000 in 2016 (Mariki, 2014). In the Tarangire national park a study by Foley & Faust, (2010) reports the Elephant's average annual growth rate to be 7.1% (range 2.0 - 16.9%). The belief that elephant populations have increased in recent years is now strongly linked with increased human-elephant conflicts mitigation measures (Mariki *et al.*, 2015). In Tanzania, the above claims have become very popular among scholars, elephant-centered non-governmental organizations, and local populations. Mariki *et al.*, (2015) reported as quoted "Many of the interviewees in West Kilimanjaro said that the elephant population had considerably increased during the last few years". Another study by Foley & Faust, (2010) in the Tarangire National Park, asserted similar trends in the elephant population, especially in females and dependent calves, and their number doubled over a decade. However, the above speculation not only never took past elephants' statistics into account but also lacks the current scientific-based evidence to support such claims. The former is because studies on elephant populations, structure, and dynamics in Tanzania were done ten years ago. Moreover, the few existing old studies on the elephant population in Tanzania failed to embrace

the use of modern mathematical models that considers prior and posterior data to conclude on elephants' population trends, that is to answer the question of what magnitude have elephants increased or decreased as compared to the past.

This study, therefore, aims at estimating Elephant population by using King's population estimate formula to compute the current elephant population density which was used to estimate the current elephant's population in Tarangire NP and the Bayesian Model (BM) to model The Tarangire's past elephant population secondary data to current data collected during dry and wet seasons of the year 2022 and 2024 to investigate the changes in elephant population in Tarangire National Park, The World's most elephant populated protected area. It is hypothesized that the past elephant population in TNP will not significantly differ from the current elephant population in TNP. This study will prove or reject emerging reports of elephant population increases. This was important for the ongoing global conservation efforts of the *Loxodonta Africana*, one of the keystones and ecological engineers of the African savanna that is currently recognized as endangered by the International Union for Conservation of Nature (IUCN, 1995). Furthermore, the findings of this study will shed some light on whether the elephant population is the primary reason for increased

human-elephant interaction and conflicts, therefore informing conservation key actors on the most important drivers for increased human-elephant interaction and conflicts. This is especially necessary when considering that as these conflicts arise, the results have grim consequences for both humans and elephants, the worst outcome being death.

MATERIALS AND METHODS

Study area

Tarangire National Park covers about 2850 sq. km (Laizer, 2014), with a latitude of 3°40'–5°35' S and a longitude of 35°45'–37°00' E (Ishengoma, 2007) and is situated in the northern part of Tanzania (Fig. 1) The area sits at a mean elevation of 1200 m (Gereta, 2004). Precipitation at TNP follows a bimodal pattern and can be seen in two ways, long and short rainfall. Specifically, from March to May it is long rainfall, and from November to December, it is short rainfall. From June to October, as well as January to February, it is the dry season (Mkonyi, 2017). The anticipated annual rainfall is about 650mm (Ishengoma, 2007) and the hottest temperatures in TNP occur from December to February, with an average high temperature of 27°C. In contrast, colder temperatures occur in June and July, with an average low temperature of 16°C (Porowka, 2003).



The primary vegetation type at TNP is arid savanna vegetation, which includes riparian woodland and grassland, where the grassland contains short trees that are sparsely distributed. Similarly, the riverine grassland also contains baobab trees. The riparian woodland is primarily dominated by *Acacia tortilis*, and other types of vegetation like *Acacia drepanolobium* woodland, and *Combretum delbergia* woodlands (Abukari, 2020). Moreover, large seasonal swamps (Ishengoma, 2007), which like other sources of water, the most prolific being the Tarangire River, are essential for supplying water to a multitude of species during the dry season. There are microphyll savannas located along the riverine floodplain, and deciduous woodlands

Community

Tarangire National Park extends across three administrative districts: Simanjiro, Monduli,

and Babati. In 2012, it was reported that collectively the area had an average population density of 68.6 persons/km² (Akubari, 2020). The park is surrounded by 42 communities including Kimotorok, Mamire, Mswakini Juu (Akubari, 2018), Loiborsoit, Emboreet, Terrat, and Sukuro, to name a few. Additionally, the main ethnic groups living around TNP are Maasai, Iraqw, and Rangi, where the dominant livelihoods are pastoralism and agropastoralism (Baird, 2013).

Data collection methods

Road strip count

Road strip count is one of the most common methods used for estimating the population of wild animals (Lewis, 2022). Studies suggest that the road strip count is one of the more precise methods for estimating wildlife population (Dasmann, 1962), because population estimates from the road strip counts are usually very close to the true population. As such this study adopted the road strip count method to estimate the current elephant population in the Tarangire National Park. The road strip count involved driving on a network of roads in the area investigated to survey wildlife from the vehicles (WWF, 2000), data collected included the number of individual elephants sighted, the perpendicular distance at a 90° angle from the vehicle, age categories (young, juvenile and adults), habitat type, and metadata. The observations were made on either side of the road, and the Effective Strip Width (ESW) was determined by the perpendicular distance, but did not exceed 1000m, and the traveled distance was recorded by the odometer.

Secondary data

To inform the prior in the Bayesian analysis, the study utilized secondary data on elephant populations in the study area, particularly the population density estimates published by Bluwstein (2018). These estimates were originally derived from the 2014 TAWIRI census of the species.

Study design

This study adopted an observational study design whereby the subject recorded the animal of interest that was encountered during the survey that met the inclusion criteria in terms of observation range, which allowed for population estimates, among other analyses.

Sampling techniques

The sampling techniques and procedure included both stratified sampling and convenient observations. Specifically, TNP was divided into 2-3 blocks based on the already established road network, key resources governing the distribution of the elephant's population such as swamps and rivers, and habitats, where vehicles surveyed each block. Vehicles contained an observer, recorder, and driver in each, following the network of roads in the park in each of the blocks. The latter called for convenient sampling, for only the sighted elephant individuals were recorded for analysis.

Sample size

Since this study employed a convenient sampling technique for data collection the exact sample size was difficult to detail. However, the study covered a large portion of TNP to maximize the sample size and therefore ensure that the collected data was representative of the true elephant population currently residing in the TNP during the dry season of 2022.

Data analysis

Data were analyzed using a Bayesian model to estimate elephant population size and trends from 2014 to 2022. In addition, frequency counts were computed, and a one-way Analysis of Variance (ANOVA) followed by a Tukey Honest Significant Difference (HSD) post-hoc test was performed to assess differences among age-class categories. We conducted a Bayesian analysis to estimate the elephant population in the Tarangire National Park by integrating

historical survey data from TAWIRI (2014) with filed count from a 123.7 km² sample within the 2850 km² study area.

Bayesian framework

The prior distribution was based on the TAWIRI 2014 density estimate of 1.2 elephants/km², corresponding to a mean population of 3420 individuals. A normal prior standard deviation of 10% of the mean (342 elephants) was used to account for uncertainty in the prior estimate. This prior distribution reflects a credible population value before considering current field observation and running the posterior distributions. The likelihood function was derived from the observed count of 350 elephants in the sampled area. A Normal Poisson distribution was applied to generate a smooth likelihood function across the population range of 4500-7500 elephants. This likelihood represents the probability of observing the field count given each potential population size. The posterior distribution was computed by multiplying the prior and the likelihood and normalizing the resulting product. This posterior respected an updated population estimate that combines the prior knowledge with filed data. The 95% highest density interval (HDI) was calculated to quantify uncertainty in the estimate. Population projection was done using the posterior mean as a baseline, we projected the elephant population trends from 2014 to 2022, assuming an annual growth rate of 2%. The 95% HDI from the posterior was scaled accordingly to generate credible intervals for each year, providing a probabilistic range for the population trends over time.

Regarding the demographic structure, the elephants were classified into three categories: infants, juveniles, and mature individuals. The infant and juvenile categories were determined based on the height of an individual relative to an adult female elephant. Elephants that could walk underneath an adult female were classified as infants, while those that could not pass underneath but did not exceed the height of her shoulder were classified as juveniles.

The frequencies of individuals in each age class were first summarized to describe the population structure and sex ratio. To statistically evaluate whether the sizes of these age groups differed significantly, a one-way Analysis of Variance (ANOVA) was performed, with age class as the categorical predictor and the number of individuals per class as the response variable.

Following the ANOVA, a Tukey Honest Significant Difference (HSD) post-hoc test was applied to identify which specific age groups differed from one another. This test adjusts for multiple comparisons and provides pairwise comparisons between the three age categories. Through this approach, differences in population size between infants, juveniles, and mature elephants were assessed, allowing us to determine whether observed variations in the age structure were statistically meaningful.

RESULTS

Demography of Elephants population in Tarangire National Park

The age-structure assessment of the elephant population in Tarangire National Park shows a population dominated by mature individuals ($n = 162$), followed by juveniles ($n = 94$) and infants ($n = 89$) (Fig.2). This distribution indicates that the population is currently skewed toward older age classes. Within the mature cohort, females constitute the majority 70% ($n = 113$), while males make up 30% ($n = 49$). To statistically assess differences among age categories, a one-way ANOVA followed by a Tukey HSD post-hoc test was performed. The results show a significant difference between the mature and infant groups ($p = 0.040$), confirming that the number of mature elephants is substantially higher than that of infants. In contrast, no significant difference was detected between juvenile and infant groups, indicating that these younger age classes occur in relatively similar numbers.

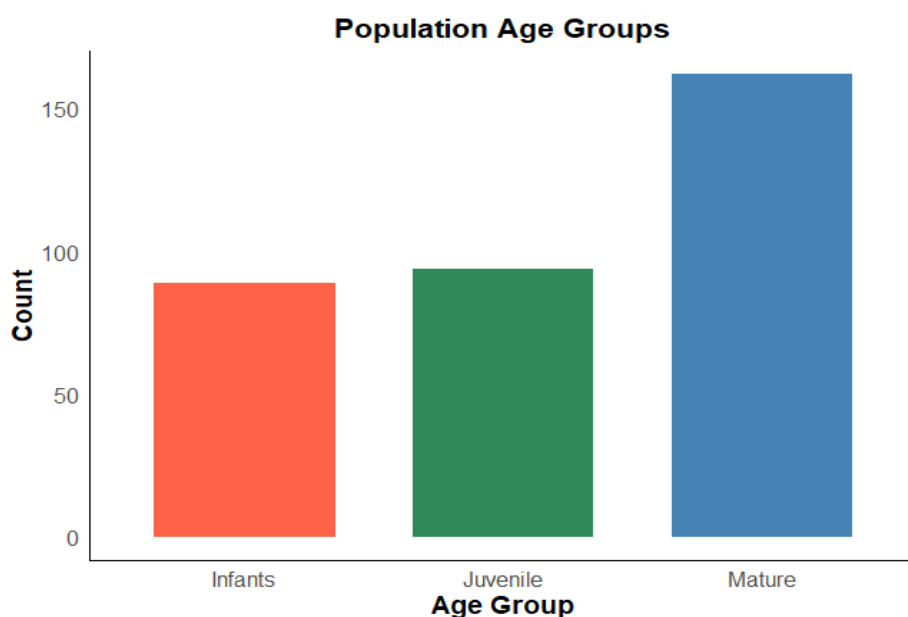


Figure 2: Demographic structure of elephants in Tarangire National Park based on field classifications of infants, juveniles, and mature individuals derived from height comparisons with adult females

The population estimate in Tarangire National Park during the

oResults from the Bayesian model analysis integrating prior density information from the 2014 TAWIRI census and the 2022 and 2024 field survey data produced a well-defined posterior distribution of elephant population size within the study area. The prior distribution, derived from the 1.2 elephants/km² estimate reported in 2014, was centered around the expected population size of approximately 3,420 elephants and displayed moderate uncertainty. In contrast, the likelihood function, informed by the 2022 and 2024 field sample count of 350 elephants observed within 123.7 km², was strongly skewed toward higher

population values, reflecting the higher density implied by the field observations.

The combination of 2014 prior and 2022 and 2024 likelihood yielded a posterior distribution with a clear unimodal peak (**Fig. 3**). The posterior mean, population size was 5,799 elephants, and the posterior mode occurred at the same approximate value, indicating strong agreement between the most likely and average estimates. The 95% Highest Density Interval (HDI) ranged from 5,387 to 6,201 elephants, representing the narrowest interval containing 95% of the posterior probability mass. This interval reflects reduced uncertainty relative to the prior alone and highlights the constraining influence of the 2024 field observations.

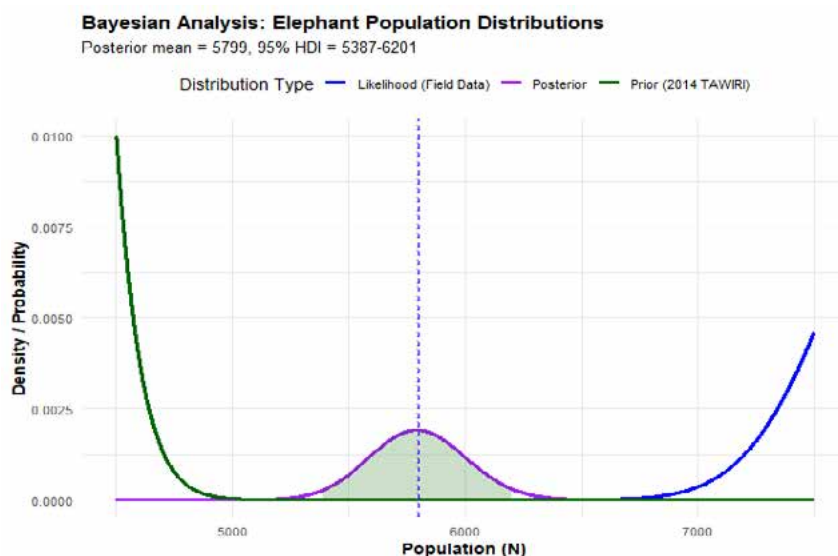


Figure 3. Bayesian prior, likelihood, and posterior distributions of elephant population size in the study area. The dark green curve represents the 2014 TAWIRI prior based on a density of 1.2 elephants/km². The blue curve shows the likelihood derived from the late dry season of 2022 and wet season of 2024 field count of 350 elephants from 123.7 km². The purple curve depicts the posterior distribution, with the shaded region indicating the 95% Highest Density Interval (HDI). The posterior mean population estimate was 5,799 elephants (95% HDI: 5,387–6,201).

The Bayesian analysis, incorporating the 2014 TAWIRI prior and the 2022 and 2024 field survey likelihood, produced posterior population estimates for elephants for each year between 2014 and 2022. Year-specific posterior means were projected forward under an annual growth rate informed by posterior uncertainty. The resulting population trajectory showed

a steady increase over the nine-year period (Fig.4). Intermediate annual estimates were: 5,741 elephants in 2015, 5,856 in 2016, 6,092 in 2018, 6,339 in 2020, 6,595 in 2021, and 6,726 in early 2022. The 95% credible intervals for annual estimates remained relatively narrow across all years, indicating consistent confidence in the population projections.

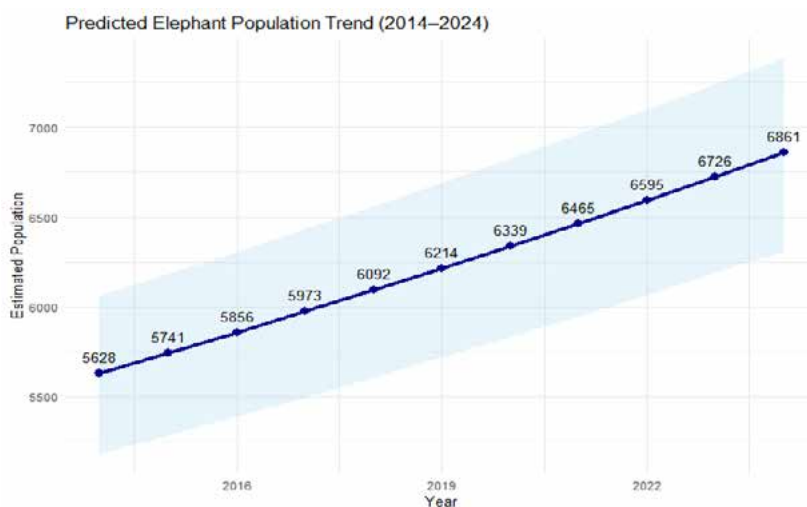


Figure 4: Estimated elephant population size in Tarangire National Park, showing an increase from approximately 5,628 elephants in 2014 to 6,861 elephants in 2022

DISCUSSION

The demographic structure of the African elephant population in Tarangire National Park is dominated by mature individuals, while juveniles and infants constitute a smaller proportion of the population. This pattern may be partly explained by historical poaching, which disproportionately targeted adult elephants and subsequently reduced the survival of orphaned calves, ultimately suppressing overall population growth. Similar argument was reported by Parker *et al.* (2021), who found that poaching not only removes reproductive adults but also indirectly decreases population growth by lowering orphan elephant survival rate, thereby compounding long-term demographic declines. Moreover, a female-biased sex ratio was also observed among mature age classes. This pattern is consistent with the known social structure and life-history characteristics of elephants, whereby males disperse from family groups upon reaching sexual maturity (approximately 10-14 years) and are therefore less frequently recorded in herd-based population surveys (Schulte, 2000; Athira & Vidya, 2021). In addition, historically higher mortality among males partly due to selective poaching for ivory may have contributed to the current skewed sex ratio (Campbell-Staton *et al.*, 2021; Mondol *et al.*, 2014).

The relatively low proportion of infants compared to mature individuals suggests moderate recruitment within the population. Although the population appears stable and increasing, the current age distribution indicates that reproductive output may not be fully keeping pace with the number of aging adults. If low recruitment persists over time, future population growth may slow, and the population could face reduced reproductive momentum. Continued monitoring of infant and juvenile survival is therefore essential to understand whether the observed age structure reflects natural variation, survey timing, or emerging demographic constraints.

The results of this study (Fig. 3 and 4) reveal that the elephant population in Tarangire National Park has increased substantially and now exceeds previous estimates. This growth is likely linked to strengthened anti-poaching measures and improved protection across northern Tanzania's protected areas. Reduced poaching pressure directly lowers adult mortality, while decreased disturbance in breeding herds may indirectly enhance reproductive performance by improving calf survival and reducing stress-related reproductive suppression. These combined effects offer a plausible explanation for the population increase observed in this study.

These findings align with broader regional patterns. Morrison (2015) documented a dramatic recovery of elephants in northern Tanzania following severe declines in the late 1970s, with numbers rising from fewer than 500 to more than 7,500 individuals by 2014 after improvements in law enforcement and habitat conditions. Similarly, Valeix (2008) and Hariohay *et al.* (2023) emphasize the importance of strengthened protection and human–elephant coexistence strategies in facilitating population recovery across elephant landscapes. Additionally, Kalwij (2010) reported annual population growth rates of approximately 5% in areas where poaching was reduced following the international ivory trade ban further supporting the interpretation that enhanced protection has played a central role in the recent recovery of elephants in Tarangire.

Overall, the demographic patterns documented in this study reflect a recovering elephant population benefiting from sustained conservation interventions. However, the age structure underscores the need for continued monitoring to ensure that recruitment remains sufficient to sustain long-term population growth.

CONCLUSIONS AND RECOMMENDATIONS

The elephant population in Tarangire National Park is currently dominated by mature individuals, with juveniles and infants occurring in lower proportions, suggesting moderate recruitment within an otherwise increasing population. Bayesian estimates indicate that the population has more than doubled since 2014, reflecting strong recovery attributed to improved protection and sustained conservation interventions. Although the population is stable and growing, the age structure highlights the need for continued monitoring to ensure that recruitment remains sufficient to maintain long-term population resilience.

To sustain this positive trajectory, it is recommended that demographic monitoring be strengthened through regular age-structured assessments that focus on calf and juvenile survival, and that these data be integrated with spatial information on elephant movements to better understand how habitat use influences reproductive success. Targeted efforts to improve calf survival particularly in areas where recruitment appears low should be prioritized, alongside the continued reinforcement of anti-poaching and law-enforcement measures that have underpinned recent population recovery. Furthermore, expanding community-based conservation initiatives, including early-warning systems and human elephant coexistence strategies, will be essential for reducing conflict and maintaining local support for conservation. Together, these actions will help ensure that the growing elephant population remains demographically stable, ecologically resilient, and capable of sustaining long-term recovery.

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TERRESTRIALITY IN ARBOREAL RED-TAILED MONKEYS (*CERCOPITHECUS ASCANIUS*) IN ISSA VALLEY, WESTERN TANZANIA.

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ABSTRACT

Red-tailed monkeys (*Cercopithecus ascanius*) are primarily arboreal, although sometimes exhibit terrestriality in response to local ecology, especially food types and distribution. How frequently terrestriality appears in red-tailed monkeys and the factors that influence this behavior, are not well understood. Given the risks that primates face when on the ground, we investigated the drivers of terrestrial behavior in a habituated group of red-tailed monkeys that live in a mosaic woodland habitat in the Issa Valley, Katavi region. We hypothesized that seasonality and vegetation would impact terrestrial behavior. Specifically, we predicted that individuals would be more terrestrial during in the dry season when water and arboreal foods are scarce. Moreover, we expected to find an interaction between vegetation type and activity, with more terrestrial behaviors found in the woodland during travel through forest patches due to higher predation risk in more open areas and feeding due to presence of mushroom while more rest and social activity being observed in the forest due to minimal risks from predators. We collected data from 2016 to 2022 (n=3177 observations) on a single red-tailed troop, including 332 observations of terrestrial behavior. The results indicate that red-tailed monkeys terrestriality is influenced by activity, vegetation and season. The red-tailed monkeys tend to rest arboreally and during the wet season reduced terrestriality, except in riparian forest where the effect is reversed. In the dry season, red-tailed monkeys spent about the same time feeding arboreally as terrestrially, this difference was largely driven by mushroom consumption, an exclusive wet-season resource, which was entirely accessed terrestrially. Our results have implications for understanding the drivers of terrestriality and especially the behavioral responses of an otherwise arboreal species living in a savanna woodland environment.

Keywords: *Arboreality, Red-tailed monkeys, seasonal, terrestriality, vegetation*

INTRODUCTION

Primates exhibit the widest diversity of arboreal adaptations among vertebrates, and arboreality is considered the ancestral condition for the order (Cartmill, 1974; Fleagle & Lieberman, 2015). Over the course of their evolutionary history, primates including papionins, hominins, and some guenons, have repeatedly left trees to adopt primarily terrestrial lifestyles (Fleagle and Lieberman, 2015), and such a transition is thought to have played a central role in human evolutionary history (Potts, 1998; Richmond *et al.*, 2001; Kingdom, 2003; Patel *et al.*, 2009). This transition has been linked to predation risk (Isbell, 1994; Jaffe & Isbell,

2009), food and water availability (Orkin & Pontzer, 2011; Gilbert & Stouffer, 1989), and thermoregulatory demands (Wheeler, 1984).

Comparative research across and within primate species that vary in their patterns of ground use can help inform the selective pressures that shape terrestriality across species (Fleagle, 2015). For example, increased terrestriality in typically arboreal primates is associated with multiple ecological drivers including canopy cover, predation risk, maximum temperature, precipitation, primate species richness, human population density, distance to roads and species-specific traits including body mass, group size, and degree

of frugivory (Eppley *et al.*, 2022; Estrada and Marshall, 2024). Whilst most studies compare terrestrial versus arboreal species, within-species variation can also provide a complementary perspective (Eppley *et al.*, 2016; 2022). For example, capuchins (Wright *et al.*, 2019), lemurs (Eppley *et al.*, 2016), and guenons (Pozo-Montuy and Serio-Silva, 2007) alter ground use depending on canopy connectivity, food distribution, and predation suggest that within-species variation provide insights in to flexible terrestrial behavior and adaptation to environmental pressures. Here, we investigate the dynamics of ground use in red-tailed monkeys (*Cercopithecus ascanius*) living in the Issa Valley, western Tanzania—a savanna-woodland mosaic ecosystem with limited arboreal cover and elevated terrestrial risks in to how these monkeys adapt to their environment and respond to ecological pressures.

The adaptability of guenons (genus *Cercopithecus*) to diverse habitats, from rainforests to savannah woodlands, offers insights in to primate terrestriality, particularly for species like the red-tailed monkey (Kingdon, 2007). Studying guenons' ecological flexibility and evolutionary history can help understand how primates develop terrestrial behaviors in response to environmental pressure (Elton, 2006). This knowledge can inform our understanding of red-tailed monkeys' adaptability to different environments and their potential for terrestriality

Red-tailed monkeys (*Cercopithecus ascanius*) are known to inhabit a range of environments beyond dense forest, including more open and savannah-like habitats, where they exhibit flexibility in their terrestrial behavior (Almeida-Silva *et al.* 2005; Gilbert and Stouffer 1989; Pozo-Montuy and Serio-Silva 2007). Research in to guenons in mosaic landscapes can reveal crucial information about their ability to cope with habitat alteration, resource utilization in disturbed environments and potential behavioral plasticity (Estrada, 2024).

Red-tailed monkeys at the Issa Valley, in Tanzania, inhabit a mosaic environment of miombo woodland interspersed with thin, riverine forest strips (McLester *et al.*, 2019). Comparisons of Issa red-tailed monkey diet and ranging compared to those in large contiguous forests in Uganda revealed that Issa red-tailed monkey exhibit a more generalized and flexible diet, incorporating a wider variety of food items available across these distinct habitat types and characterized by larger home ranges and more daily travel distance across between fragmented forest patches and exploit resources distributed across the woodland and riverine areas (McLester *et al.*, 2020). This contrasts with population in large, contiguous Uganda forests, which tend to have a more specialized diet, often focusing on abundant fruit resources and typically have smaller home ranges due to the consistent availability of resource within their forest environments (McLester *et al.*, 2019).

Forest structure may play an important role in terrestrial behavior, with some arboreal species spending substantial time on the ground. For instance, capuchins in Eastern Brazil's tropical dry forests spend up to 44% of their time terrestrially (Estrada & Marshall 2024). Similarly, red-tailed monkeys in certain habitats have been observed adapting their behavior to suit local ecologies including spending time on the ground, particularly in areas with limited arboreal cover or fragmented forests (Pozo-Montuy and Serio-Silva 2007). These observations highlight how vegetation types and forest structure impact primate ranging and feeding activities, leading to varying degrees of terrestrial adaptation even within typically arboreal groups. For example, studies have shown that primates including capuchins, lemurs, and guenons may increase their terrestrial activity when foraging for specific food items found on the forest floor or when canopy connectivity is limited (Georgie *et al.*, 2016).

In contrast, open woodlands and savannah-woodland mosaic habitats, characterized by

lower tree density and increased seasonality, present more spatially and temporally sporadic food sources (Piel *et al.*, 2015). Chimpanzees in Issa therefore increase overall terrestriality in open woodland compared to closed forest, but do not necessarily spend more time locomoting on the ground compared to chimpanzees in more forested (Drummond *et al.*, 2022).

Animal movement and habitat use have long been linked to the distribution and abundance of food resources. For arboreal primates, utilizing novel terrestrial resources may be advantageous. It has been argued that the shift to terrestriality from an arboreal lifestyle among catarrhines was likely initiated to exploit terrestrial food resources (reviewed in McCrossin *et al.* 1998). Given the wider spatio-temporal distribution of resource and predation risks in these environment (Estrada & Marshall. 2024), the quality and diversity of available resource creates a crucial role for red tailed monkey terrestriality. For example, chimpanzees in savannah-woodland at the Issa Valley, Tanzania, travel further and preferentially exploit woodland food searching during dry seasons when fruit is most abundant in woodland compared to other vegetation types (Hernandez-Aguilar, 2009; Piel *et al.*, 2017). Red-tailed monkeys are omnivorous and diverse in their diet, consuming a wide range of plant species, invertebrates, and sometimes even meat (Lambert, J. 2002, Hohmann, 2009, Marshall., *et al.* 2017). However, the specific composition of their diet often shifts with the seasons, reflecting the availability of different food items as fruits availability decline, particularly during drier period, primate increasing their consumption of other food items for instance, they may turn winder variety of insect including caterpillars and ants (Isbell.1998, Harris & Chapman, 2007). Seasonal variation in terrestriality at Issa may be influenced by food and water availability and distribution, with similar trends shown in Costa Rica, observed Mantled howler monkeys (*Alouatta palliata*) utilize a ground water source particularly during periods when arboreal water source might be scarce (Gilbert and Stouffer

1989). Likewise, studies in the fragmented landscape of Tabasco Mexico where Mexican Black Howler monkey (*Alouatta pigra*) where observed gaining access to food sources on the ground frequently to forage when arboreal food are scarce (Pozo-Montuy and Serio-Silva, 2007).

Arboreal primates are also more vulnerable to predation in open forest or at forest edges, where they are more exposed and visible, compared to closed canopy forests (Jaffe and Isbell 2009., Fornof *et al.* 2023). At Issa valley, forest configuration influences the terrestrial activities which means that red tailed monkeys may not travel to distant patches as crossing more open habitats may pose a higher predation risk. Red-tailed monkeys in the Issa Valley live sympatrically with several species of predators including Leopards, chimpanzees, crowned-hawk eagles (Piel *et al.* 2017). Studies have reported that leopards in savannah-woodland mosaics hunt most in intermediate vegetation coverage despite prey being most abundant in dense vegetation (Balme *et al.*, 2007).

Primate associations with other species, such as ungulates, often appear to be related to food acquisition (Morgan-Davies, 1960) or anti-predation benefits, particularly in terrestrial environments where predation pressure can be high. For example, some terrestrial primates forage alongside Kirk's dik-diks (*Madoqua kirkii*), potentially gaining advantages from increased vigilance or foraging efficiency (Danchin *et al.*, 2004). More recent studies have also explored there mixed-species association in terrestrial habitats, highlighting their complexity and variability (Fichtel, 2012).

The anti-predator response in primates, such as vigilance, hiding, aggregation, and alarm calling, is more closely linked to perceived risk rather than directly affecting time spent terrestrially. Studies like Herzog *et al.* (2020) work on vervet monkeys demonstrate that these primates adjust their behavior based on perceived predation risk, particularly in open or high-risk environments. Similarly, research

at sites like Issa, where chimpanzees and other primates exhibit cautious terrestrial behavior, suggest that perceived risk influenced their use of terrestrial habitats. For example, Issa red-tailed monkeys have been observed using isolated forest patches that are only accessible by travelling terrestrially through woodland, visually scanning for threats in open areas before quickly moving between patches (Fornof *et al.* 2023) indicating that their behavior is shaped by the perceived risk of predation rather than an inherent avoidance of terrestrial environments.

The Issa valley serves as a unique habitat for red-tailed monkeys, however there is limited understanding and comprehensive research about the factors driving red tailed monkey terrestriality. Despite extensive work into red-tailed dietary ecology and polyspecific associations (Struhsaker, 1978; Gathua, 2000; Tweheyo and Obua, 2001; Chapman *et al.*, 2002; Bryer *et al.*, 2015; Struhsaker, 2017), the terrestriality of red-tailed monkeys remains understudied. We tested three hypotheses: (H1) terrestriality differs between miombo woodland and riparian forest due to canopy openness and predation risk; (H2) terrestriality increases in the dry season when water and arboreal foods are scarce; and (H3) activity

type predicts terrestrial use differently across vegetation types, with feeding and traveling more common in woodland and social/resting more common in riparian forest.

METHODOLOGY

Study site

We collected data on single troops of red-tailed monkeys from the Issa Valley, Tanzania, where they have been studied since 2012 (McLester, Brown, *et al.*, 2019). The Issa Valley is located in western Tanzania in Katavi region (Fig. 1). The region is characterized by two distinct seasons: wet from November to April and dry (<100mm monthly rainfall) from May to October (Piel *et al.*, 2017) and the entire area is characterized as a savanna-mosaic, with valleys containing evergreen riverine forest and separated by steep slopes with expansive flat plateaus of miombo woodland, rocky outcrops and seasonally inundated grasslands (Drummond-Clarke *et al.* 2022). Issa is home to numerous predator's species such as leopards (*Panthera pardus*), lions (*P. leo*), spotted hyenas (*Crocuta crocuta*), wild dogs (*Lycaon pictus*), crowned-hawk eagles (*Stephanoaetus coronatus*), and various small carnivores (Piel *et al.* 2019).

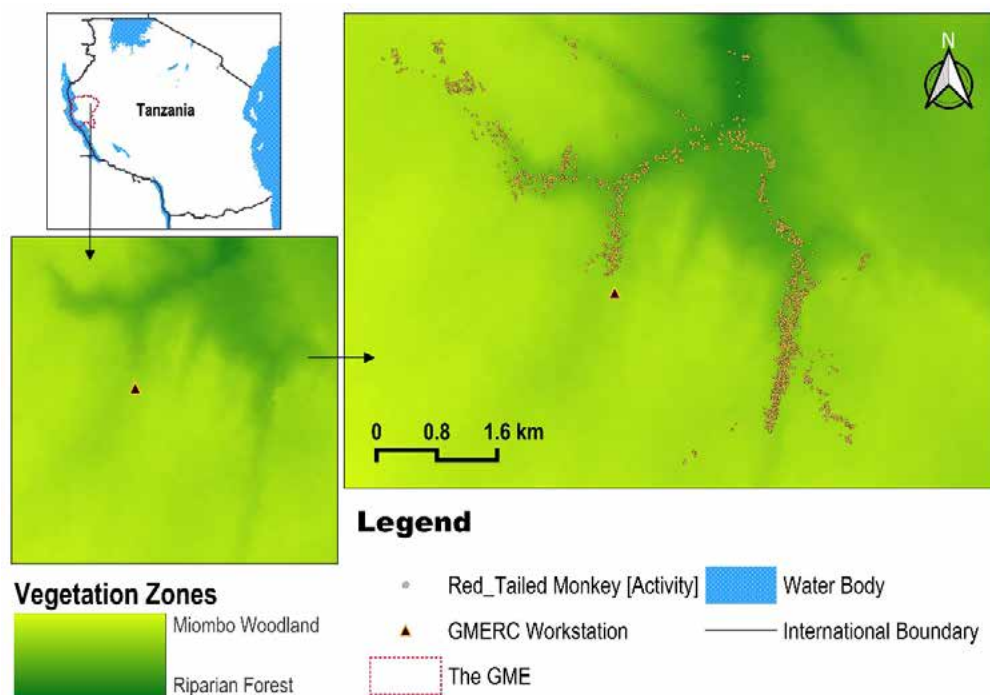


Figure 1. The map show distribution of activities in red-tailed monkey within two vegetation type (miombo woodland and riparian forest) at Issa Valley, Western Tanzania

Data collection

We collected data from January 2016 to December 2022. The study group was followed between four and eight days/month. We collected data from sunrise (~7h), around the point the group left their sleeping site, until they arrived at the next sleeping site (~19h). Scan samples and focal follows were taken at every 5 minute intervals for all visible individuals, recording time, location, vegetation type (classified as either riparian forest or miombo woodland), activity, and substrate (Strier 1986; Dias & Strier 2003). Activities were categorized as resting, traveling, feeding, vigilance, social, or drinking. Feeding was further coded by food type (fruits, new leaves, mature leaves, flowers, mushrooms, or insects). Social interactions included play, embraces, sexual inspections, copulations, and chases. Substrate was only recorded as terrestrial if no limbs were on arboreal substrates. In total, we collected 3177 observations scans including 332 observations of terrestrial behavior in both vegetation.

Data Analysis

Statistical Modeling

A generalized linear mixed model (GLMM) with a binomial error distribution and logit link function in R V4.4.1 was employed to investigate the effects of activity, vegetation, season and their interactions on the probability of terrestriality in red-tailed monkeys. The model controlled for non-independence by incorporating date as a random intercept, with a variance of 0.96. Post-hoc contrasts tested specific predictions (woodland vs riparian; dry vs wet season).

RESULTS

Factors Influencing Red-tailed Monkey Terrestriality

The GLMM revealed significant main effects of activity, vegetation, and season on terrestriality. Specifically:

Activity type effect: The probability of terrestriality was significantly lower when monkeys were resting compared to feeding ($\beta = -1.163$, $p = 0.0024$), indicating a strong tendency to rest arboreally (Fig. 2).

Seasonal effect: The wet season significantly decreased the probability of terrestriality compared to the dry season ($\beta = -0.783$, $p = 0.0095$). In the dry season Riparian forests show higher terrestriality across all activities than miombo woodlands, except during traveling, where miombo woodland terrestriality is slightly more common. In the wet season miombo woodland, terrestriality is lowest during rest and increases slightly

during feeding and travelling. In response to both seasons, terrestriality peaks during social activities in Riparian forests and miombo woodland (Fig.2).

Vegetation structure and Season Interaction effect:

A significant interaction between riparian forest and the wet season ($\beta = 0.794$, $p = 0.014$) suggested that the suppressive effect of the wet season on terrestriality was nullified or reversed within riparian forest habitats. Across most activities and both seasons, red-tailed monkeys in Riparian forests tend to have a higher predicted probability of being terrestrial compared to those in miombo woodlands (Fig.2).

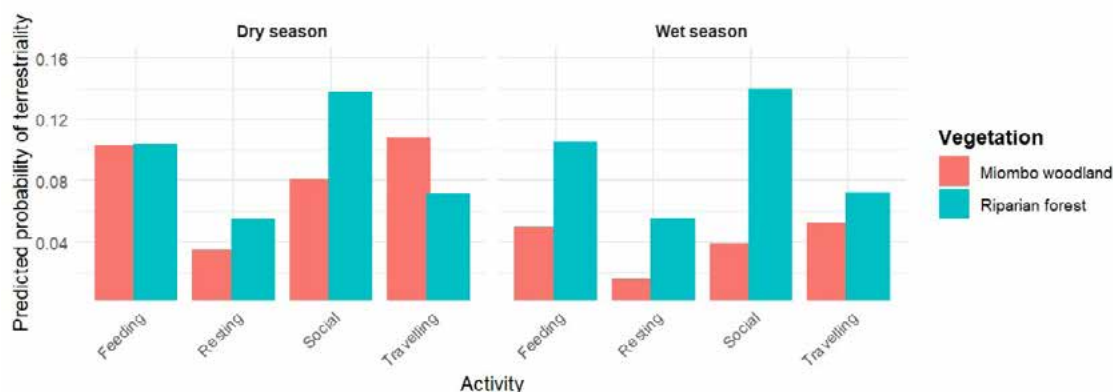


Figure 2. The predicted probability of terrestriality in red-tailed monkeys across different activities where drinking merged with feeding, vigilance merged with resting, and social encompasses fighting, grooming, sex and play (feeding, resting, social, traveling) in miombo woodlands (red) and Riparian forest (green) during dry and wet seasons.

DISCUSSION

Red-tailed monkeys (*Cercopithecus ascanius*) in the Issa valley exhibit distinct habitat utilization patterns. They consistently use riparian forests as their primary habitat, spending most of their time in trees for food and predator protection (McLester *et al.*, 2019, Fornof *et al.* 2023). Woodlands, conversely, are exploited opportunistically for specific food resources (McLester *et al.*, 2019). This adaptation is driven by the lower proportion and quality of riparian forest cover in these environments, which leads to a wider distribution of resources

(Tapper *et al.*, 2019, McLester *et al.*, 2019). Red-tailed monkeys in the Issa Valley, we're observed frequently descending to the ground for resting, traveling, foraging, and socializing. This terrestrial behavior is considered rare for arboreal monkeys and is likely an adaptation to the mosaic environment. A study on howler monkeys (*Alouatta spp.*) in a dense forested environment found that spent most of their time in the canopy, with limited terrestrial behavior (Bez, 1993). In contrast, capuchin monkeys in a mosaic environment exhibit more terrestrial behavior (Wright *et al.*, 2019). This comparison highlights how different environments can

influence primate behavior driving adaptations towards terrestriality.

Our results indicate that Issa red-tailed monkeys flexibly utilize ground use in response to local ecology, especially food types and distribution. Wet-season mushroom (Schulze *et al.* 2025) and invertebrate availability likely promote terrestrial feeding and travelling during dry-season across woodland links forest patches (Eppley *et al.*, 2016, McLester *et al.*, 2019). This supports hypotheses that resource distribution and predation risk shape arboreal primates' terrestriality (Isbell, 1994; Orkin & Pontzer, 2011; Eppley *et al.*, 2022). The higher vigilance and reduced feeding in woodland are consistent with anti-predator strategies (Herzog *et al.*, 2009). By examining within-species variation, our study contributes to understanding repeated evolutionary shifts to terrestriality in primates (Fleagle & Lieberman, 2015). The findings help explain increased terrestrial behavior in Issa red-tailed monkeys, by showcasing a new terrestrial foraging niche developed in response to the environmental factors examined in this study. Numerous studies have revealed arthropod prey to be an important part of their diets. In Kakamega forest, the diet of red-tail monkeys included 25.1 % of insects (Cords, 1986). A similar study showed that the average amount of time a red-tailed monkey spent feeding on insect in Kakamega forest and Kibale National Park was 28 % and 21 % respectively (Chapman *et al.*, 2002).

Our results also support previous studies that have described resource acquisition as the main driver of primate terrestriality, particularly water, during dry periods (Eppley *et al.*, 2022., Estrada & Marshall 2024). Robust capuchins (*Sapajus spp*) in the Urucum Massif, Brazilian Pantanal increased their terrestrial activity during dry seasons compared to wet seasons, and are generally more terrestrial than capuchins in rainforest environments (Wright *et al.*, 2019). Future studies at Issa should focus on whether drinking behavior is a key driver of terrestrial behavior, and whether variation in

canopy height influence mushroom availability throughout the season. The Issa Valley provides a valuable study case where guenons persist in savanna-mosaic habitats, highlighting the ecological flexibility, as seen in Issa red-tailed monkeys. The red-tailed monkey's adjustment of terrestrial behavior based on activity, vegetation, and season reflects strategies to optimize resource acquisition while managing predation risk

Limitations and future work

There are likely multiple drivers of Issa red-tailed monkey's terrestrial behaviour, from foraging and drinking to predator avoidance. Subsequent studies should focus on these potential drivers and attempt to differentiate their impact depending on where monkeys spend their time. There may also be individual-level variation in ground use with body size, sex, age, or personality (Tabacow *et al.*, 2009). For example, vulnerable age-sex classes (e.g., juveniles, females with infants) may be less inclined to leave the safety of arboreality in the face of increased risk on the ground, as observed in wild rhesus monkeys (*Macaca mulatta*) in India (Anand and Radhakrishna, 2022). However, Issa specific characteristic might pose challenges for studying terrestriality, but exploring these limitations could lead to a deeper understanding of primate behavior

Implications for hominid evolution

Studying guenon terrestriality offers important insights for reconstructing the ecological and behavioural pathways that shaped hominid evolution. Guenons exhibit varying degrees of terrestriality and arboreality and thus provide a valuable comparative framework for understanding how shifts in habitat use influence social systems, locomotion, and morphology. By examining species that have independently adopted more ground-based lifestyles such as *C. aethiops* (Isbell *et al.*, 2002) and *C. campbelli* (Arnaud *et al.*, 2013), we can explore the ecological pressures, such as predation risk, resource distribution, and

interspecific competition, that drive transitions from forest canopy to terrestrial niches.

These behavioural and ecological adaptations can illuminate the selective contexts in which early hominins evolved increased terrestriality. For instance, future work can address how ground use influences group cohesion, mating systems, vigilance behaviour, and dietary flexibility—all factors thought to have been critical in the evolution of early hominids (*Australopithecus* and *Homo*), Foley (1989) and Isbell (2002). Moreover, and investigating the anatomical correlates of terrestriality in guenons, such as limb proportions and joint mobility. Delson (2000) and Gebo (1993), provides a comparative model for interpreting fossil postcranial remains. Thus, guenon terrestriality offers a living laboratory for testing hypotheses about the adaptive significance and evolutionary consequences of terrestrial behaviour in our own lineage (Hoare *et al.*, 2019).

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PRELIMINARY FAUNAL BIODIVERSITY ASSESSMENT OF COMMUNITY-MANAGED COASTAL FORESTS IN MKINGA DISTRICT, NORTH-EASTERN TANZANIA

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ABSTRACT

A survey of selected Village Land Forest Reserves (VLFRs) within the Mkinga District in the Tanga region was undertaken in October 2024. The VLFRs studied included Dima, Mazola Kilifi (hereafter referred to as Mazola), and Mbuta forest reserves. The study aimed to assess the biodiversity (fauna) of these forest reserves. A combination of methods was employed to ensure each taxon is adequately surveyed. In this study, smaller vertebrates (small mammals, amphibians, reptiles, and birds) were sampled using standard trapping and fixed-radius point count methods. Larger mammals were studied using the standard line transect method (transect walks), camera traps, and opportunistic observations. Indirect methods of detection (animal signs and informal discussions with residents) were also used to assess the presence of fauna. A total of 4 amphibian species and 21 reptile species were recorded during the survey, including two species of conservation concern: Central African rock python (*Python sebae*), which is Near Threatened (NT) and the Usambara vine snake (*Thelotornis usambaricus*) that has a 'Vulnerable (VU)' IUCN status. Seventy-four (74) species of birds were detected in the forests surveyed. The number of bird species detected in each forest was 32, 48, and 31 for Dima, Mazola, and Mbuta, respectively. Of these, IUCN Redlist species include 2 Endangered (EN) species, namely Bateleur Eagle (*Terathopius ecaudatus*) and Martial Eagle (*Polemaetus bellicosus*) and 2 Vulnerable (VU) species, Tawny Eagle (*Aquila rapax*) and Southern Ground Hornbill (*Bucorvus leadbeateri*). Moreover, four species of migratory birds were recorded, among which two species were palearctic, and two were intra-African migrants. Twenty-seven (27) medium to large-sized mammalian species were recorded, five of which are of conservation concern. The study also noted the presence of human activities in the forest reserves, including logging, charcoal production, and livestock grazing. This baseline study highlights the importance of Community-based Forest Management in the conservation of coastal forest species, as they harbour many species, including those of conservation concern, although experiencing disturbances that threaten their existence. It is suggested that further surveys be conducted in different seasons to provide more information on the forests, including population trends.

Key words: Biodiversity, Coastal forests, species of conservation concern

INTRODUCTION

The terrestrial coastal forests of Tanzania are part of the Coastal Forests of Eastern Africa biodiversity hotspot, one of the highest priority areas for conservation (Mittermeier *et al.*, 2004, Burgess *et al.*, 2017). The coastal

forests of Tanzania support many endemic and near-endemic species of fauna and flora (Burgess *et al.*, 1993). Many of the coastal forest endemic or near-endemic vertebrates are under threat. For instance, as of 2017, 12 of the 27 bird species were categorised as Near Threatened or with a higher status of threat, as

are 13 of 20 mammals, four of six amphibians and eight of 16 reptiles, two mammal species, the Aders' duiker *Cephalophus adersi* and the Rondo galago *Galagoides rondoensis*, are categorised as Critically Endangered (Burgess *et al.*, 2017). Despite their ecological significance, these forests are experiencing disturbances that threaten their existence (Mugisha *et al.*, 2016). The main threat being losses of habitat in the coastal forest as result of anthropogenic activities fuelled by rapid urbanisation, industrialisation and population growth (Burgess *et al.*, 2017). Coastal forests in Tanzania were more extensive in the past, but due to uncontrolled anthropogenic activities only small patches ranging in size from 1 to 50 ha remain (Burgess *et al.*, 2000) that serve as refuge for the forest specialist, endemic and threatened species.

The East Usambara Mountains of north-east Tanzania are one of the 13 mountain blocks that comprise the Eastern Arc Mountains (Leonard *et al.*, 2010). The two main districts of Tanzania comprising the East Usambara landscape are Muheza and Mkinga. WWF Tanzania is implementing a five-year project (2022-2026) known as the East Usambara Restoration Project (EURP) that aims to restore and enhance sustainable conservation of the forests, wetlands, and associated critical species in the Usambara Sub-montane, lowland, and coastal forests for wildlife protection, increased ecosystem services, and improved local livelihoods. The lowland coastal forests of

East Usambara Landscape in Mkinga district, including areas adjacent to Mkomazi National Park and around Uмба River, are important habitats for various wildlife species, including elephants (*Loxodonta africana*), Angola black-and-white colobus monkeys (*Colobus angolensis*), buffaloes, and dik-dik. However, the habitats, dispersal areas, and migratory routes are highly disturbed by human activities, including crop cultivation, charcoal making, and livestock grazing. This underscores the need for habitat and wildlife inventories to inform sustainable conservation efforts. The Mkinga District Council has established Village Land Forest Reserves (VLFRs), facilitated by WWF. However, little was clearly known about what has been achieved in terms of biodiversity in these VLFRs prior to the present study. A fauna biodiversity assessment was conducted in three selected VLFRs namely Dima, Mazola and Mbuta. Biodiversity assessments are crucial in effectively managing and conserving protected areas, including VLFRs, as they provide a baseline understanding of the present species, distribution, and ecological relationships, monitor changes in biodiversity over time, and identify trends and potential threats. The general objective of this study was to assess the fauna diversity of VLFRs. Specifically, the survey has provided detailed baseline information: species richness, composition, abundance and diversity of fauna present covering various taxa including amphibians, reptiles, birds and mammals.

MATERIAL AND METHODOLOGY

Study Area and Study Sites

The study was conducted in three villages within Mkinga District in the Tanga Region: The Dima, Mazola Kilifi (hereafter referred to as Mazola), and Mbuta villages (**Fig 1**). Field surveys were conducted by University of Dar es Salaam (UDSM) biodiversity experts and WWF Tanzania staff in October 2024 in the

respective VFRs of Dima, Mazola and Mbuta. The Mkinga district is approximately between latitude 7°41'00" and 9°25'00"S and longitude 33°40'00" and 35°40'00"E. The district has a warm, humid tropical climate with two rainy seasons (March–May and October–December), and an annual rainfall average of around 750 mm. The study areas are within the eastern Usambara Landscape, comprising 1785.25 ha (Dima), 2850.5 ha (Mazola), and 2247.4 ha (Mbuta).

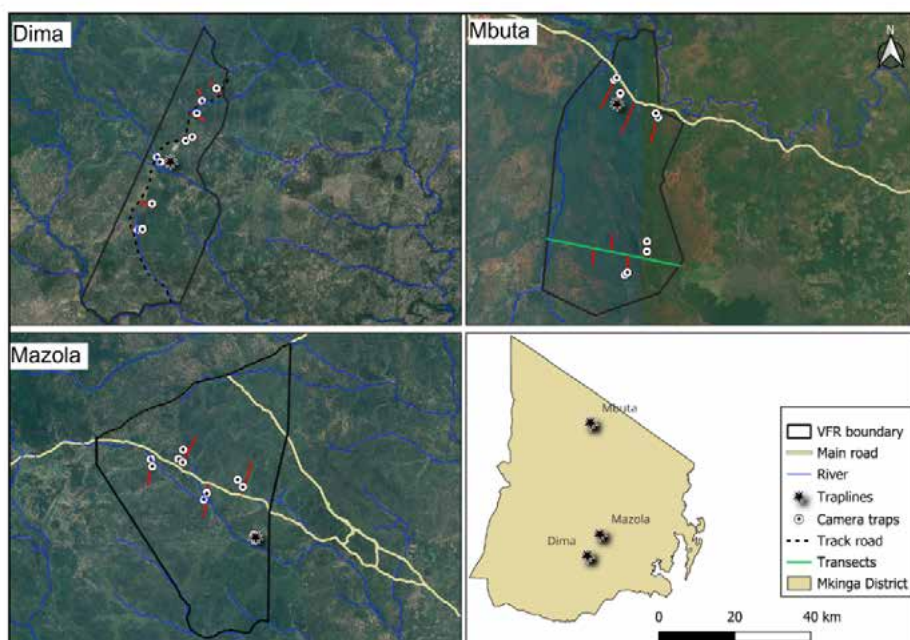


Figure 5: Map of study sites showing the transects walked, camera trap points, and trap lines

Study design, data collection and data analysis methods

Herptiles (amphibians and reptiles) and small mammals

Due to the variety of body sizes, habitats occupied, foraging modes and activity patterns displayed by the small terrestrial vertebrates (small mammals, reptiles and amphibians), there is no general method that can be used to sample all species (McDiarmid *et al.* 2012). Standard techniques were used to assess the presence of small mammals, amphibians, and reptiles and obtain an index of abundance. Drift

fence array with bucket pitfall traps, Sherman and Tomahawk live traps were used to record the presence of small terrestrial vertebrates (excluding bats and birds) in the study VFRs. These methods have been shown to be effective elsewhere in East Africa (Howell 2002, Magige and Senzota 2006, McCleery *et al.*, 2022). Reptiles and amphibians, on the other hand, were also opportunistically recorded when they were found outside of the systematic (regular) timed surveys of either mammal or bird counts. Time-constrained searches were planned to be done during the night hours; however, that was not possible due to WWF policy that doesn't allow their vehicles to operate at night.

In each VLFR, eleven 20-litre buckets spaced 5m were buried so that their rims were level with the ground. Small holes were drilled in their base to allow water to drain from them in case it rains. A drift fence made of black plastic sheeting approximately 0.5 m high supported by wooden pegs in a straight-line array was stapled to guide animals into the buckets. The traps stayed for four nights in each site and were checked for captures in the morning hours. In addition, 22 Sherman live traps (11 in each side of the bucket trapline) spaced 5m and 10m from the bucket trapline were set. Two Tomahawk cage traps were placed purposively, one on either sides of the bucket pitfall trapline. The traps were baited with peanut butter and sardines (Magige and Senzota 2006, McCleery *et al.*, 2022). Checking of traps and rebaiting was done in the morning. Old bait was removed and buried away from the trap line. The trapped animals were identified to species level whenever possible using field guide books (Harper *et al.*, 2010, Kingdon 2015, Spawls *et al.*, 2018, Stevenson and Fanshawe 2020, Menegon *et al.*, 2020) and identification keys. The animals captured were marked by a permanent marker pen, photographed and released at the place of original capture.

We also utilised indirect observation methods, including identifying animal signs and engaging in informal discussions (FGD) with local residents. Animal signs such as footprints and dung provide valuable evidence of species presence and activity without direct sightings (McCleery *et al.*, 2022). Residents often observe wildlife in daily activities, we thus engaged with the local residents from each village and asked them on the species present and we used guide books to confirm the species that they described in their local language.

Data analysis

The gathered data was entered into Microsoft Excel. The data were analysed using PAST 4 software. Species diversity was computed using the Shannon-Wiener index and a diversity t-test was used to test significant variation in species

diversity between samples. The Kruskal-Wallis test was used to test significant variation in species abundance between samples (VLFRs and habitat types) after a normality check with the Shapiro-Wilk test. Statistical significance was set at $\alpha \leq 0.05$ for all the tests performed. Trap success per each trap used was calculated by the formula (Kiwia, 2009)

Trap success =

$$\text{Trap success} = \frac{\text{Number of individuals captured}}{\text{Trapping effort}} \quad \text{Where:}$$

$$\text{Trapping effort} = \text{Number of traps} \times \text{Number of trapping nights}$$

Birds

Birds were sampled using the fixed-radius point count method. To counteract with edge effect, a line transects of 1km were established 200 meters from the forest edges. Birds were recorded in a total of 70 points that were marked using the handheld GPS. In each point, all birds observed with naked eyes, binoculars or heard within a 25-meter radius in 10 minutes (Hostetler and Main, 2016) were identified and recorded using the field guide to Birds of East Africa book (Stevenson and Fanshawe, 2020). We conducted the birds survey during the morning hours (0630-0930hours) and late evening hours (1600 – 1800 hours) so as to align with the peak period of bird activity, ensuring optimal detection rates. Birds were also recorded opportunistically.

Data analysis

Data obtained from the field were carefully organised, cleaned and descriptive analysis was carried out using Microsoft Excel. Inferential statistics were achieved through the use of statistical software called PAleontological STatistic “PAST”. Shannon (H) index was used successful to compute species diversity of surveyed VLFRs, Bray Curtis index was used to compute species similarity of the selected VLFRs, diversity t-test was applied in comparing the diversity among VLFRs. Normality of data were tested using the Shapiro Wilk test while abundances were tested for

significant variation using the Kruskal-Wallis test. Presence of forest-dependent bird species was also assessed. Bird species were classified as Forest Specialists (FF species) which are the birds of the forest interior and are likely to disappear when the forest is modified, Forest Generalists (F species) occurs in undisturbed forest but are able to exist in modified and fragment forests and at the forest edges. In addition to these group, there is Forest visitors (f species) which are the birds recorded in forest but are not dependent on it (Werema, 2021). Conservation status or threat categories were checked in IUCN Red List online, a facility which provides current information on threat categories. We assessed the Globally Threatened species (Critically Endangered (CR), Endangered (EN), Vulnerable (VU) and Least Concern (LC) for all species. IUCN publishes the IUCN Red List of Threatened Species (updated yearly, compiling information from a network of conservation organisations to rate which species are most endangered. The IUCN Red list is the world's most comprehensive inventory of the global conservation status of biological species

Medium and large-sized mammals

For the purposes of this study, large and medium-sized mammals were defined as any mammal whose known average weight, as given by Stuart & Stuart (2006), exceeds 5 kg (Njoroge *et al.*, 2009). The survey of medium-sized to large mammals was conducted primarily using the standard line transect distance sampling method (Norton-Griffiths, 1978; Kiwia, 2006; Buckland *et al.*, 2015). This involved walking along pre-established transects and recording species of medium-sized and large mammal groups (1 or more individuals) seen, the number of individuals, GPS location, and time. The perpendicular distance of the group's centre from the transect line was estimated by eye. The transect walks/ counts was done in the morning and/or evening and covered different areas of the VLFRs to ensure adequate area coverage. Transects were located at least 500m apart to avoid double

counting. In addition, camera traps (CTs), animal signs such as dungs and tracks and the reports of residents was used to establish species lists of the fauna in the selected village forest reserves. The presence of human activities as an indication of disturbance in the forest was also opportunistically recorded. In each VLFR, 10 Camera traps which were approximately 30cm above the ground, were set to take photos 24- hours per for at least 5 nights. At least 2 CTs were deployed in each transect at intervals of 200m purposively at sites which were suitable floors for wild animals" trails and other signs. Data such as date, name of forest reserve, time, camera ID, GPS coordinate of the camera location, and habitat type were recorded. CTs provided additional information on the mammal species checklist.

Data analysis

Species richness was compiled from direct encounters, animal signs (footprints and dungs), camera traps and information gathered from the local residents. For line transect data; Density of mammals was computed using the formula;

$$Density = \frac{\text{Total animals counted (n)}}{\text{Total Area Searched (A)}}$$

(Norton-Griffiths, 1978; Buckland *et al.*, 2015)

where n = total number of individuals of each species, A = Total transect lengths (L) x Effective trip width (2y). Effective trip width = 2y (2 x mean sighting distance(y))

$$\text{Mean sighting distance (y) per species} = \frac{\text{Total sighting distances}}{\text{Total sightings}}$$

Since most animals in the study sites were very shy (Authors observation), very few could be sighted directly and counted during the transects walks, hence we mainly relied on other methods such as Camera traps and animals' signs including dungs and footprints.

For CT data, % frequency of each species was computed as % of CTs with image of certain species. The frequency of CTs images were compared among the VLFRs using Chi-square test. To test if there is an association between

the frequency of trapped mammals in CTs and VLFRs and habitat chi-square test was employed. Frequency data was presented in percentage

For animal signs, we counted the number of dung piles encountered per kilometre of transect walked. The checklist of all medium-sized to large mammals was compiled from data obtained using the combination of direct transect counts, CTs, animal signs and interviews with local residents.

RESULTS AND DISCUSSION

Herptiles and small mammals

From live trapping, opportunistic encounters and reports from local residents, we recorded a total of 4 amphibian species (Plate 1) from 4 families and 21 reptiles (Plate 2) from 13 families for all three VLFRs (Table 1). Out of all reptile species recorded, 11 species were recorded from opportunistic encounters, 6 from reports during FGD with village residents, 3 from bucket pitfall traps and the Great-plated lizard (*Broadleysaurus major*) was captured in Tomahawk and Sherman traps. For amphibians, two species were captured in bucket pitfall traps and the other two species were recorded opportunistically. Small occurrence of amphibian species is due to that most amphibians are active during the night and night searches were not conducted but also the assessment was conducted in the dry period and most water bodies within the VLFRs were completely dry. In addition to, being unable to

conduct night searches led to missing nocturnal reptiles such as Chameleons and snakes that are active and easily seen during the night hours.

The difficulty in documenting many amphibians and reptiles without conducting night audio-visual searches is attributed by the fact that many amphibians and reptiles are primarily nocturnal and they emerge after sunset to hunt or mate, making them visible during the day (Spence-Bailey *et al.*, 2010; Hamer and Parris, 2011). Furthermore, it becomes cooler and more humid during night hours and this favours amphibians who hide during the day to avoid desiccation (von May *et al.*, 2019). Lastly, the cryptic colouration of many amphibians and reptiles makes them difficult to spot in day hours but they can easily be spotted at night with aid of headlamps (McDiarmid *et al.* 2012). Most amphibian and reptile species recorded in this study fall in the 'Least Concern (LC)' IUCN Red List category with the exceptions of Central African rock python that is Near Threatened (NT), the Usambara vine snake that is Vulnerable (VU) and Belliana's hinged tortoise that is Data Deficient (DD) (Table 1). Tanzania's lowland coastal forests consist more than 50 amphibian species and most of them are widely distributed in Tanzania or Sub-Saharan Africa which leads to most of them being categorised as Least Concern (Barratt, 2017). Most of these widespread coastal amphibians are poorly understood and may contain poorly defined species boundaries that may represent more undescribed cryptic species (Barratt, 2017; Zimkus *et al.*, 2017).



Plate 1: Some amphibian species recorded in the survey



Plate 2: Representative reptile species recorded in the survey

Table 1: Checklist of reptiles, amphibians and small mammals recorded in the surveyed VLFRs

S/N	Species name	Common name	FAMILY	IUCN status	Method
Reptiles					
1	<i>Kinixys beliana</i>	Bell's hinged tortoise	Testudinidae	DD	Opportunistic
2	<i>Chamaeleo dilepis</i>	Flap necked chameleon	Chamaeleonidae	LC	Opportunistic
3	<i>Varanus albigularis</i>	White-throated monitor	Varanidae	LC	Opportunistic
4	<i>Latastia longicaudata</i>	Southern long-tailed lizard	Lacertidae	LC	Opportunistic
5	<i>Heliobolus spekii</i>	Speke's sand lizard	Lacertidae	LC	Opportunistic
6	<i>Broadleysaurus major</i>	Great plated lizard	Gerrhosauridae	LC	Sherman and Tomahawk
7	<i>Trachylepis planifrons</i>	Tree skink	Scincidae	LC	Opportunistic
8	<i>Trachylepis maculilabris</i>	Speckle-lipped skink	Scincidae	LC	Opportunistic
9	<i>Hemidactylus platycephalus</i>	Tree gecko	Gekkonidae	LC	Bucket pitfall
10	<i>Hemidactylus squamulatus</i>	Nyika gecko	Gekkonidae	LC	Bucket pitfall
11	<i>Lygodactylus picturatus</i>	Yellow-headed dwarf gecko	Gekkonidae	LC	Bucket pitfall
12	<i>Dendroaspis polylepis</i>	Black mamba	Elapidae	LC	Opportunistic
13	<i>Dendroaspis angusticeps</i>	Green mamba	Elapidae	LC	Interview
14	<i>Boaedon fuliginosus</i>	Brown house snake	Lamprophiidae	LC	Opportunistic
15	<i>Psammophis mossambicus</i>	Olive sand snake	Psammophiidae	LC	Interview
16	<i>Naja melanoleuca</i>	Forest cobra	Elapidae	LC	Opportunistic
17	<i>Thelotornis usambaricus</i>	Usambara vine snake	Colubridae	VU	Opportunistic
18	<i>Causus defilippii</i>	Snouted night adder	Viperidae	LC	Interview
19	<i>Bitis arietans</i>	Puff adder	Viperidae	LC	Interview
20	<i>Python natalensis</i>	Southern African Rock Python	Pythonidae	LC	Interview
21	<i>Python sebae</i>	Central African Rock Python	Pythonidae	NT	Interview
Amphibians					
1	<i>Phrynobatrachus acridoides</i>	East African puddle frog	Phrynobatrachidae	LC	Opportunistic
2	<i>Ptychadena mascareniensis</i>	Mascarene grass frog	Ptychadenidae	LC	Opportunistic
3	<i>Sclerophrys gutturalis</i>	Guttural toad	Bufonidae	LC	Bucket pitfall
4	<i>Kassina senegalensis</i>	Senegal running frog	Hyperoliidae	LC	Bucket pitfall

Twenty-five (25) small mammals from 7 species were captured in Dima and Mazola while Mbuta did not capture any small mammal (Plate 3). Mazola was more diverse ($H'=1.4328$) than Dima ($H'=1.17$) although the difference was not significant as shown by diversity t test ($t=0.93$, $df=21.9$, $p=0.36$). Two unidentified bat species were recorded opportunistically in Mazola and Mbuta VLFRs that suggest the use of mis-nets in the future surveys that could help identify these bat species.

Overall, for a total of 12 trap nights, were able to capture 33 individuals from 13 species in Dima, Mazola and Mbuta VLFRs. Among all captured animals 7 were small mammals followed by 4 reptiles and 2 amphibians (Figure 2). Of the three VLFRs, Mazola had the highest combined captured species diversity ($H'=1.714$), followed by Dima ($H'=1.714$) and Mbuta ($H'=0.6931$). The trend was the same for captured species abundance where Mazola had the highest abundance of 16 individuals followed by Dima with 15 individuals and

Mbuta with only 2 individuals. Despite the difference in species abundance between the three VLFRs, Kruskal-Wallis test showed that the difference was insignificant ($H=1.248$, $p=0.4662$). Overall, the most captured species

was *Acomys sp* (11), followed by Ochre bush squirrel (6), *Gerbiliscus sp* (3), Tree gecko (2), *Crocidura sp* (2) and Great-plated lizard (2) while the rest were captured once.



Plate 3: Some species of small mammals captured in traps

Species accumulation curves (Figure 2) varied between Dima, Mazola and Mbuta VLFRs for the four trap nights that trap stayed. Dima species accumulation curve flattens at the end, indicating that the number of species will not increase even if more trapping effort (nights) were added. For Mazola the species

accumulation curve does not appear to approach the asymptote level, indicating that if there was more trapping effort, we could get additional species. Likewise, for Mbuta the species accumulation curve does not level off indicating there could be additional species if trapping effort was increased.

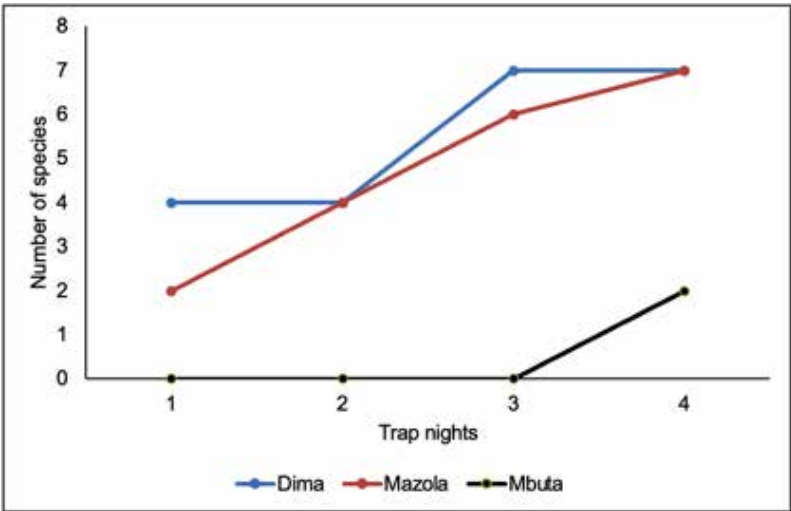


Figure 6: Species accumulation curve for the traps used in Dima, Mazola and Mbuta VLFRs

Birds Species richness

Overall, we recorded a total of 74 bird species (32 bird species in Dima, 48 bird species in Mazola and 31 bird species in Mbuta) from 37 families, in which 60 bird species were recorded from systematic surveys at 70 points, while 14 bird species were recorded through opportunistic sightings (Plate 4). The most common bird species was Slate-coloured Boubou (*Laniarius funebris*) counted in 57 points and the least common was Sombre Greenbul, counted in 16 points.

Birds Species Abundance

A total of 790 individual birds were recorded from systematic point surveys across the three

Village Land Forest Reserves (VLFRs), with 172 individuals in Dima, 407 in Mazola, and 271 in Mbuta. The most abundant species was the Slate-coloured Boubou *Laniarius funebris* (11.39%), followed by the Northern Brownbul *Phyllastrephus strepitans* (8.61%) and the Emerald-spotted Wood Dove *Turtur chalcospilos* (6.2%). Among the 37 recorded families, three Accipitridae (hawks and eagles), Malaconotidae, and Nectariniidae each contained at least five species. Kruskal-Wallis's test revealed no significant differences in bird abundance among the surveyed VLFRs ($p = 0.4307$).



Plate 4: Some of the bird species recorded during the study in the selected VLFRs

Birds Species composition

The VLFRs had 18 bird species in common, eight, 16 and three bird species only recorded at Dima, Mazola and Mbuta, respectively. Nevertheless, Mbuta and Dima had the fewest bird species in common as compared to Mazola and Mbuta. Mazola VLFR recorded the highest bird species over the other two studied VLFRs. This record is attributed by diverse nature of vegetation type in Mazola VLFRs which provided a conducive environment for birds of variety species to inhabit.

Birds Diversity

The most diverse village forest reserve (VLFR) was Mazola, followed by Dima VLFR, while the least diverse VLFR among the three surveyed was Mbuta, based on the Shannon diversity index (H') values of 3.428, 3.133, and 3.028, respectively. Using diversity t-test, the results revealed significant differences in bird diversity between Mazola and Dima forests, as well as between Mazola and Mbuta ($t = -3.9446$, $p = 9.65E-05$ and $t = 5.263$, $p = 2.28E-07$). In contrast, the bird diversity of Dima and Mbuta VLFRs was found to be not significantly different ($t = 1.2027$, $p = 0.22983$). In the context of coastal forest ecosystems, variation in bird diversity across different forest reserves is often influenced by a combination of habitat structure, floristic composition, and degree of anthropogenic disturbance (Burgess *et al.*, 2007). In the case of Mazola VLFR, the observed high diversity value, as compared to VLFRs, can be attributed to a unique set of ecological and environmental factors that make Mazola an ecologically favourable habitat for a range of bird species. Studies conducted in tropical and coastal forests shows areas with complex vegetation structure and high heterogeneity generally support higher avian diversity (Kissling *et al.*, 2012). Mazola VLFR, due to its relative habitat complexity characterised by a mixture of young and mature trees as well as minimal fragmentation compared to other forest like Dima provides various ecological niches for both resident and migratory bird species hence higher bird diversity.

Birds Migration Status

Four migratory bird species were identified within the three surveyed Village Land Forest Reserves (VLFRs) of Dima, Mazola, and Mbuta, representing both Palearctic and Intra-African migration systems. The Palearctic migrants, *Merops apiaster* (European Bee-eater) and *Plegadis falcinellus* (Glossy Ibis), undertake extensive seasonal movements between breeding grounds in the Northern Hemisphere and wintering areas in sub-Saharan Africa. In contrast, the Intra-African migrants, *Cossypha natalensis* (Red-capped Robin-chat) and *Eurystomus glaucurus* (Broad-billed Roller), exhibit regional movements within the African continent, likely driven by rainfall patterns and food availability. This pattern highlights the ecological significance of the VLFRs as important stopover and foraging habitats for both intercontinental and intra-continental migratory bird species.

The presence of migratory birds in the Village Land Forest Reserves (VLFRs) of Mkinga underscores the ecological significance of coastal forests, which serve as critical habitats for both resident and migratory species. Coastal forests are essential for migratory connectivity, providing vital resources such as food, water, and shelter that sustain birds throughout various phases of their life cycles. For instance, the European Bee-eater (*Merops apiaster*), a Palearctic migrant, relies on these forests during the northern winter months when resources are scarce in its breeding grounds. Mkinga's VLFRs offer abundant insect prey for this species, allowing it to refuel and complete its migratory journey. Intra-African migrants, such as the Broad-billed Roller *Eurystomus glaucurus*, also rely on the Mkinga VLFRs for seasonal resources. These reserves, with their structurally diverse forest patches, support high insect populations that are vital for insectivorous species like the roller, particularly during the non-breeding season. The migrant birds benefit from the habitats provided by these forests, which underscore the role of Mkinga's VLFRs in supporting both year-round residents and

seasonal visitors. Migratory birds additionally play significant ecological roles within these ecosystems for instance, species like the European Bee-eater and Broad-billed Roller help regulate insect populations, while seed-dispersing birds such as the oriole contribute to forest regeneration. These interactions enhance ecosystem complexity and resilience, promoting the health and stability of coastal forests.

Birds forest-dependent species

The VLFRs harbour both forest-dependent and forest visitor's species. Four species of birds were the forest generalist species (F species) including Collared Sunbird, Narina Trogon, Black-backed Puffback and Red-capped Robin-chat while 10 bird species were the forest visitors (f-species) including Lizard Buzzard, Red-eyed Dove, Emerald-spotted Wood Dove, Broadbill Roller, Crowned Hornbill, African Golden Oriole, Common Bulbul, Green-backed Camaroptera, Rattling Cisticola and White-browed Robin-chat. The composition of forest-dependent bird species of selected Village Land Forest Reserves (VLFRs), featuring forest generalists (F species) and some forest visitors (f species) but lacking forest specialists' species (FF), suggests these forests are of moderate ecological maturity with limited habitat specialisation. The absence of forest specialists, which typically rely on dense, mature forest conditions, indicates that these VLFRs may lack the structural complexity associated with primary forests, possibly due to disturbance or insufficient forest continuity. The presence of forest generalists and visitors shows these reserves still provide valuable resources, such as food and shelter, that support a range of bird species from neighbouring habitats. This bird community structure highlights a need for targeted conservation and restoration efforts, like reforestation and connectivity enhancement, to foster a more complex forest ecosystem that could eventually support a broader array of species, including forest specialists, thus enhancing biodiversity in the VLFRs.

IUCN status

Based on the IUCN red list, the majority of the species recorded fell under the category of Least Concern with the exceptional of two species namely Bateleur *Terathopius ecaudatus* and Martial Eagle *Polemaetus bellicosus* which have been listed as Endangered (EN) bird species by IUCN. These species are listed as Endangered as a result of their rapid population decrease in past four decades (40 years) major threats of habitat loss, this is due to the fact that many of large trees that were previously used for nesting are being cut for charcoal, settlement and converted to agricultural fields. (Birdlife International 2024). On other hand, two species namely Tawny Eagle *Aquila rapax* and Southern Ground Hornbill *Bucorvus leadbeateri* are listed as Vulnerable (VU) bird species by IUCN. The Southern Ground Hornbill is listed by IUCN as Vulnerable to extinction due to its rapid population decline especially outside protected areas. Threats include habitat loss and fragmentation along with their low reproductive rate increases the risk of further declines and makes recovery more difficult (Birdlife International 2024).

Medium and large sized mammals

Overall, a total of 28 species of medium-sized-to-large mammals belonging to 13 families was recorded in the study; 20 species, 24 species, 27 species in Dima, Mazola and Mbuta (Table 2) VLFRs respectively. Of these, five species are of conservation concerns; Angola black-and-white colobus (*Colobus angolensis*), African Buffalo (*Syncerus caffer*), African Savanna Elephant (*Loxodonta Africana*), Lesser Kudu (*Tragelaphus imberbis*) and Lions (*Panthera leo*) listed in IUCN red list as 'Vulnerable', 'Near Threatened', 'Endangered', 'Near Threatened' and 'Vulnerable' respectively. Interestingly, Caracal (*Caracal caracal*) was only recorded in Mazola Forest reserve in one of CTs. African civet (*Civettictis civetta*) and Red bush squirrel (*Paraxerus palliates*) were only recorded in Mbuta CTs. As stated previously, most animals were shy presumably due to high

interactions between human, livestock and wild animals in these VLFRs thus only 7 species were encountered and counted during transect walks through direct observations in the three VLFRs. However, the signs of various species were encountered during transect walks (Plate 5). One of the common species recorded in all three VLFRs is Natal red duiker (*Cephalophus natalensis*) which whose dung pellets were regularly seen, images found in CTs and reported by local residents. To the best of our knowledge and current literature, this is the first record of presence of this duiker species in coastal forest of Tanga in Northeast Tanzania. Eight species were observed directly with Banded mongoose (*Mungos mungo*) having the highest density (65 animals per km² and the least being white throated guenon (*Cercopithecus mitis*). The results of direct observations may not be realistic as the majority of animals were not seen during transect counts hence their density (abundance) may be underestimated. For analysis, we relied much on CTs data. As compared to Kilemo *et al* (2023) study on elephants in the area, there was no direct sighting of elephants in this study. However, lots of dungs were observed in all the three forest reserves.

A total of 30 camera traps captured 49 individuals of medium to large-sized mammals belonging to 14 species in all the three VLFRs (Plate 6). Mazola ($H'=2.164$) had the highest species diversity, followed by Mbuta ($H'=1.936$) and Dima ($H'=1.538$). The pattern was the same in abundance where Dima, Mazola and Mbuta had 20, 10 and 19 individuals captured, respectively. Despite this there is no significant variation in diversity and abundance among the three VLFRs (Kruskal-Wallis test, $H=2.645$, $p=0.152$).

The Natal red duiker was the species with the highest camera capture frequency (53.3%), followed by Dik-dik and Bushy-tailed mongoose, both with a 13.3 % frequency. Bushbuck and Bush duiker had a 10% capture frequency, while the rest had a capture frequency less than 10%. The chi-square independence test revealed that there is no association between the frequency of camera-captured species among the three VLFRs ($\chi^2 = 21.02$, $df = 26$, $p = 0.74$). The same test also showed that there is no association between the frequency of camera-captured species and the habitat type in which the camera traps were set ($\chi^2 = 35.74$, $df = 39$, $p = 0.62$).

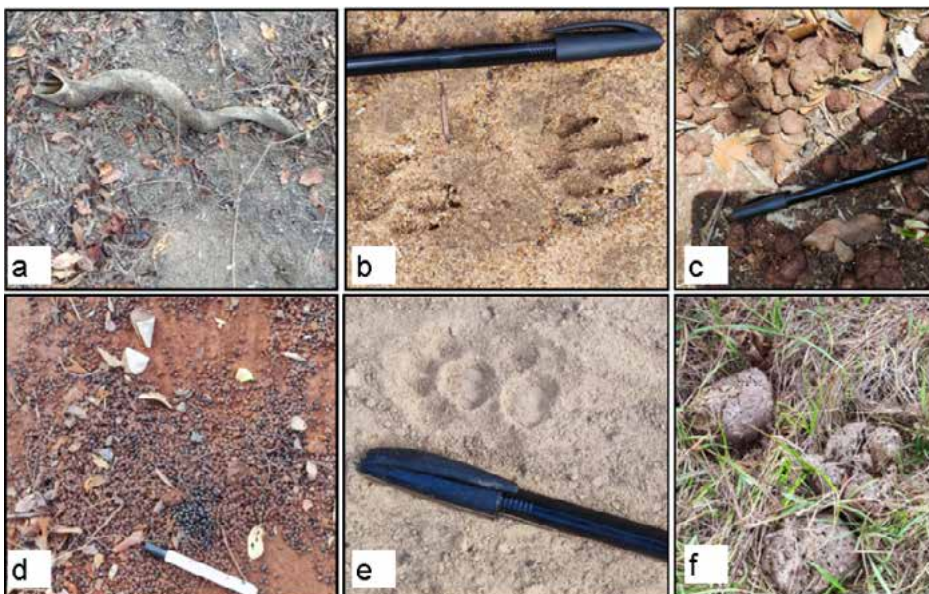


Plate 5: Some of animal signs recorded in the field survey: a) Lesser kudu horn b)Cape hare footprint c)Bushpig dungs d) Dik-dik droppings e) Zorilla footprint f) Elephant dungs.

Table 2: Checklist of medium-sized to large mammal species recorded in the three VLFRs. Note: LC=Least Concern, VU = Vulnerable, NT = Near threatened, EN = Endangered.

S/n	Family	Common name	Scientific name	IUCN Status	Method	VLFR
1	Viverridae	African civet	<i>Civettictis civetta</i>	LC	Camera traps, signs, interview	Dima, Mazola, Mbuta
2	Viverridae	Large-spotted genet	<i>Genetta maculata</i>	LC	Camera traps	Dima, Mazola
3	Suidae	Bushpig	<i>Potamochoerus larvatus</i>	LC	Camera traps, signs, interview	Dima, Mazola, Mbuta
4	Suidae	Warthog	<i>Phacochoerus africanus</i>	LC	Interview	Dima, Mazola, Mbuta
5	Sciuridae	Ochre bush squirrel	<i>Paraxerus ochraceus</i>	LC	Visual, camera traps	Dima, Mazola, Mbuta
6	Sciuridae	Red bush squirrel	<i>Paraxerus palliatus</i>	LC	Camera traps	Mbuta
7	Mustelidae	Honey badger	<i>Mellivora capensis</i>	LC	Interview	Mazola
8	Mustelidae	Zorilla	<i>Ictonyx striatus</i>	LC	Interview	Dima, Mazola, Mbuta
9	Macroscelididae	Four-toed Elephant shrew	<i>Petrodromus tetradactylus</i>	LC	Interview	Dima, Mazola
10	Leporidae	Cape hare	<i>Lepus capensis</i>	LC	Opportunistic sightings, interview	Dima, Mazola, Mbuta
11	Hystriidae	Crested Porcupine	<i>Hystrix cristata</i>	LC	Camera traps, interview	Dima, Mazola
12	Hyaenidae	Spotted hyena	<i>crocota crocuta</i>	LC	Interview	Dima
13	Herpestidae	Banded mongoose	<i>Mungos mungo</i>	LC	Visual, Signs, interview	Mazola, Mbuta
14	Herpestidae	Bushy-tailed mongoose	<i>Bdeogale crassicauda</i>	LC	Camera traps	Dima, Mazola, Mbuta
15	Felidae	Caracal	<i>Caracal caracal</i>	LC	Camera traps	Mazola
16	Felidae	Lions	<i>Panthera leo</i>	VU	Interview	Mbuta
17	Erinaceidae	Four-toed Hedgehog	<i>Atelerix albiventris</i>	LC	Interview	Mbuta
18	Elephantidae	African Savanna Elephant	<i>Loxodonta africana</i>	EN	Signs, interview	Dima, Mazola, Mbuta

S/n	Family	Common name	Scientific name	IUCN Status	Method	VLFR
19	Cercopithecidae	Angola black-and-white colobus	<i>Colobus angolensis</i>	VU	Opportunistic sightings, interview	Dima, Mazola, Mbuta
20	Cercopithecidae	Olive Baboon	<i>Papio anubis</i>	LC	Signs, interview	Mazola, Mbuta
21	Cercopithecidae	Vervet monkey	<i>Chlorocebus pygerythrus</i>	LC	Visual, camera traps, interview, signs, opportunistic sightings	Dima, Mazola, Mbuta
22	Cercopithecidae	White throated guenon	<i>Cercopithecus mitis</i>	LC	Visual, camera traps, interview	Dima, Mazola, Mbuta
23	Bovidae	African Buffalo	<i>Syncerus caffer</i>	NT	Signs, interview	Mazola, Mbuta
24	Bovidae	Bushbuck	<i>Tragelaphus scriptus</i>	LC	Visual, camera traps, signs, interview	Dima, Mazola, Mbuta
25	Bovidae	Common duiker	<i>Sylvicapra grimmia</i>	LC	Camera traps, interview	Mazola, Mbuta
26	Bovidae	Kirk's Dik-dik	<i>Madoqua kirkii</i>	LC	Visual, camera traps, signs, interview	Dima, Mazola, Mbuta
27	Bovidae	Lesser Kudu	<i>Tragelaphus imberbis</i>	NT	Signs, interview	Dima, Mazola, Mbuta
28	Bovidae	Natal red duiker	<i>Cephalophus natalensis</i>	LC	Visual, camera traps, interview, signs	Dima, Mazola, Mbuta



Plate 6: Selected Images of species of mammals retrieved from the camera traps and selected animals' signs observed in the study sites

Our findings in terms of mammal species captured align with Leonard *et al.* (2010), who also reported the widely distributed mammal species African civet, Bushy-tailed mongoose, Large-spotted genet and Bush squirrel in three village landscape forests the East Usambara Mountains using camera traps. Studies show that camera trapping is more efficient in small to large-sized terrestrial mammals and the probability of detecting more species depends on having a large sampling effort (number of cameras multiplied by the number of days) (Rovero 2010). These findings demonstrate the ecological significance of the three VLFRs as they harbour diverse and abundant wild animals, some of which are vulnerable, forest-dependent, migrant bird species. However,

during field observations, we also observed the ongoing anthropogenic activities such as charcoal production, livestock grazing, settlements and road kills threaten the existence of these forest reserves

The study provides the first detailed information of biodiversity in selected VLFR as part Eastern Usambara Coastal forests landscape in Tanzania. This is a baseline information for future studies and monitoring of biodiversity in coastal forests in Tanga, particularly in the Mkinga district. The findings indicate the three VLFRs harbour many species, including species of conservation concern listed in the IUCN red list, thus underscoring the importance of conservation in these forests.

This survey provides the preliminary findings. Further studies on the fauna of these forests and other un-surveyed or partially surveyed forests within the landscape will be conducted.

CONCLUSIONS AND RECOMMENDATIONS

The results clearly indicate that the three forest reserves are rich in biodiversity and some of which are of conservation concerns: avifauna, mammals, reptiles and amphibians. A total of twenty-seven (27) medium and large-sized mammalian species, seventy-four (74) bird species, twenty-one (21) reptilian species and four (4) amphibian species were encountered and recorded in the study area. Using CTs data, Natal red duikers (53%) were the most abundant large mammal species, followed by dik-diks (13%), and African civets, caracals and porcupines were among the least abundant species. For birds, slate-colored boubou (11%, $n=970$), followed by Northern brownbul (8.6%, $n=68$). There was a significant difference in the diversity of birds among the forest reserves; the bird diversity was highest in Mazola ($H' = 3.43$) and Mbuta ($H' = 3.03$) had the lowest bird diversity. A total of seven (7) small mammal species were recorded in all three VLFRs; Mazola ($H' = 1.43$) has the highest species diversity of small mammals, followed by Dima ($H' = 1.17$) and no (Zero) species were captured in Mbuta. ($H' = 0.00$) though the difference was not significant.

This biodiversity survey in the recently demarcated Village Land Forest Reserves identified significant conservation opportunities and challenges in protecting these areas. The findings underscore high conservation value of the community-based forest reserves which are also faced with human pressures impacting habitat quality. We recommend the following actions:

- i. To Protect Potential Elephant Corridors: Evidence of forest use by elephants through the presence of dung was observed in all three reserves, suggesting the area may

serve as a corridor or dispersal zone for the elephants. This underscores the need for intensified conservation to support elephants' natural migration routes and mitigate human-wildlife conflicts that may arise from restricted movement.

- ii. Expand Protection to the Uмба River Habitat: The Uмба River, located just adjacent to Mbuta VLF, provides critical refuge and is the only flowing water source for wildlife in the area particularly during the dry season. Expanding the reserve boundary to include the river or creating a buffer zone with targeted conservation could protect this essential habitat, particularly during water-scarce periods.
- iii. Address Encroachment and Resource Extraction: Despite an established land-use plan, we observed ongoing activities such as charcoal production, pole cutting, livestock grazing, and small-scale mining that degrade forest resources. In Dima Forest, encroachment from settlements and farming was particularly high. Strengthening the enforcement of land-use regulations and supporting alternative livelihoods could reduce these pressures.
- iv. Enhanced Protection for Species of Conservation Concern: The forest reserves host migratory and vulnerable species, which add ecological value to the area. Protecting these species' habitats would help sustain their populations and enhance biodiversity. WWF's support could amplify these conservation outcomes with focused efforts on priority species.
- v. Develop Alternative Livelihoods to Ease Forest Dependency: To reduce the reliance on forest resources, introducing and supporting alternative livelihood options, such as agroforestry, beekeeping, and zero-grazing livestock practices, would lessen the economic pressure to extract forest resources. These alternatives would not only benefit the environment but also improve community well-being.

- vi. Strengthening Conservation Education and Training for Village Game Scouts: Enhancing the skills of local game scouts in biodiversity monitoring through citizen science and conservation data collection would build local capacity for ongoing conservation. Training sessions in species identification, data recording, and reporting would make their patrols and monitoring more effective.
- vii. Increase and Systematise Patrols: Regular, well-coordinated weekly patrols and surprise checks would help prevent illegal activities and improve law enforcement in the reserves. Equipping game scouts with essential resources and tools would empower them to uphold conservation measures consistently.
- viii. Conduct Seasonal Studies to Capture Species Variation: Wildlife distribution, abundance, and diversity are often seasonal, especially in the context of wet and dry periods. A follow-up study during the wet season would provide a more accurate picture of species presence and diversity in these reserves.
- ix. Mitigate Roadkill Risks in Village Land Forest Reserve Areas: Specifically, the road crossing Mazola Village Forest Reserve poses a significant risk for wildlife, leading to roadkill incidents that harm local biodiversity. Implementing wildlife crossings, speed limits, and warning signals on these roads could reduce wildlife mortality and enhance safe passage for species across reserve boundaries.

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MAMMALIAN SPECIES COMPOSITION AND DIVERSITY IN THE AGA KHAN UNIVERSITY'S LANDS, ARUSHA TANZANIA

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ABSTRACT

In Arusha City, the green spaces in peri-urban rangelands are facing escalating pressures from human settlement, agriculture, livestock grazing, and climate change. These pressures threaten biodiversity and compromise ecosystem functions that underpin environmental stability and human well-being. Thus, this study assessed the composition, richness and diversity of mammalian species within the landscapes of Aga Khan University (AKU) in Arusha region to improve their conservation status in the area. The surveyed mammals were categorized by size: large (ungulates and other large herbivores), medium (primates and dik-diks), and small (rodents). Biodiversity surveys were conducted across two AKU sites Manyara (3,200 acres) and Two Rivers (500 acres), located on the outskirts of Arusha City. Thirteen transects representing major habitat types, including mixed forests, grasslands, cultivated lands, hills, and riverine systems, were systematically surveyed, with direct and indirect observations for medium and large mammals, pitfall traps and Sherman traps were used to sample small mammals, amphibians, and reptiles. A total of 11 medium and large mammal species were recorded, with higher occurrence in the relatively less disturbed Manyara site. Small mammals were more abundant and diverse, with 93 individuals representing eight species, and exhibited strong spatial variation across habitat types. Cultivated and disturbed peri-urban areas supported higher rodent abundance, reflecting their role in nutrient cycling, soil aeration, seed predation and dispersal, and as a critical prey base for higher trophic levels. This study demonstrates that peri-urban university lands can function as important ecological spaces that support mammal diversity and sustain essential ecosystem processes despite anthropogenic pressures. The findings provide baseline data for biodiversity monitoring, inform restoration of degraded habitats, and highlight the importance of integrating small and large mammal conservation into urban land-use planning to enhance ecosystem resilience and long-term sustainability.

Keywords: Aga Khan University's lands, Arusha, Species composition and diversity, Small-Medium-Large mammals, Peri-urban rangelands,

INTRODUCTION

The Arusha Climate & Environmental Research Centre (ACER) of the Aga Khan University (AKU) manages approximately 4,000 acres of peri-urban rangeland in Arusha,

northern Tanzania. This landscape forms part of a rapidly transforming rangeland ecosystem that supports diverse mammal species and provides important ecosystem services. ACER works in collaboration with national research institutions, including the Tanzania Wildlife

Research Institute (TAWIRI), to generate evidence that informs biodiversity conservation and sustainable land management. Peri-urban rangelands in northern Tanzania are increasingly threatened by land-use change associated with human settlement growth, livestock expansion, wood extraction and climate variability, contributing to long-term habitat degradation and biodiversity loss (IPBES, 2019; Wiethase *et al.* 2023). These pressures are particularly acute for mammalian communities, which are sensitive to habitat fragmentation and resource depletion.

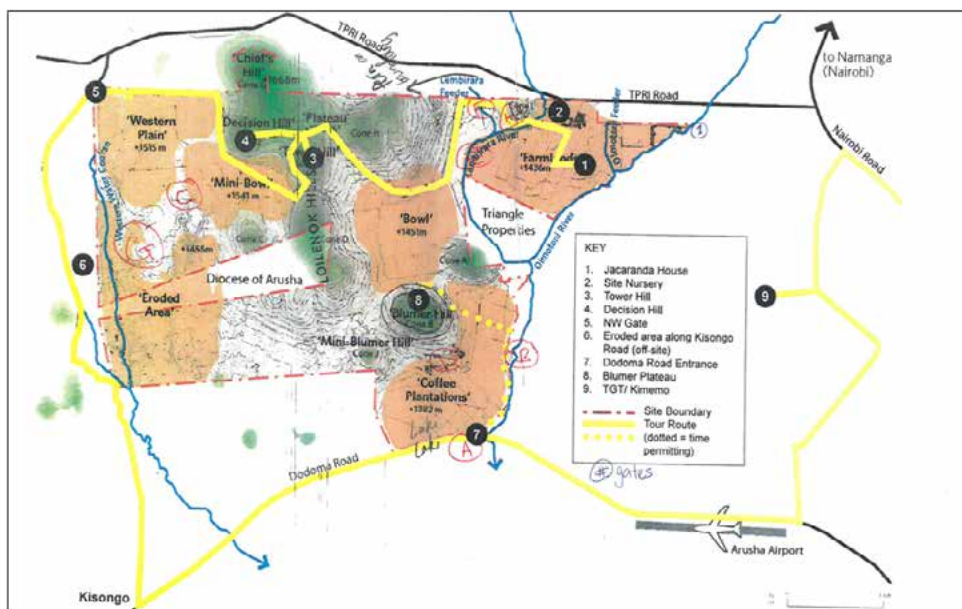
Large mammals are critical for maintaining ecosystem structure and function through grazing, browsing, seed dispersal, and trophic regulation (Sinclair *et al.*, 2015). However, expanding human activities near protected and unprotected lands have reduced habitat availability and increased human–wildlife interactions, leading to population declines and local extirpations in many rangeland systems (IUCN, 2023). Meanwhile, small mammals play an equally vital but often overlooked role in ecosystem functioning. They contribute to soil processes, seed dispersal, and food-web stability and are widely recognized as sensitive indicators of habitat condition and environmental change (Avenant & Cavallini, 2007; Mulungu *et al.*, 2015). Declines in small mammal diversity may signal early stages of ecosystem degradation. Despite their ecological importance, localized data on the status and distribution of both large and small mammals within peri-urban rangelands remain limited, particularly for a university’s living laboratory landscapes such as ACER.

Therefore establishing baseline information on mammalian species composition and diversity is essential for guiding conservation planning, restoration efforts, and their long-term monitoring. This study, therefore, aimed at assessing the composition richness and diversity of mammalian species within the Manyara and Two Rivers sites of the AKU Arusha land. The findings contribute an updated mammal species inventory, identify conservation priorities, and provide baseline data to support sustainable land management, biodiversity monitoring, and evidence-based decision-making on human-animal co-existence.

MATERIALS AND METHODS

Study area

The study was conducted at the AKU’s two sites (Manyara & Two Rivers) in the Arusha region. The main Manyara site is located at the edge of Arusha, 13 km west of Arusha city centre, 3 km north of Arusha airport, and overlooks the pastoral herding communities and smallholder farms to the west and north (Map.1). The site has 3,200 acres of variable land that is ecologically important. The site has seven hills, degraded parts on the western side with gully erosions, and productive land, which is currently used for farming activities. For the last 10 years, AKU has been operating a reforestation program to address overgrazing and deforested areas. Therefore, the site’s peri-urban location and active re-greening initiative provide suitable condition for survey. The second site is the Two Rivers site having about 500 acres, located between two



tributaries at Usa River town located about, 14 km east of Arusha city centre (Map.2). Currently, about 80% of the land is under cultivation, and the remaining 20% consists of scattered trees, shrubland, and open grass areas. The general characteristics of the area include high precipitation, fertile soil, and diverse vegetation. These contrasting environmental conditions combined with their geographic location provide a strong basis and rationale for biodiversity assessment.



Map 7: AKU Arusha Manyara Site - dotted redline is the study area boundary (Source: TAWIRI GIS Office, 2025)

Map 2: Two Rivers Site – left top-down frames are total cultivated and uncultivated areas respectively. Right top-down frames are grassland and shrubland (Source: TAWIRI GIS Office, 2025).

Transect designs

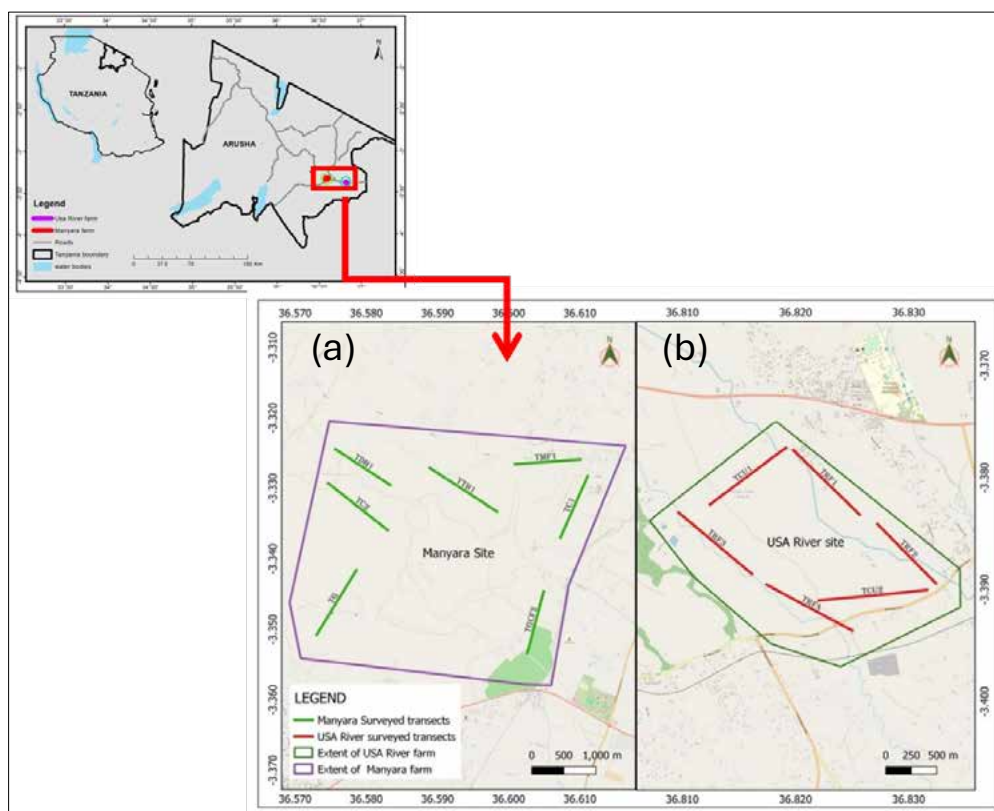
A total of thirteen transects were established, of which seven (7) were in the Manyara site: Mixed forest consisting of Eucalyptus plantation and natural forests (TMF1), Graviileia and Coffee forest (TGCF1), Cultivation site one (CT1), Cultivation site two (TC2), Tower Hill (TTH1), Decision Hill (TDH1), and Grassland (TG1). At Two Rivers site in Usa-river, 6 transects were studied; Riverine Forest 1 (TRF1), Riverine Forest 2 (TRF2), Riverine Forest 3 (TRF3), Riverine Forest 4 (TRF4), Cultivation USA 1 (TCU1), and Cultivation USA 2 (TCU2). In each zone, a transect of 1km was established for surveys of both flora and fauna (Map 3a-b). All transects were performed through walking, with the aim of documenting the fauna species.

Direct observation of their presence and the indirect presence of fecal/pellets, foot signs, and breeding signs were used.

Large mammal surveys

Walking line transects

Two observers walked along the established transects and stopped on every wildlife encounter and/or sighting for identification of the species name, location, and habitat type. Indirect sighting, wildlife signs like faeces, footprints, holes, trails, and feeding signs were observed and identified for each of the species' names, and their locations were recorded using GPS within each transect.



Map 3 (a&b): Study site and Transect layout at Manyara and Two rivers site-Usa-River

Small mammals surveys

i. Pitfall traps with drift fence array

Five transects in Manyara sites (cultivation areas-TC1 & TC12, Decision Hill-TDH1, Mixed Forest-TMF1, and Tower Hill-TTH1) and three transects in Tow rivers sites at USA river were established. Small mammals, amphibians, and reptiles were assessed using pitfall traps with a drift fence array. Seven (7) pits were dug deep enough to insert ten-liter buckets (Photo 1). The bucket traps were buried such that their rims were on the same level as the ground and were set in a “Y” shape (Fig. 1). A drift fence of plastic sheets, approximately 0.5 m high, was created and supported by wooden poles to guide animals encountering the fence into the buckets. Pits were spaced five meters from each other and the traps were checked twice a day, early morning and late evening (Photo 1).

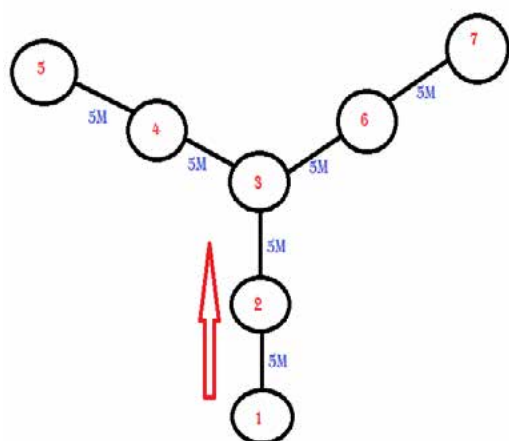


Figure 1: Sketch diagram showing how bucket pitfall traps were arranged in a “Y” shape during the trapping of small mammals, amphibian and reptiles



Photo 1: Bucket pitfall traps with drift fence array

ii. Sherman traps

In each transect established, a total of 16 Sherman traps (Photo 2) were placed in a grid of four lines as presented in Figure 2 with each grid having 4 traps that were spaced at a distance of 30 m from one another. (Fig. 2)



Photo 2: Sherman traps placed on ground

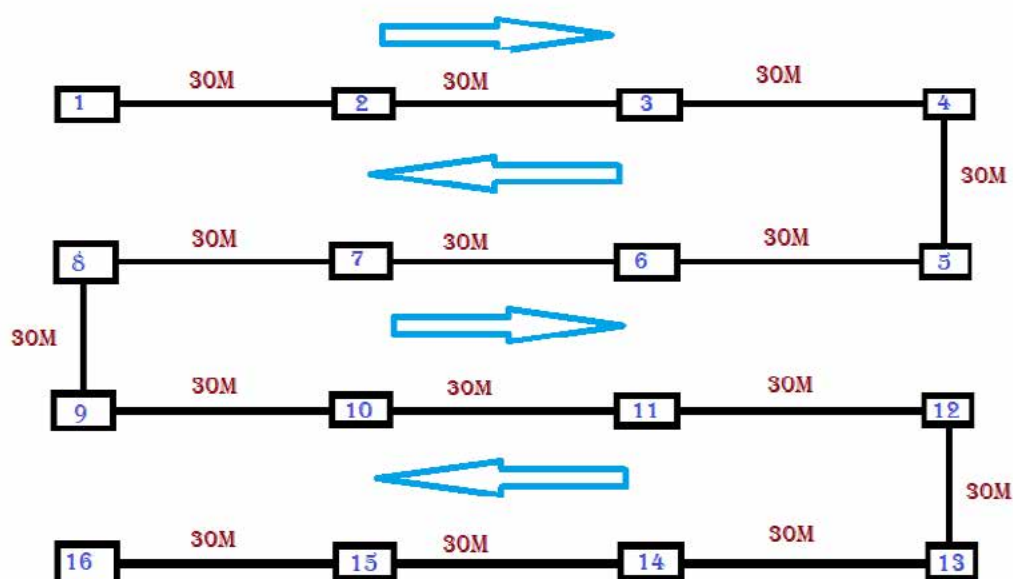


Figure 2: Sketch diagram showing a design how Sherman traps were laid down in a grid of four lines specifying a distance from one trap to another and distance from one grid to another

Data analysis

The collected data for this study was analyzed in Microsoft 365 Excel (the newest, most up-to-date Excel version available). For large and medium sized mammals, calculation on species richness estimates was performed following Seaby & Henderson (2007). This uses three biodiversity indices: richness, diversity and abundance to explain habitat types with the highest biodiversity in the mammal communities. However, considering that the collected data on small mammals was few, no statistical analysis was performed. Instead, only a descriptive analysis was conducted.

RESULTS

Large and medium sized mammals

A total of 11 mammal species belonging to 8 families, 5 orders and 11 genera were recorded

from both direct and indirect methods in both surveyed sites of Manyara and USA (Table 1). The species belonged to five major groups: carnivora, ungulate, primate, rodent and insectivore. The Manyara site supported higher species abundance of large mammals ($n = 7$) while the Two Rivers site accommodated only 4 species (table 1). The Manyara site also had a higher species diversity of $H' = 1.609$, while the two rivers site had a species diversity of $H' = 1.099$. Due to species habitat preferences, species such as blue monkey and black and white colobus monkey were only recorded in the Two Rivers site (Table 1). We consider this to be low number in terms of diversity as compared to the vegetation types in the two areas. All the listed species are classified as Least-Concern (LC) and are not a focus of species conservation because they are still plentiful in the wild.

Table 3. Large mammals observed in the study area

S/N	Order	Family	Common name	Scientific name	No. of Individuals observed	Animal signs	Study site	IUCN status
1	Tubulidentata	Orycteropodidae	Aardvark	<i>Orycteropus afer</i>	0	Holes	BS	LC
2	Lagomorpha	Leporidae	African hare	<i>Lepus victoriae</i>	2		M	LC
3	Carnivora	Herpestidae	Banded Mongoose	<i>Mungos mungo</i>	52		M	LC
4	Primates	Cercopithecidae	Blue Monkey	<i>Cercopithecus mitis</i>	5		2R	LC
5	Artiodactyla	Bovidae	Bush-back	<i>Tragelaphus sylvaticus</i>	2		M	LC
6	Primates	Cercopithecidae	Eastern black-and-white colobus	<i>Colobus guereza</i>	3		2R	LC
7	Artiodactyla	Bovidae	Common duiker	<i>Sylvicapra grimmia</i>	4		BS	LC
8	Carnivora	Viverridae	Common Genet	<i>Genetta genetta</i>		Spoors	M	LC
9	Artiodactyla	Bovidae	Dik-dik	<i>Madoqua kirkii</i>	6		BS	LC
10	Carnivora	Felidae	Serval Cat	<i>Leptailurus serval</i>		Spoors	2R	LC
11	Carnivora	Hyaenidae	Spotted hyena	<i>Crocuta crocuta</i>		Feaces	M	LC

Key: BS=seen in Both sites; M=seen only in Manyara site; 2R=Seen only in Two rivers site

Small mammals

A total of 93 individuals of small mammals (rodents and shrews) belonging to 8 species were collected and identified (*Aethomys chrysophilus*, *Arvicanthis niloticus*, *Crocidura Spp*, *Dasymys incomtus*, *Gerbilis tatera*, *Mastomys natalensis*, *Mus minutoides*, *Xerus rutilus*; Photo 3). Of these, six (6) were collected in the Manyara site and five (5) in the two rivers site-Usa-river. More small mammals were recorded in the transect created in the cultivated area (TCU1) at Usa River with 22 individuals, followed by the cultivated area (TC1) in the Manyara site with 19 individuals, Decision Hill (TDH2) with 16 individuals, the cultivated area (TC2) in Manyara with 12 individuals, the mixed forest (TMFI) with 11, the Tower Hill (TTH1) with 8, two rivers forest in USA River (TRF2) with 3 and two rivers forest in USA River (TRF1) with only 2 individuals. Such a low proportion underscores the unique ecological niche and conservation importance of the AKU for sustaining different species of small mammals.

The finding on the distribution and abundance of small mammals across transects surveyed and their major sites visited revealed significant spatial variations. In the Mixed Forest (TMF1), five species of adult rodents were captured, comprising of *Mastomys natalensis*, *Aethomys chrysophilus*, *Arvicanthis niloticus*, *Crocidula Spp*, and *Dasymys incomtus*. Notably, all captures occurred within the forest areas of the Manyara site. Four species were captured in Tower Hill including *Gerbilis tatera*, *Dasymys incomtus*, *Mastomys natalensis* and *Aethomys chrysophilus* (Table 2). *Mastomys natalensis* was the most frequently captured species in the AKU area with 64 individuals accounting for more than 68.82% of approximately 93 records. The least frequently captured individuals came from the *Arvicanthis niloticus*, *Mus minutoides* and *Xerus rutilus* species with a single individual being captured in each species (Table 2; Photo 3). By contrast, the cultivated areas accommodated higher individual numbers indicative of increased numbers of rodents in these disturbed environments

Table 4. Occurrence frequency of 8 small mammal species in the study area.

Order	Family	Scientific name	Common name		Transect ID									Total	%
				IUCN status	TC1	TC2	TCU2	TDH1	TMF1	TRF1	TRF2	TTH1			
Rodentia	Muridae	<i>Aethomys chrysophilus</i>	Red rock rat	LC	1	1	9		2			3	16	17.2	
Rodentia	Muridae	<i>Arvicanthis niloticus</i>	African grass rat	LC					1				1	1.1	
Eulipotyphla	Soricida	<i>Crocidura Spp</i>	-	LC			1		1				2	2.1	
Rodentia	Muridae	<i>Dasymys incomtus</i>	African marsh rat	LC					4			2	6	6.5	
Rodentia	Muridae	<i>Gerbilis tatera</i>	-	LC								2	2	2.1	
Rodentia	Muridae	<i>Mastomys natalensis</i>	Multimammate rat	LC	18	11	12	16	3		3	1	64	68.8	
Rodentia	Muridae	<i>Mus minutoides</i>	African pygmy mouse	LC						1			1	1.1	
Rodentia	Sciuridae	<i>Xerus rutilus</i>	Unstriped ground squir	LC						1			1	1.1	
Total					19	12	22	16	11	2	3	8	93	100.0	

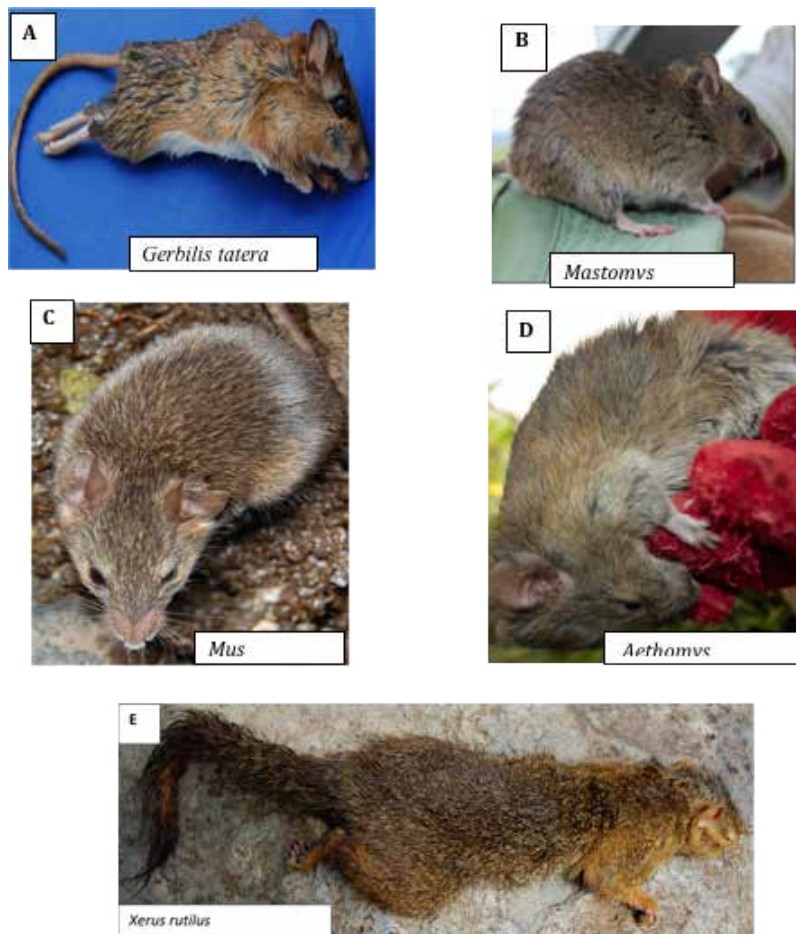


Photo 3: Encountered small mammals species

DISCUSSION

Medium and large mammals form a key component in the study sites. The study recorded relatively few medium and large mammals, likely due to their vulnerability to environmental disturbances, large home range requirements, and potential human–wildlife conflicts (Lwin *et al.*, 2021). Habitat loss is a key driver of wildlife decline (Lwin *et al.*, 2021). Low detection may also be linked to the short survey period. Species such as bushbuck, Common Genet, and dik-dik were consistently recorded, while crested porcupine and scrub hare, observed previously by Werema, (2013), were not detected during this study. Conversely, species like Common duiker, Serval Cat, and Spotted hyena were recorded in this survey but not in earlier studies, indicating an updated and expanded species list. The presence of the

black-and-white colobus monkey at the Two Rivers site highlights the existence of healthy forest patches supporting rich biodiversity. These findings emphasize the need to protect and preserve remaining habitat patches. Regular surveys and additional techniques, such as camera traps and opportunistic observations, are recommended to further enrich species inventories.

Small mammal species are important as they maintain ecosystem balance, support plant regeneration, biodiversity, and food webs, act as indicators of environmental health, and contribute to scientific research and agriculture. In Africa, including Tanzania, small mammals play a key role in sustaining forests, grasslands, and farmlands by maintaining healthy ecosystems.

A total of 93 individuals representing 8 small mammal species were recorded, reflecting habitat complexity in the surveyed transects. Compared to Werema *et al.* (2013), who recorded 10 species in AKU conservation areas, this survey added three new records: *Aethomys chrysophilus*, *Arvicanthis niloticus*, and *Xerus rutilus*. These additions enhance the small mammal checklist for AKU, suggesting improving habitat quality and availability of essential resources. Significant spatial variation in species distribution among transects likely reflects differences in food availability. While *Mastomys natalensis* was the frequently captured species in the AKU area, the higher abundance of *Mastomys natalensis* varied across habitats and transects. The highest numbers of individuals found in the cultivated land probably due to greater availability of food and nesting resources which may have enhanced their reproductive success.. This pattern aligns with findings by Makundi (1999), Werema *et al.* (2013), Magige (2016), Nkwabi *et al.* (2018), and Rwebuga *et al.* (2023), who also observed higher densities of small mammal in cultivated compared to non-cultivated habitats. Most recorded species were seed and grass eaters, abundant in cultivated areas, though their prevalence may also indicate habitat disturbance.

CONCLUSION AND RECOMMENDATIONS

The study demonstrates that the AKU–ACER landscapes support important populations of large, medium, and small mammals within a peri-urban setting. Mammal diversity was higher at the Manyara site than at the Two Rivers one, though both sites remain valuable for conservation. The presence of a black-and-white colobus monkey at Two Rivers indicates relatively healthy forest conditions. Overall, the persistence of mammals of conservation importance highlights the need for stronger habitat protection, improved ecological management, and long-term monitoring, particularly given the short and seasonal nature of the survey. The study recommends

protecting critical forest, riparian, and natural habitats; integrating mammal conservation into land-use and development planning to maintain ecological connectivity; establishing long-term and seasonal mammal monitoring programmes; restoring degraded habitats using native vegetation; safeguarding species of conservation concern through site-specific management; and strengthening institutional capacity, awareness, and stewardship among AKU stakeholders and surrounding communities.

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BIRD SPECIES COMPOSITION, DIVERSITY, AND ABUNDANCE: THE CASE OF THE AGA KHAN UNIVERSITY LANDS IN ARUSHA, TANZANIA

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ABSTRACT

Urban green spaces play an important role in harbouring avifauna species, and have been considered to possess a vital role in the conservation of animals including birds against urban alterations. Despite little being known about the birds' species distribution, diversity, richness, and abundance in the urban green spaces particularly in Arusha region. This study tried to fill that gap by assessing the composition, abundance and diversity of birds in Aga Khan University – Arusha Climate and Environmental Research Centre in Arusha city. The study employed ground transect bird surveys in 13 created transects. About 1533 individuals' birds belonging to 18 orders, 46 families, 146 species and 104 genera were recorded in the area. Helmeted Guineafowl (*Numida meleagris*) was frequently observed (n = 129 individuals; 8.398%). Two bird species were classified by IUCN Red List as of conservation concern, the African white-backed vulture (*Gyps africanus*) categorised as Critically Endangered (CR, n = 1); and the Tawny Eagle *Aquila rapax* as Vulnerable (VU, n = 1), and the rest were of Least Concern (LC). We observed a higher abundance of birds in Decision Hill, which can be ascribed to the complex habitats. In this survey we also recorded forest-dependent bird species at two rivers sites in the Usa River. Knowing what natural resources (i.e., birds) are and their distribution in an area is critical to providing land managers with options that combine both biodiversity conservation and proper land use plans. Urban green spaces are proved to have high potential in supporting bird biodiversity, thus government, policy makers and all biodiversity stakeholders should provide clear actions to establish, manage and protect the green spaces to support birds' species conservation.

Key words: Abundance, birds, diversity, habitats, threatened species, urban green spaces

INTRODUCTION

Urban Green spaces are usually, but not always, comprised of vegetation and associated with natural elements in town and cities (Łopucki and Kiersztyn, 2015). Currently there has been growing interest in urban green spaces research due to evidence that nature positively impacts human wellbeing (Taylor and Hochuli, 2017; Palliwoda and Priess, 2021). Urban green spaces provide

habitats for biodiversity including birds which occur in many ranges of habitats, and hence are sensitive to environmental change and have great resonance with the public (Masolele *et al.*, 2017; Werema *et al.*, 2019; Mgelwa *et al.*, 2023). Birds act as biodiversity indicators for several reasons. Birds are also more sensitive to environmental contaminants than other vertebrates, therefore can act as bio-indicators (Ducatez and Shine, 2017; Senzaki *et al.*, 2020; Seri and Rahman, 2021; Habibullah *et*

al., 2022; Nkwabi *et al.*, 2025). Besides, often birds are the focus of volunteer efforts and there is the possibility to involve communities in monitoring schemes and action (Lyamuya *et al.*, 2021). The Aga Khan University (AKU) is a university-level research centre that provides a unique “living lab” for climate change and environmental sustainability studies. Like many other rangeland areas, the AKU Arusha land sites have been observed to face critical threats to birds which are popular subjects for research and monitoring, and long-term datasets of bird counts often provide a useful resource as indicators of ecological change (Senzaki *et al.*, 2020; Habibullah *et al.*, 2022; Nkwabi *et al.*, 2025).

Urban green spaces offer homes for different wild animals and are considered to play a vital role in the conservation of animals including birds, in the face of urban alterations (Schultz *et al.*, 2012; Werema *et al.*, 2019; 2025; Bhakti *et al.*, 2025). In some urban landscapes, these patches have been regarded as biodiversity hotspots (Horta *et al.*, 2018; Bhakti *et al.*, 2025) and wild animal species inhabiting are determined by size of the patch, and access to other patches or landscapes of the surrounding area (Bux *et al.*, 2021; Werema *et al.*, 2025; Bhakti *et al.*, 2025). In some occasions, these patches can provide refuge for species whose native habitats have been largely lost (Smith, *et al.*, 2018; Gentili *et al.*, 2024), even allowing for the inhabitation of forest specialists (Maseko *et al.* 2020; Werema *et al.*, 2025). The patches of forest, thickets or woodlands in urban landscapes may also provide dispersal corridors or stepping stones among habitat patches (Sandström *et al.*, 2006; Evans *et al.*, 2009). There are a few birds’ studies in Tanzania which have been conducted in urban green spaces which have natural vegetation to assess how patches of vegetation, are of conservation importance. No studies have been conducted in Arusha region, particularly in a well-established green space like AKU-ACER. Many avifauna studies have been done in Tanga (Kakoko and John, 2013), Morogoro (Rija *et al.* 2014) and Dar es Salaam (Harvey

and Howell, 1987; Mlingwa, 1992; Werema *et al.*, 2019; John & Kagembe, 2022; Werema and Wilson, 2022; Werema *et al.*, 2025). Tanzania is becoming more populated. Towns and cities are increasing in size and human activities continue to reduce the already fragmented natural vegetation resulting in reduced green space in urban areas.

The Arusha Climate and Environmental Research Centre (ACER) faces a challenge of biodiversity loss of both flora and fauna due to degradation resulting from peri-urban proliferation of human activities related to crop cultivation, livestock keeping, and construction and development of human settlements (Werema *et al.*, 2013). Increased human population and their emerging technologies are posing a threat to biodiversity resource, while climate change is compounding these challenges. Currently, the Government is advocating for improving conservation of our biodiversity knowledge. Therefore, this study aimed at assessing the distribution, richness, diversity and abundance of bird species in two sites located on the ACER.

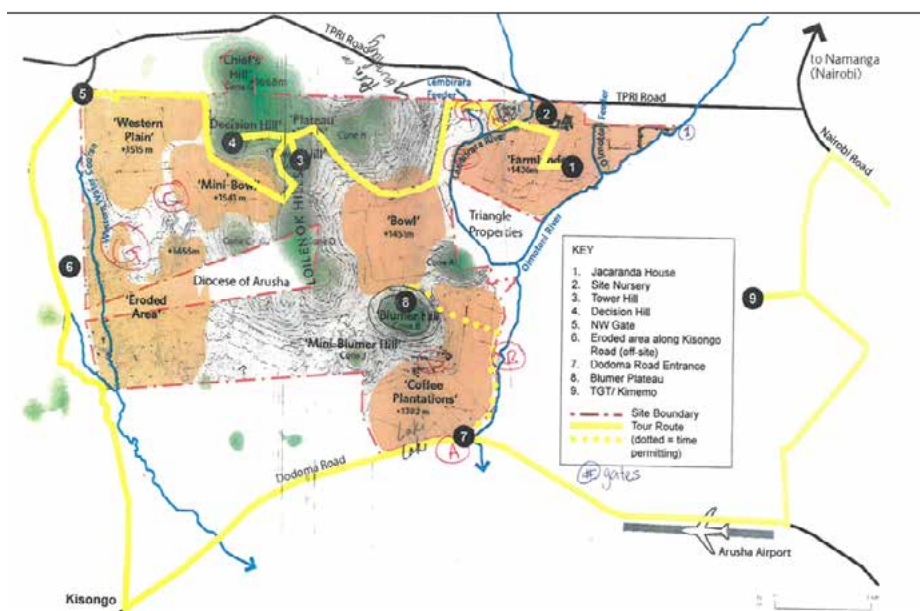
MATERIALS AND METHODS

Study area

This study was conducted in the Arusha Municipality in Tanzania. The Arusha municipality lies between 1.6° and 4.0° latitude south and 34.5° and 37.5° longitude east. Two study sites from Arusha Municipal were chosen to study the avifauna diversity, composition and abundance. The two sites selected include Manyara and Two rivers site, all of which are located in the Aga Khan University. The Manyara site is located on the outskirts of Arusha, 13 km west of Arusha city centre, 3 km north of Arusha airport, and overlooks the pastoral herding communities and smallholder farms to the west and north (Map. 1). The site has 3,200 acres of variable land that is ecologically important. It has seven hills, degraded areas on the western side with gully erosions, and productive land which is currently used for

farming activities. For the last 10 years AKU has been operating a reforestation programme

to address overgrazing and deforested areas and support biodiversity.



Map 8: AKU Arusha Manyara Site - dotted redline is the study area boundary (Source: TAWIRI GIS Office, 2025)

The Two Rivers site is 2km by 1km, about 500 acres and located between two tributaries at Usa River and 14km east of Arusha city centre (Map. 2). Currently, about 80% of the land is under cultivation and the remaining 20%

consists of scattered trees, shrubland, and open grass areas. The general characteristics of the area include high precipitation, fertile soil, and diverse vegetation.

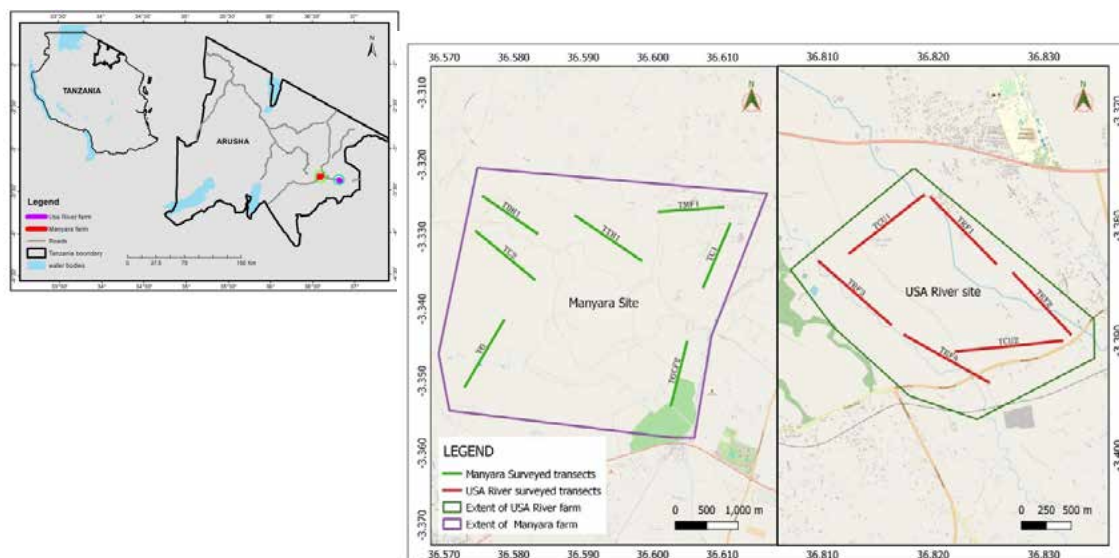


Map 9: AKU Arusha Two Rivers Site – left top-down frames are total cultivated and uncultivated areas respectively. Right top-down frames are grassland and shrubland (Source: TAWIRI GIS Office, 2025).

Transects designs

A total of thirteen transects were established (Map. 3), of which seven (7) transects were in the Manyara site: namely, Mixed forest consisting of Eucalyptus plantation and natural forests (TMF1), Graviella and Coffee forest (TGCF1), Cultivation site one (CT1), Cultivation site two (TC2), Tower Hill

(TTH1), Decision Hill (TDH1) and Grassland (TG1). At Two river sites 6 transects were established namely Riverine Forest 1 (TRF1), Riverine Forest 2 (TRF2), Riverine Forest 3 (TRF3), Riverine Forest 4 (TRF4), Cultivation 1(TCU1) and Cultivation 2 (TCU2). All Transects established were of 1km length and all transects were attended by walking.



Map 10: Study site and Transect layout at Manyara and two River site-USA-River in Arusha City (Source: TAWIRI GIS Office, 2025).

Data collection

Data were collected by walking in a line transect of 1km in each sub zone as per selected habitat in the survey. The survey was conducted from December 2023 to January 2024; we assessed bird species at 06:45 AM – 11:45AM and 16:00 PM – 18:30 PM on every day of the survey. Transects were placed based on vegetation types and accessibility. Two observers walked along transects recording all bird species encountered and their abundance. The Stevenson and Fanshawe (2020) guidebook was used in identification of species.

Data analysis

Species richness and diversity estimates were obtained using the Species Diversity

and Richness (SDR) computer programme (Seaby & Henderson 2007). The abundance of birds per transects in each site surveyed were computed using the PAST 4.03 statistical software (Hammer *et al.*, 2001). Additionally, birds were categorized based on their migratory status as Intra-African Migrant (IAM), Palearctic Migrant (PAM), and Local Movement (LoM). Such a categorization may provide a temporal dimension to the findings, shedding light on the seasonal movements and ecological dependencies of different species. This detailed information from the bird survey is instrumental for conservation planning and management, contributing to a more holistic understanding of avian biodiversity in Northern Tanzania.

RESULTS

Bird species richness

Overall, a total of 146 birds species recorded in the survey provided a valuable insight into the avian diversity of the region, with each species

contributing to the overall ecological richness of the area. These species of birds belong to 1533 individuals, 18 orders, 46 family and 104 genera. Transects in grassland (TG) had the highest number of species (51), followed by Tower Hill-TTH (45), while Gravillea and Coffee forest - TGCF had the least 17 species (Fig.1)

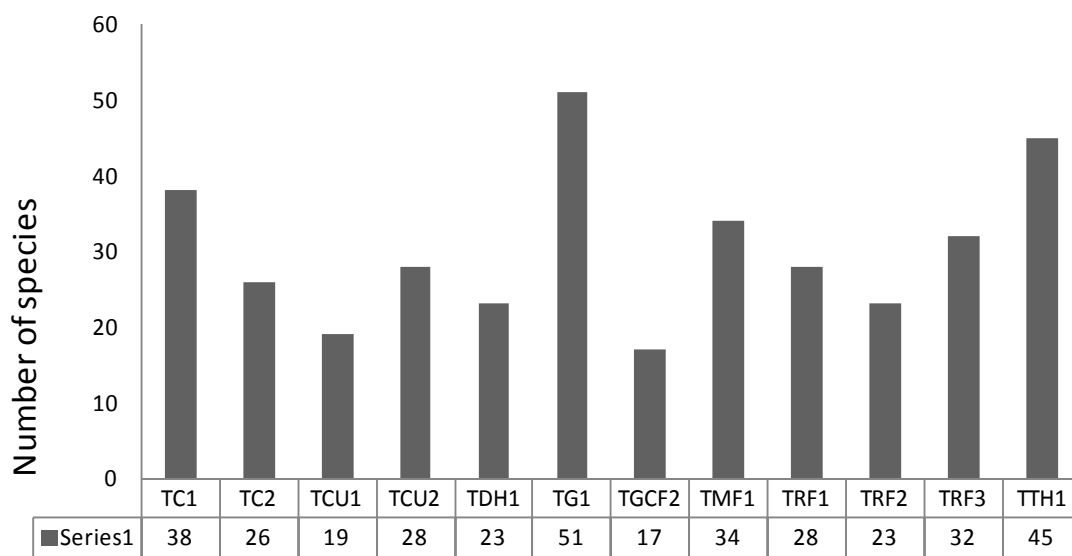


Figure 1: Presentation of richness of bird species (number of species) in different transects surveyed in the Aga Khan University conservation areas, Northern Tanzania. TMF1= Transect Mixed Forest one; TGCF1=Transect Gravillea and Coffee Forest one; TC1=Transect Cultivation one; TC2= Transect Cultivation two; (TTH1= Transect Tower Hill one; TDH1=Transect Decision Hill one; TG1=Transect Grassland one; TRF1= Transect Riverine Forest one; TRF2=Transect Riverine Forest two; TRF3=Transect Riverine Forest three; TRF4=Transect Riverine Forest four; TCU1=Transect Cultivation one and TCU2=Transect Cultivation two.

Migration status

Two species classified by IUCN Red List of conservation concern were recorded. The first species observed was the African white-backed vulture *Gyps africanus* categorized by IUCN Red List as Critically endangered-CR, n = 1; and the second was the Tawny Eagle *Aquila rapax* categorized as Vulnerable (VU, n = 1), while the rest were of Least concern (LC) (Appendix 1). Five listed under the African-Eurasian Migratory Waterbird Agreement (AEWA) were recorded in the AKU environmental

conservation areas (Table 1). This included the Klaas's Cuckoo (*Chrysococcyx klaas*), an Intra-African Migrant (IAM) migrating within African continent which was recorded from Manyara site at TTHI. The following Palearctic migrants (PAM) were recorded from different areas of Manyara and Usa River sites including the European Roller (*Coracias garrulus*) observed in TG1, the Lanner Falcon (*Falco biarmicus*) recorded in TC1, the Red-rumped Swallow (*Cecropis daurica*) observed in TTH1, the White Stork (*Ciconia ciconia*) observed in TCU2 and TRF2 (Table 1).

Table 1: Migrating bird species recorded in different transects created in AKU areas

English IOC	Scientific Name	Manyara				Two River			Grand Total
		TC2	TG1	TTH1	Total	TCU2	TRF2	Total	
European Roller	<i>Coracias garrulus</i>		4		4				4
Klaas's Cuckoo	<i>Chrysococcyx klaas</i>			2	2				2
Lanner Falcon	<i>Falco biarmicus</i>	2			2				2
Red-rumped Swallow	<i>Cecropis daurica</i>			6	6				6
White Stork	<i>Ciconia ciconia</i>					2	3	5	5
Grand Total		2	4	8	14	2	3	5	19

Abundance of Birds

The result from transects created at TDH1 in Manyara site showed the highest value of mean abundance of birds (5.696 ± 1.136), followed by the mixed forest in Manyara site

(5.324 ± 1.487), and the two rivers forest-TRF1 (5.214286 ± 1.170). Low mean abundance was recorded in the second forest transect TRF2 (2.826087 ± 0.306 ; Fig. 2). The overall AKU mean abundance of bird species was low.

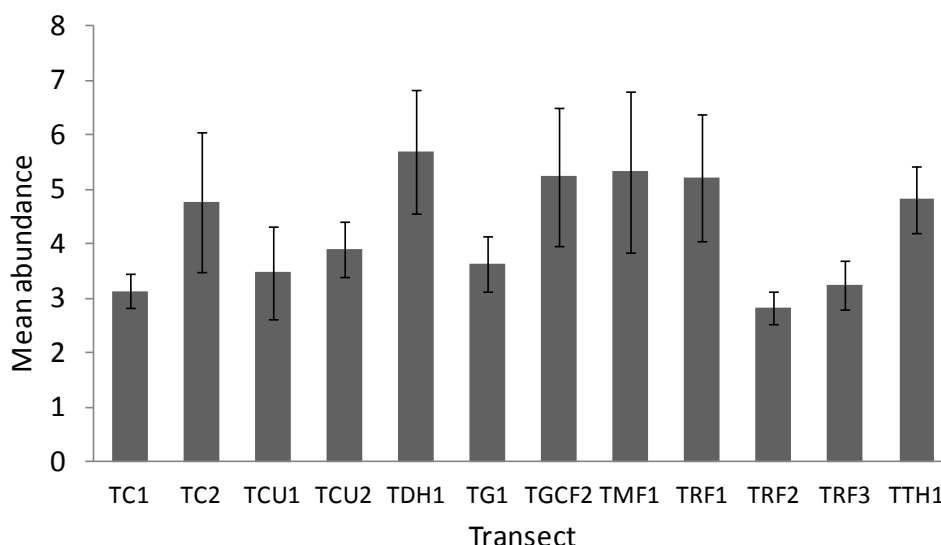


Figure 2: Bird abundance in different transects in the Aga Khan University campus areas, Northern Tanzania. Vertical lines indicate $\pm$ SE.

Bird diversity

Diversity of birds in grassland (TG1) was the highest ($H' = 3.631$), followed by Tower Hill ($H' = 3.506$), cultivated-TC1, ($H' = 3.487$), mixed forest-TMF1 ($H' = 2.893$), Decision Hill-TDH1 ($H' = 2.771$), cultivated-TC2 in

Manyara ($H' = 2.725$), and cultivated-TCU1 ($H' = 2.629$). Two rivers sites include; transect forest-TRF3 ($H' = 3.254$), cultivated-TCU2 in USA River ($H' = 3.132$), two rivers forest-TRF2 ($H' = 3.033$), two rivers forest TRF1 ($H' = 2.944$), and the least diversity in Gravillea and Coffee forest - TGCF1 ($H' = 2.485$).

Common bird species

The Helmeted Guineafowl (*Numida meleagris*) was frequently observed (129 individuals; 8.398%) in the AKU conservation area, followed by the Red-eyed Dove (*Streptopelia semitorquata*) (97 individuals; 6.315%) see Figure 6. Other bird species which were commonly observed included the Common Bulbul (*Pycnonotus barbatus*), the Speckled

Mousebird (*Colius striatus*), the Yellow Bishop (*Euplectes capensis*), Reichenow's Seedeater (*Crithagra reichenowi*), the Baglaffeht Weaver (*Ploceus baglaffeht*), the Bleating Warbler (*Camaroptera brevicaudata*), the Winding Cisticola (*Cisticola marginatus*), the African Firefinch (*Lagonosticta rubricata*), the Tawny-flanked Prinia (*Prinia subflava*), and the Tropical Boubou (*Laniarius major*, Fig. 3).

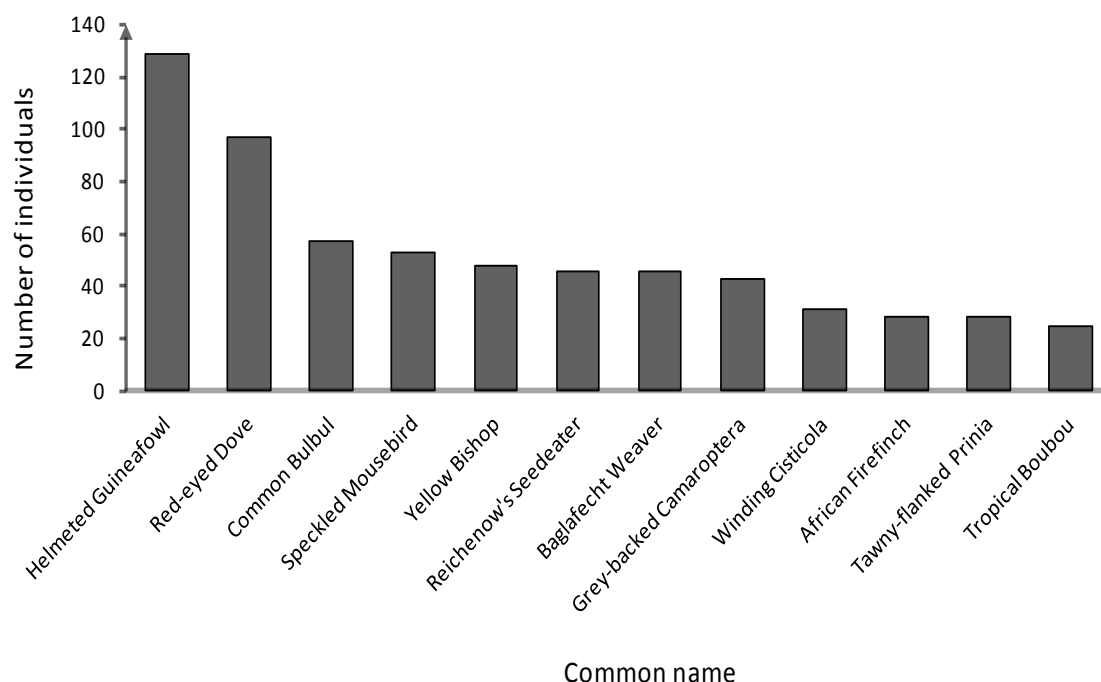


Figure 3: Presentation of selected bird species commonly observed in higher numbers in the Aga Khan University campus areas, Northern Tanzania.

Forest dependent species

In this survey we also recorded forest dependent bird species see Table 2. The species were recorded at two rivers site in Usa River. The Common Bulbul (*Pycnonotus barbatus*) was not forest dependant species but we included

it in this group because the species shares a similar family (Pycnonotidae) with the species presented here. The Common Bulbul was frequently observed (n=57) in the study area in many census points (Table 2) and hence can not be called a habitat specialist but rather can be termed as a habitat generalist.

Table 2: Common Bulbul and Forest dependent bird species recorded in AKU environmental conservation areas.

English IOC	Scientific Name	Manyara					Two rivers						Grand Total
		TC1	TGCF2	TMF1	TTH1	Total	TCU1	TCU2	TRF1	TRF2	TRF3	Total	
Common Bulbul	<i>Pycnonotus barbatus</i>	10	4	10	7	31	4	2	11	5	4	26	57
Grey-olive Greenbul	<i>Phyllastrephus cerviniventris</i>	-	-	-	-	-	-	-	5	-	2	7	7
Kenrick’s Starling	<i>Poeoptera kenricki</i>	-	-	-	-	-	-	-	-	-	8	8	8
Little Greenbul	<i>Eurillas virens</i>	-	-	-	-	-	-	-	3	5	-	8	8
Shelley’s Greenbul	<i>Arizelocichla masukuensis</i>	-	-	-	-	-	-	-	2	-	-	2	2
Sombre Greenbul	<i>Andropadus importunes</i>	-	-	-	-	-	-	-	-	2	-	2	2
Yellow-bellied Greenbul	<i>Chlorocichla flaviventris</i>	-	-	-	-	-	-	-	-	-	2	2	2
Grand Total		10	4	10	7	31	4	2	21	12	16	55	86

DISCUSSION

This study recorded a total of 1533 individuals in 146 bird species within the entire AKU-ACER area indicating that the centre had sufficiently different landscapes which accommodated a variety of habitat heterogeneity while providing habitat requirements for breeding sites, nesting materials, and cover for shading and protection of birds from predators. The study further showed that the habitat types found in ACER were significant areas for the conservation of biodiversity, and hotspot habitats for bird watching. Among many other bio-indicators, groups of fauna species such as birds are excellent for the health of the environment and ecological/ environmental change and can be used to monitor changes. The results obtained, indicated an increase of bird species when compared to Werema *et al.* (2013) bird survey in AKU areas which reported 7,574 individuals of 83 species of birds. The higher number of individuals reported in Werema *et al.* (2013) was caused by the presence of queleas which can be ten times more abundant in agricultural lands when encountered. In this survey we did

not observe this species, which is why we have a low number of individuals reported.

Observation of birds of conservation concern in AKU areas, including the African white-backed Vulture (*Gyps africanus*) and the Tawny Eagle (*Aquila rapax*) as classified by the IUCN Red List as threatened implies that the habitat is developing to habitat complexity and will continue to gain a heterogeneous condition that can provide areas for exploitation of the available resources for animals. The African white-backed vulture is known as one of the oldest vultures in the world in the family Accipitridae, which also includes other birds such as eagles, kites, buzzards and hawks. The white-backed vulture, like many African vultures, has suffered a rapid decline (Bamford *et al.*, 2009; Ogada *et al.*, 2012, Kendall *et al.*, 2012; 2014; Cook *et al.*, 2025). Literature show that, in 1988 when it was assessed for the first time, it was classified as a species of least concern with a wide range and large population. When it was re-assessed in 2007, it was put in the Near Threatened species IUCN Red List and in 2012, when it was confirmed that in the next three generations the white-backed vulture

would decrease by 50%. It was classified as Endangered and in October 2015, it was further uplisted to Critically Endangered because the decline had reached a magnitude that puts the vulture at extreme risk of extinction.

The decline in the number of vultures is the result of human induced activities and other ecological changes including climate change related factors (Bamford *et al.*, 2009; Ogada & Buij, 2011; Ogada *et al.*, 2012; Kendall *et al.*, 2012; Williams *et al.*, 2025). The Tawny Eagle belongs to the family Accipitridae. It is perhaps the most highly opportunistic of all of its taxonomic clan but it is facing rapid decline throughout its range. Both birds are threatened due to numerous factors, particularly loss of habitat as a result of anthropogenic activities and global warming, largely through poisoning and other anthropogenic mortality (mainly through contact with various man-made objects) which are perhaps driving the once numerous Tawny Eagle to the brink of extinction.

These findings signify the importance of wild habitats in expanding urban areas as important habitats for conservation of biodiversity. The ACERC once protected from human activities like agriculture will present natural wild habitats in the growing city of Arusha. Therefore, it will present an opportunity for the conservation of specific threatened species and these species can be considered as 'charismatic' species of the area, while we recognize the role of urban habitats in human health growing, it is also important to acknowledge the relative importance of charismatic species to urban human communities in conservation and management strategies.

Bird species recorded in the surveyed area of AKU provided a valuable insight into the avian diversity, richness and abundance in the areas, with each species contributing to their overall ecological richness and diversity. Higher abundance of birds in TDH1 can be ascribed to the availability of grass layers and shrub patches (complex habitats) that provide potential protection from predators

and exploitation of available resources. Also, individuals were observed moving to nearby transect areas to forage on grasses and shrubs. Higher species abundance observed in TDH1 can also be due to immigration of birds that are adapted to forage on long grasses and also due to increased visibility of food items (Nkwabi *et al.* 2015). It is generally accepted that animal species abundance and diversity increase with increasing habitat complexity, given that more complex habitats offer a greater variety for potential exploitation of available resources (Tews *et al.*, 2004; Smith *et al.*, 2014). For example, Brown (2008) reported that bird species abundance and diversity increased with increase in grass height in America. A study by Tu *et al.* (2020) also reported increase in bird species evenness as grass height and density increases in urban green spaces of Taiwan. However, the present results conform more to the intermediate disturbance hypothesis developed by Connell (1978) which suggests that the highest abundance and diversity of living things is maintained at intermediate scales of disturbance as in Decision Hill. Nkwabi *et al.* (2015) reported that both bird species diversity and composition change with grass height.

Common bird species in AKU conservation area

A higher number of Helmeted Guineafowl in AKU may imply that there is a greater availability of resources in those habitats, including presence of reliable breeding sites, nesting materials, and cover for shading and protection from predators. It is possible that the increase in numbers of the Helmeted Guineafowl was accelerated after stopping agriculture in the AKU areas. Investigation by Ratcliffe & Crowe (2001b) on the effects of agriculture on populations of Helmeted Guineafowl in the Midlands of KwaZulu-Natal province, South Africa presented that the populations of Helmeted Guineafowl declined with increasingly intensive agriculture which resulted in the disappearance of edge habitat and associated, optimally fragmented habitat

mosaic which resulted in the disappearance of food and cover for birds. However, more research is needed to establish the reason for this in the study area.

Forest dependent species

The presence of forest dependent species including the Grey-olive Greenbul (*Phyllastrephus cerviniventris*), Kenrick's Starling (*Poeoptera kenricki*), the Little Greenbul (*Eurillas virens*), Shelley's Greenbul (*Arizelocichla masukuensis*), the Sombre Greenbul (*Andropadus importunes*) and the Yellow-bellied Greenbul (*Chlorocichla flaviventris*), in the forest of two rivers indicate that there was habitat heterogeneity of the area surveyed. Most of the bird species that are habitat specific and forest dependent in the study area signify the importance of maintaining the habitats in pristine condition. This is because either destruction or changes due to anthropogenic or climatic effects can even result in local extinction of more than half of bird species in the area. There is no recent credible and reliable documentation of this species in the two sites surveyed, Manyara and two rivers-Usa River. Werema *et al* (2013) did not report the presence of such species. The species was probably not there and hopefully they have immigrated after the areas have started to resume their former status after reducing human activities.

Birds seasonality and migration

The presence of five species of birds listed under the African-Eurasian Migratory Waterbird Agreement (AEWA) in AKU indicated that the habitats have improved after the reduction of human activities, by comparison with the survey by Werema *et al.* (2013) in AKU which recorded and reported only two species.

Study limitation

It should be taken into consideration that, the data analyzed in this report were from a short-term survey conducted in early wet

season only, which might limit or influence the obtained results. Therefore, we think we would obtain more accurate and comprehensive results if we collected data from both dry and wet seasons. In addition, by conducting these surveys at different intervals of time, the obtained results will provide us with insights into the environmental health of the Aga Khan University conservation area. This will allow us to suggest better mitigation measures and management decisions for its conservation. It is expected that the protection of the AKU will be improved from the time this study was conducted to ensure sustainable biodiversity conservation.

CONCLUSION AND RECOMMENDATIONS

This study concludes that the ACERC landscapes are vital for bird conservation as they harbor a total of 146 bird species, 18 orders, 46 family and 104 genera recorded in the study area which have provided a valuable insight into the avian diversity of the region, with each species contributing to the overall ecological richness and diversity of the area. These results emphasize the need to increase conservation efforts of bird species in the ACER. We also suggest that the adjacent human communities be involved in the conservation of biodiversity in the area as encroachment on the boundaries. is growing Birds were seen in every area and habitat types visited. Both terrestrial and aquatic birds along two rivers are available in their respective habitats. Importantly, the AKU area has the essential life expectancy resources in terms of shelter, nesting and forage for birds that support survival. The data generated from this study will guide future research, conservation planning, and if possible will recover and open up tourism in the Manyara and Two Rivers sites.

Locally, the study has demonstrated the role of one of the remnant forests found in the urban landscape in conservation of local birds. When designing and planning for new towns, planners should think of allocating large remnant patches

in the urban landscapes in order to preserve the fauna and flora biodiversity that depend on these green spaces or thickets. Further surveys will make recommendations on other fauna including ecosystem service providers such as pollinators (e.g. butterflies and bees), This will enable a deeper understanding of the health of the ecosystem of the AKU.

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Appendix 1: Checklist of birds observed in the Aga Khan University campus areas

Key: LC= Least Concern , VU= Vulnerable; CR=Critically Endangered;

SN	Order	Family	English IOC	Scientific Name	IUCN	Migratory
1	Accipitriformes	Accipitridae	African Goshawk	<i>Accipiter tachiro</i>	LC	LOM
2	Accipitriformes	Accipitridae	African Harrier-Hawk	<i>Polyboroides typus</i>	LC	LOM
3	Accipitriformes	Accipitridae	African Hawk-Eagle	<i>Aquila spilogaster</i>	LC	LOM
4	Accipitriformes	Accipitridae	Augur Buzzard	<i>Buteo augur</i>	LC	PAM
5	Accipitriformes	Accipitridae	Black Kite	<i>Milvus migrans</i>	LC	IAM
6	Accipitriformes	Accipitridae	Black-winged Kite	<i>Elanus caeruleus</i>	LC	LOM
7	Accipitriformes	Accipitridae	Common Buzzard	<i>Buteo buteo</i>	LC	IAM
8	Accipitriformes	Accipitridae	Lizard Buzzard	<i>Kaupifalco monogrammicus</i>	LC	LOM
9	Accipitriformes	Accipitridae	Long-crested Eagle	<i>Lophaetus occipitalis</i>	LC	PAM
10	Accipitriformes	Accipitridae	Tawny Eagle	<i>Aquila rapax</i>	VU	LOM
11	Accipitriformes	Accipitridae	White-backed Vulture	<i>Gyps africanus</i>	CR	LOM
12	Apodiformes	Apodidae	African Palm Swift	<i>Cypsiurus parvus</i>	LC	LOM
13	Apodiformes	Apodidae	Little Swift	<i>Apus affinis</i>	LC	LOM
14	Apodiformes	Apodidae	Scarce Swift	<i>Schoutedenapus myoptilus</i>	LC	LOM
15	Apodiformes	Apodidae	White-rumped Swift	<i>Apus caffer</i>	LC	LOM
16	Ciconiiformes	Ciconiidae	Marabou Stork	<i>Leptoptilos crumenifer</i>	LC	LOM

SN	Order	Family	English IOC	Scientific Name	IUCN	Migratory
17	Ciconiiformes	Ciconiidae	White Stork	<i>Ciconia ciconia</i>	LC	IAM
18	Coliiformes	Coliidae	Blue-naped Mousebird	<i>Urocolius macrourus</i>	LC	LOM
19	Coliiformes	Coliidae	Speckled Mousebird	<i>Colius striatus</i>	LC	LOM
20	Columbiformes	Columbidae	Emerald-spotted Wood Dove	<i>Turtur chalcospilos</i>	LC	LOM
21	Columbiformes	Columbidae	Laughing Dove	<i>Spilopelia senegalensis</i>	LC	LOM
22	Columbiformes	Columbidae	Red-eyed Dove	<i>Streptopelia semitorquata</i>	LC	LOM
23	Columbiformes	Columbidae	Red-faced Cisticola	<i>Cisticola erythrops</i>	LC	LOM
24	Columbiformes	Columbidae	Ring-necked Dove	<i>Streptopelia capicola</i>	LC	LOM
25	Columbiformes	Columbidae	Tambourine Dove	<i>Turtur tympanistria</i>	LC	LOM
26	Coraciiformes	Alcedinidae	Malachite Kingfisher	<i>Corythornis cristatus</i>	LC	LOM
27	Coraciiformes	Alcedinidae	Striped Kingfisher	<i>Halcyon chelicuti</i>	LC	LOM
28	Coraciiformes	Coraciidae	European Roller	<i>Coracias garrulus</i>	LC	PAM
29	Coraciiformes	Meropidae	Little Bee-eater	<i>Merops pusillus</i>	LC	LOM
30	Coraciiformes	Meropidae	White-fronted Bee-eater	<i>Merops bullockoides</i>	LC	LOM
31	Cuculiformes	Cuculidae	African Emerald Cuckoo	<i>Chrysococcyx cupreus</i>	LC	LOM
32	Cuculiformes	Cuculidae	Diederik Cuckoo	<i>Chrysococcyx caprius</i>	LC	LOM
33	Cuculiformes	Cuculidae	Green Malkoha	<i>Ceuthmochares australis</i>	LC	LOM
34	Cuculiformes	Cuculidae	Jacobin Cuckoo	<i>Clamator jacobinus</i>	LC	LOM
35	Cuculiformes	Cuculidae	Klaas's Cuckoo	<i>Chrysococcyx klaas</i>	LC	IAM
36	Cuculiformes	Cuculidae	Red-chested Cuckoo	<i>Cuculus solitarius</i>	LC	LOM
37	Cuculiformes	Cuculidae	White-browed Coucal	<i>Centropus superciliosus</i>	LC	LOM
38	Falconiformes	Falconidae	Lanner Falcon	<i>Falco biarmicus</i>	LC	PAM
39	Galliformes	Numididae	Helmeted Guineafowl	<i>Numida meleagris</i>	LC	LOM
40	Galliformes	Phasianidae	Crested Francolin	<i>Ortygornis sephaena</i>	LC	LOM
41	Galliformes	Phasianidae	Harlequin Quail	<i>Coturnix delegorguei</i>	LC	LOM
42	Galliformes	Phasianidae	Red-necked Francolin	<i>Pternistis afer</i>	LC	LOM
43	Galliformes	Phasianidae	Yellow-necked Spurfowl	<i>Pternistis leucoscepus</i>	LC	LOM
44	Gruiformes	Gruidae	Grey Crowned Crane	<i>Balearica regulorum</i>	LC	LOM
45	Musophagiformes	Musophagidae	Brown-faced Go-away-bird	<i>Corythaixoides personatus</i>	LC	LOM
46	Musophagiformes	Musophagidae	White-bellied Go-away-bird	<i>Crinifer leucogaster</i>	LC	LOM

SN	Order	Family	English IOC	Scientific Name	IUCN	Migratory
47	Palecaniformes	Ardeidae	Black-headed Heron	<i>Ardea melanocephala</i>	LC	LOM
48	Passeeriformes	Fringillidae	African Citril	<i>Crithagra citrinelloides</i>	LC	LOM
49	Passeeriformes	Muscicapidae	African Grey Flycatcher	<i>Bradornis microrhynchus</i>	LC	LOM
50	Passeeriformes	Muscicapidae	Ashy Flycatcher	<i>Muscicapa caerulescens</i>	LC	LOM
51	Passeeriformes	Muscicapidae	Collared Palm Thrush	<i>Cichladusa arquata</i>	LC	LOM
52	Passeeriformes	Muscicapidae	White-browed Robin-Chat	<i>Cossypha heuglini</i>	LC	LOM
53	Passeeriformes	Muscicapidae	White-browed Scrub Robin	<i>Cercotrichas leucophrys</i>	LC	LOM
54	Passeeriformes	Zosteropidae	African Yellow White-eye	<i>Zosterops senegalensis</i>	LC	LOM
55	Passeriformes	Corvidae	Pied Crow	<i>Corvus albus</i>	LC	LOM
56	Passeriformes	Laniidae	Long-tailed Fiscal	<i>Lanius cabanisi</i>	LC	LOM
57	Passeriformes	Remizidae	Mouse-colored Penduline Tit	<i>Anthoscopus musculus</i>	LC	LOM
58	Passeriformes	Alaudidae	Fischer's Sparrow-Lark	<i>Eremopterix leucopareia</i>	LC	LOM
59	Passeriformes	Alaudidae	Flappet Lark	<i>Mirafra rufocinnamomea</i>	LC	LOM
60	Passeriformes	Alaudidae	Rufous-naped Lark	<i>Mirafra africana</i>	LC	LOM
61	Passeriformes	Cisticolidae	Buff-bellied Warbler	<i>Phyllolais pulchella</i>	LC	LOM
62	Passeriformes	Cisticolidae	Rattling Cisticola	<i>Cisticola chiniana</i>	LC	LOM
63	Passeriformes	Cisticolidae	Tawny-flanked Prinia	<i>Prinia subflava</i>	LC	LOM
64	Passeriformes	Cisticolidae	Trilling Cisticola	<i>Cisticola woosnami</i>	LC	LOM
65	Passeriformes	Cisticolidae	Winding Cisticola	<i>Cisticola marginatus</i>	LC	LOM
66	Passeriformes	Cisticolidae	Yellow-breasted Apalis	<i>Apalis flavida</i>	LC	LOM
67	Passeriformes	Cisticolidae	Zitting Cisticola	<i>Cisticola juncidis</i>	LC	LOM
68	Passeriformes	Dicruridae	Common Square-tailed Drongo	<i>Dicrurus ludwigii</i>	LC	LOM
69	Passeriformes	Dicruridae	Fork-tailed Drongo	<i>Dicrurus adsimilis</i>	LC	LOM
70	Passeriformes	Estrildidae	African Firefinch	<i>Lagonosticta rubricata</i>	LC	LOM
71	Passeriformes	Estrildidae	Black-and-white Mannikin	<i>Lonchura bicolor</i>	LC	LOM
72	Passeriformes	Estrildidae	Bronze Mannikin	<i>Lonchura cucullata</i>	LC	LOM
73	Passeriformes	Estrildidae	Common Waxbill	<i>Estrilda astrild</i>	LC	LOM
74	Passeriformes	Estrildidae	Crimson-rumped Waxbill	<i>Estrilda rhodopyga</i>	LC	LOM
75	Passeriformes	Estrildidae	Cut-throat Finch	<i>Amadina fasciata</i>	LC	LOM
76	Passeriformes	Estrildidae	Green-winged Pytilia	<i>Pytilia melba</i>	LC	LOM

SN	Order	Family	English IOC	Scientific Name	IUCN	Migratory
77	Passeriformes	Estrildidae	Red-cheeked Cordon-bleu	<i>Uraeginthus bengalus</i>	LC	LOM
78	Passeriformes	Estrildidae	Red-throated Twinspot	<i>Hypargos niveoguttatus</i>	LC	PAM
79	Passeriformes	Fringillidae	Reichenow's Seedeater	<i>Crithagra reichenowi</i>	LC	LOM
80	Passeriformes	Fringillidae	Streaky Seedeater	<i>Crithagra striolata</i>	LC	LOM
81	Passeriformes	Fringillidae	Yellow-fronted Canary	<i>Crithagra mozambica</i>	LC	LOM
82	Passeriformes	Fringillidae	White-bellied Canary	<i>Crithagra dorsostriata</i>	LC	LOM
83	Passeriformes	Hirundinidae	Barn Swallow	<i>Hirundo rustica</i>	LC	LOM
84	Passeriformes	Hirundinidae	Black Saw-wing	<i>Psaldoprocne pristoptera</i>	LC	LOM
85	Passeriformes	Hirundinidae	Lesser Striped Swallow	<i>Cecropis abyssinica</i>	LC	LOM
86	Passeriformes	Hirundinidae	Mosque Swallow	<i>Cecropis senegalensis</i>	LC	LOM
87	Passeriformes	Hirundinidae	Red-rumped Swallow	<i>Cecropis daurica</i>	LC	PAM
88	Passeriformes	Laniidae	Northern Fiscal	<i>Lanius humeralis</i>	LC	LOM
89	Passeriformes	Laniidae	Red-backed Shrike	<i>Lanius collurio</i>	LC	LOM
90	Passeriformes	Leiotrichidae	Arrow-marked Babbler	<i>Turdoides jardineii</i>	LC	LOM
91	Passeriformes	Malaconotidae	Black-backed Puffback	<i>Dryoscopus cubla</i>	LC	LOM
92	Passeriformes	Malaconotidae	Black-crowned Tchagra	<i>Tchagra senegalus</i>	LC	LOM
93	Passeriformes	Malaconotidae	Brown-crowned Tchagra	<i>Tchagra australis</i>	LC	LOM
94	Passeriformes	Malaconotidae	Brubru	<i>Nilaus afer</i>	LC	LOM
95	Passeriformes	Malaconotidae	Grey-headed Bushshrike	<i>Malaconotus blanchoti</i>	LC	LOM
96	Passeriformes	Malaconotidae	Grey-headed Kingfisher	<i>Halcyon leucocephala</i>	LC	LOM
97	Passeriformes	Malaconotidae	Orange-breasted Bushshrike	<i>Chlorophoneus sulfureopectus</i>	LC	LOM
98	Passeriformes	Malaconotidae	Slate-colored Boubou	<i>Laniarius funebris</i>	LC	LOM
99	Passeriformes	Malaconotidae	Tropical Boubou	<i>Laniarius major</i>	LC	LOM
100	Passeriformes	Monarchidae	African Paradise Flycatcher	<i>Terpsiphone viridis</i>	LC	LOM
101	Passeriformes	Motacillidae	Mountain Wagtail	<i>Motacilla clara</i>	LC	LOM
102	Passeriformes	Motacillidae	Pangani Longclaw	<i>Macronyx aurantiigula</i>	LC	LOM
103	Passeriformes	Muscicapidae	Grey-backed Camaroptera	<i>Camaroptera brevicaudata</i>	LC	LOM
104	Passeriformes	Muscicapidae	Red-capped Robin-Chat	<i>Cossypha natalensis</i>	LC	LOM
105	Passeriformes	Muscicapidae	Rüppell's Robin-Chat	<i>Cossypha semirufa</i>	LC	LOM

SN	Order	Family	English IOC	Scientific Name	IUCN	Migratory
106	Passeriformes	Muscicapidae	Spotted Palm Thrush	<i>Cichladusa guttata</i>	LC	LOM
107	Passeriformes	Nectariniidae	Amethyst Sunbird	<i>Chalcomitra amethystina</i>	LC	LOM
108	Passeriformes	Nectariniidae	Beautiful Sunbird	<i>Cinnyris pulchellus</i>	LC	LOM
109	Passeriformes	Nectariniidae	Collared Sunbird	<i>Hedydipna collaris</i>	LC	LOM
110	Passeriformes	Nectariniidae	Scarlet-chested Sunbird	<i>Chalcomitra senegalensis</i>	LC	LOM
111	Passeriformes	Nectariniidae	Variable Sunbird	<i>Cinnyris venustus</i>	LC	LOM
112	Passeriformes	Oriolidae	Black-headed Oriole	<i>Oriolus larvatus</i>	LC	LOM
113	Passeriformes	Passeridae	Chestnut Sparrow	<i>Passer eminibey</i>	LC	LOM
114	Passeriformes	Platysteiridae	Chinspot Batis	<i>Batis molitor</i>	LC	LOM
115	Passeriformes	Ploceidae	Baglafecht Weaver	<i>Ploceus baglafecht</i>	LC	LOM
116	Passeriformes	Ploceidae	Black Bishop	<i>Euplectes gierowii</i>	LC	LOM
117	Passeriformes	Ploceidae	Chestnut Weaver	<i>Ploceus rubiginosus</i>	LC	LOM
118	Passeriformes	Ploceidae	Fan-tailed Widowbird	<i>Euplectes axillaris</i>	LC	LOM
119	Passeriformes	Ploceidae	Golden-backed Weaver	<i>Ploceus jacksoni</i>	LC	LOM
120	Passeriformes	Ploceidae	Lesser Masked Weaver	<i>Ploceus intermedius</i>	LC	LOM
121	Passeriformes	Ploceidae	Red-collared Widowbird	<i>Euplectes ardens</i>	LC	LOM
122	Passeriformes	Ploceidae	Southern Red Bishop	<i>Euplectes orix</i>	LC	LOM
123	Passeriformes	Ploceidae	Spectacled Weaver	<i>Ploceus ocularis</i>	LC	LOM
124	Passeriformes	Ploceidae	Thick-billed Weaver	<i>Amblyospiza albifrons</i>	LC	LOM
125	Passeriformes	Ploceidae	Yellow Bishop	<i>Euplectes capensis</i>	LC	LOM
126	Passeriformes	Ploceidae	Yellow-bellied Eremomela	<i>Eremomela icteropygialis</i>	LC	LOM
127	Passeriformes	Pycnonotidae	Common Bulbul	<i>Pycnonotus barbatus</i>	LC	LOM
128	Passeriformes	Pycnonotidae	Grey-olive Greenbul	<i>Phyllastrephus cerviniventris</i>	LC	LOM
129	Passeriformes	Pycnonotidae	Little Greenbul	<i>Eurillas virens</i>	LC	LOM
130	Passeriformes	Pycnonotidae	Shelley's Greenbul	<i>Arizeolocichla masukuensis</i>	LC	LOM
131	Passeriformes	Pycnonotidae	Sombre Greenbul	<i>Andropadus importunus</i>	LC	LOM
132	Passeriformes	Pycnonotidae	Yellow-bellied Greenbul	<i>Chlorocichla flaviventris</i>	LC	LOM
133	Passeriformes	Sturnidae	Kenrick's Starling	<i>Poeoptera kenricki</i>	LC	LOM
134	Passeriformes	Sturnidae	Superb Starling	<i>Lamprotornis superbus</i>	LC	LOM
135	Passeriformes	Sturnidae	Violet-backed Starling	<i>Cinnyricinclus leucogaster</i>	LC	LOM
136	Passeriformes	Sylviidae	Banded Warbler	<i>Sylvia boehmi</i>	LC	LOM

SN	Order	Family	English IOC	Scientific Name	IUCN	Migratory
137	Passeriformes	Vangidae	White-crested Helmetshrike	<i>Prionops plumatus</i>	LC	LOM
138	Passeriformes	Viduidae	Pin-tailed Whydah	<i>Vidua macroura</i>	LC	LOM
139	Pelecaniformes	Ardeidae	Western Cattle Egret	<i>Bubulcus ibis</i>	LC	LOM
140	Pelecaniformes	Scopidae	Hamerkop	<i>Scopus umbretta</i>	LC	LOM
141	Pelecaniformes	Threskiornithidae	Hadada Ibis	<i>Bostrychia hagedash</i>	LC	LOM
142	Piciformes	Indicatoridae	Lesser Honeyguide	<i>Indicator minor</i>	LC	LOM
143	Piciformes	Lybiidae	Red-and-yellow Barbet	<i>Trachyphonus erythrocephalus</i>	LC	LOM
144	Piciformes	Lybiidae	White-eared Barbet	<i>Stactolaema leucotis</i>	LC	LOM
145	Piciformes	Picidae	Cardinal Woodpecker	<i>Dendropicos fuscescens</i>	LC	LOM
146	Piciformes	Picidae	Nubian Woodpecker	<i>Campethera nubica</i>	LC	LOM

FIFTEEN YEARS OF EX-SITU RECOVERY FOR THE KIHANSI SPRAY TOAD (*Nectophrynoides asperginis*) IN TANZANIA: DEMOGRAPHY, ENVIRONMENTAL CONTROLS, AND REINTRODUCTION READINESS AT KIHANSI CAPTIVE BREEDING FACILITY

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ABSTRACT

The Kihansi Spray Toad (*Nectophrynoides asperginis*), once extinct in the wild in 2009, has been maintained at the Kihansi Captive Breeding Facility (KCBF) since 2011. Using long-term data from 2011–2025, we assessed population growth, reproduction, mortality, structure, and environmental conditions to evaluate stability and reintroduction readiness. Population size increased from 48 individuals in 2011 to 2,570 in 2019 (mean $\pm$ SD: $1,418 \pm 1,045$), following a log-linear growth rate $r = 27.6\% \text{ yr}^{-1}$ ($b=0.244$; $R^2 = 0.71$; $R^2 = 0.71$). Births consistently exceeded deaths after 2015, and the population structure shifted toward females and juveniles. Short-term monitoring (May–July 2024) showed that relative humidity was within the optimal range (85–95%) for 21–32% of intervals, while temperature (19–24 °C) met the target for 1–17%, indicating the need for improved misting and thermal regulation across the rooms. These findings demonstrate that long-term demographic stability is closely linked to environmental stability. Continuous environmental monitoring, routine calibrations, and sustainable funding for microclimate control and live-prey production are essential in order to sustain population viability. With ongoing improvements and verified restoration of spray-wetland conditions in Kihansi Gorge, the population shows readiness for phased reintroductions in the wild.

Keywords: Amphibian recovery; Assurance colony; Captive breeding; Chytridiomycosis; Conservation management; Ex-situ conservation; *Nectophrynoides asperginis*; Population dynamics; Reintroduction; Tanzania.

INTRODUCTION

Amphibians are undergoing global declines, with over 40% of species threatened with extinction due to habitat loss, climate change, disease, and environmental degradation (IUCN SSC Amphibian Specialist Group, 2024). These declines have intensified the need for integrated ex situ conservation programs that safeguard species while supporting eventual reintroductions to the wild (Mendelson *et al.*,

2019; Tapley *et al.*, 2015). Captive breeding initiatives have become vital components of amphibian recovery strategies, providing insurance populations for species facing rapid population collapses. Programs such as the Amphibian Ark and other zoological initiatives have demonstrated that successful husbandry, environmental control, and monitoring can stabilize highly threatened taxa and facilitate subsequent reintroductions (Zippel *et al.*, 2011; Harding *et al.*, 2016).

Captive breeding of the Kihansi Spray Toad (*Nectophrynoides asperginis*) is one of the most prominent examples of amphibian rescue through ex situ conservation. Endemic to the Kihansi Gorge in Tanzania's Udzungwa Mountains, this live-bearing toad was declared extinct in the wild in 2009 following habitat alteration caused by hydropower development and the subsequent loss of the gorge's characteristic spray-wetland microclimate (Channing *et al.*, 2006; Quinn *et al.*, 2003). To rescue the species, assurance colonies were founded at the Bronx and Toledo Zoos, from which individuals were later repatriated to establish the Kihansi Captive Breeding Facility (KCBF) and a satellite colony at the University of Dar es Salaam. Since 2012, the Tanzania Wildlife Research Institute (TAWIRI) has coordinated national recovery actions under the Ministry of Natural Resources and Tourism (MNRT), integrating captive operations, monitoring, and reintroduction trials with the broader Kihansi ecosystem management program (Howell *et al.*, 2021; TAWIRI, 2022). This institutional collaboration represents one of Africa's most sustained amphibian recovery efforts and provides a model for long-term zoo-based conservation in developing countries.

The success of *N. asperginis* recovery depends on maintaining both demographic and optimal environmental conditions within captivity. Because the species evolved under cool, humid spray-wetland conditions, its survival in artificial environments relies on strict control of temperature, humidity, and water quality. The long-term viability of the captive colony thus hinges on the ability of facility systems to replicate the original microclimatic condition. This study integrates fifteen years (2011–2025) of demographic monitoring with environmental data collected between May and July 2024 to evaluate the performance of the KCBF colony. Specifically, we assessed (i) long-term trends in population growth, reproduction, and mortality; (ii) shifts in age and sex structure; and (iii) short-term compliance of room environmental conditions with target operational ranges. By linking demographic and environmental indicators, the study provides an evidence-

based assessment of the facility's effectiveness and readiness for sustained reintroduction programs. The findings contribute to broader discussions within Zoo Biology on optimizing captive environments, improving adaptive management, and bridging ex situ–in situ conservation strategies for amphibian recovery

MATERIALS AND METHODS

Study species and facility

The Kihansi Spray Toad (*Nectophrynoides asperginis*) is a live-bearing amphibian endemic to the Kihansi Gorge, Tanzania, that became extinct in the wild after hydro-electric dam construction altered its spray-wetland microclimate. A captive assurance colony was established in 2000 at Bronx and Toledo Zoos (USA) and repatriated to the Kihansi Captive Breeding Facility (KCBF) in 2010 under guidance of the Tanzanian Government. The Kihansi Captive Breeding Facility (KCBF) is located in Kilombero District, Morogoro Region, Tanzania (07°34'S, 36°52'E), approximately 30 km from the original Kihansi Gorge habitat in the Udzungwa Mountains. The present analysis covers KCBF population and environmental records from 2011 to 2025.

Demographic data and analyses

Annual records of births, deaths, releases, and sex/age composition were extracted from KCBF logbooks and electronic databases (2011–2025). Year-end population size (N_t) was calculated as cumulative survivors after each year. Population trend was evaluated using a log-linear regression of $\ln N_t$ against year, following: $\ln N_t = a + b t$

where b is the slope, and intrinsic growth rate $r = e^b - 1$. Model fit was assessed by coefficient of determination (R^2). Summary statistics were generated in Python 3.11 using the pandas 2.2, numpy 1.26, matplotlib 3.9, and scipy 1.13 libraries. Monthly births and deaths were also summarized to calculate annual means and standard errors (SE), see Figure 2.

Environmental monitoring

Room temperature and relative humidity were logged from 1 May to 31 July 2024 in three rearing rooms (R1–R3) using Inkbird IBS-TH2 Plus sensors (± 0.5 °C; ± 2 % RH). Data were recorded every 10 minutes and exported as CSV. Sensors were calibrated monthly against a Vaisala HMP110 reference unit. Target operating ranges were 19–24 °C and 85–95 % RH. The proportion of readings within each target range was computed per room and compared descriptively. For monthly analyses (Figures 4A and 4B), mean $\pm$ standard error (SE) was computed for temperature and humidity across 10-minute interval data within each month and room, while the proportion of readings within target ranges (Figure 5) was summarized with 95 % Wilson confidence intervals.

Statistical and graphical outputs

Descriptive summaries, line plots, and frequency histograms were prepared from the cleaned dataset “KST Population Structure 2 (Recovered).xlsx”. Figures were produced with matplotlib 3.9, exported as vector files (.EPS, .PDF, .SVG) and high-resolution PNG, ensuring consistent typography and accessible color contrast in accordance with Zoo Biology guidelines. For Figures 2, 4A, and 4B, means $\pm$ standard errors (SE) were plotted to represent within-year or within-month variability, and Figure 5 included 95 % Wilson confidence intervals for proportional estimates.

Ethics and permits

All husbandry protocols and monitoring complied with the TAWIRI Animal Care and Use Protocol and the Conduct of Wildlife

Research regulations (2020). Captive management protocol was approved by the research development section of the Directorate of Research, Tanzania Wildlife Research Institute

Data reproducibility

Analytical scripts and processed data summaries will be archived with a DOI upon acceptance to ensure full reproducibility.

RESULTS

Population size and long-term trend

Across the 15-year period, the end-of-year population size ($N_{ten,d}$) at the KCBF increased from 48 individuals in 2011 to a peak of 2,570 in 2019, maintaining stability thereafter (overall mean $\pm$ SD = $1,418.4 \pm 1,044.9$ individuals). A log-linear regression of $\ln(N_{ten,d})$ against year revealed a significant positive trend (slope $b = 0.244$, $R^2 = 0.71$), equivalent to an annual intrinsic growth rate (r) = 27.6 % yr⁻¹ (Fig. 1). Despite short-term fluctuations (notably in 2014–2015 and 2017), population recovery has been strong, indicating progressive demographic stabilization. A two-sample Mann–Whitney U test comparing early (2011–2016) and late (2019–2025) phases showed that mean $N_{ten,d}$ rose significantly from 282.0 ± 156.4 to $2,031.3 \pm 213.2$ ($U = 0.0012$, $p < 0.01$; Cohen’s $d = 2.45$), confirming sustained population recovery.

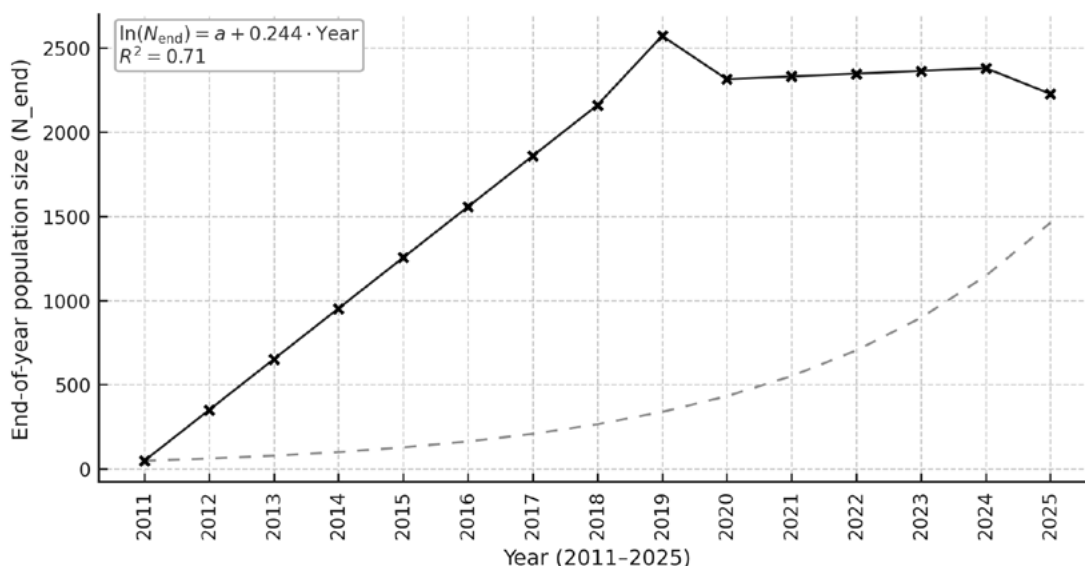


Figure 1. Annual end-of-year population size (N_{end}) of the Kihansi Spray Toad (*Nectophrynoides asperginis*) at KCBF, 2011–2025. Points show year-end totals; line indicates inter-annual trajectory.

Reproductive output and mortality

A total of 10,903 births and 4,273 deaths were recorded during 2011–2025. Mean monthly births and deaths ($\pm$ SE) across years demonstrated high variability but a consistent excess of births after 2015 (Fig. 2a,b). Repeated-measures (by ANOVA) indicated a significant main effect of event type (birth > death; $F_{1,266} = 37.2$, $p < 0.001$) and year ($F_{14,266} = 5.83$, $p < 0.001$), showing temporal variation in reproductive output. Post-hoc comparisons revealed that births exceeded deaths in 10 of 15 years ($p < 0.05$), with an overall median births-

to-deaths ratio of 2.39, signifying positive recruitment and stable demographic turnover. Annual trends in births and deaths (Fig. 2a) showed moderate fluctuations between years, with higher birth frequencies recorded from 2015 onward and minor mortality peaks in 2014 and 2020. The births-to-deaths ratio (Fig. 2b) exceeded 2.0 in most years after 2017, and no deaths were recorded between 2022 and 2024. These results demonstrate consistent reproductive dominance over mortality throughout the later monitoring period.

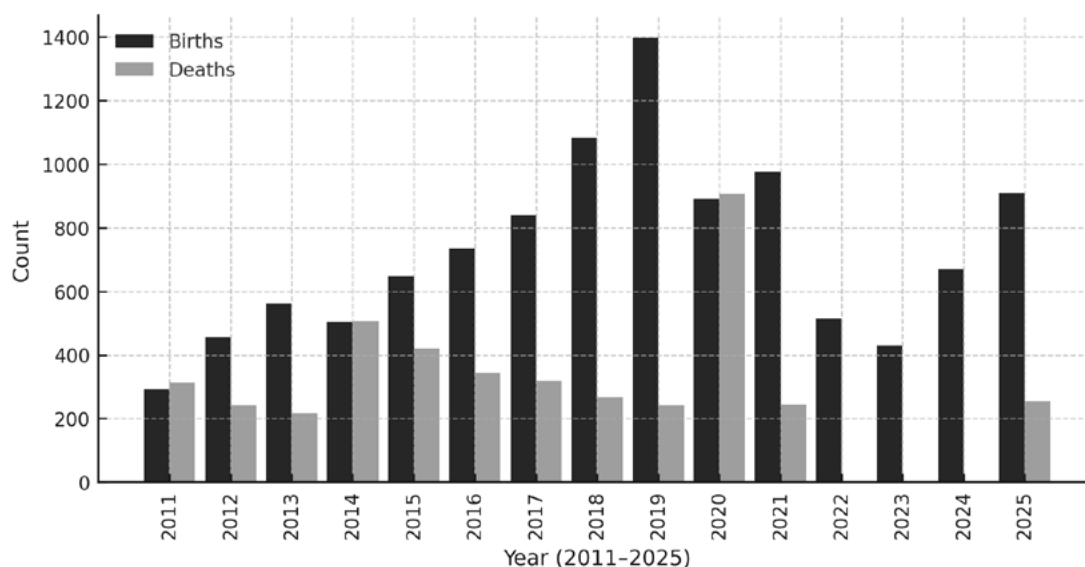


Figure 2a. Annual births and deaths of Kihansi Spray Toad at KCBF (2011–2025).

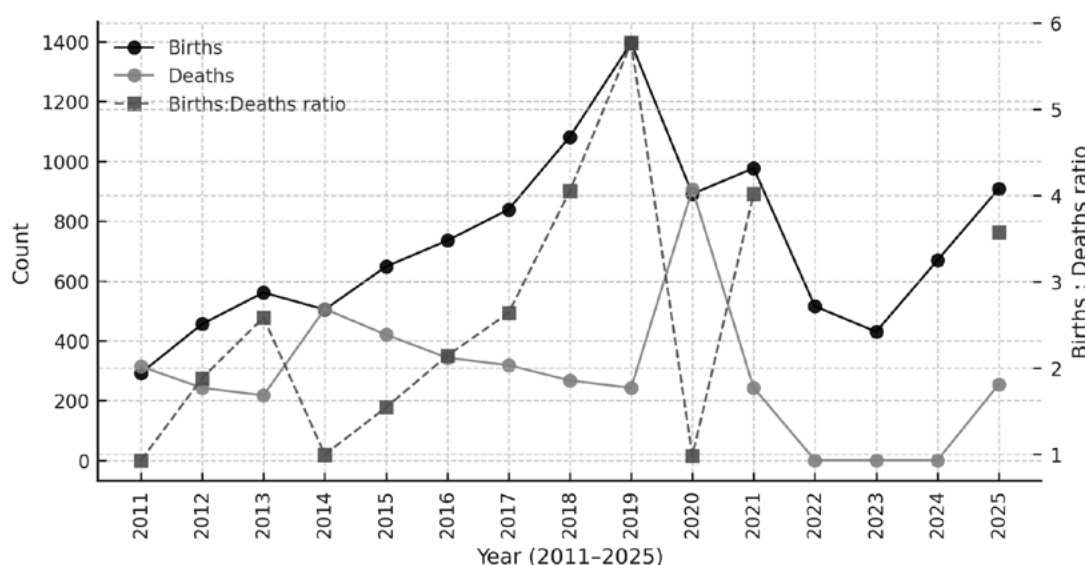


Figure 2b. Annual births, deaths, and births-to-deaths ratio of Kihansi Spray Toad at KCBF (2011–2025).

Releases

A total of 2,649 KST were released into the Kihansi Gorge between 2012 and 2024, following progressive improvements in captive population performance. Annual release numbers fluctuated between 42 and 635 individuals, with the largest cohorts recorded in 2015 and 2021. Releases were temporarily

suspended between 2017 and 2018 to allow stabilization of the captive population and resumed thereafter. Individuals released comprised a mixture of juveniles and adults, ensuring representation across reproductive age classes and promoting the establishment of a self-sustaining wild population (Fig. 3). Survival analysis after release in the wild showed that there was very low survival and

that released individuals in the wild were not seen or disappeared after three to five months. It is not known what was the main cause of poor survival although some monitoring results showed the causes might be inadequate food, predators, fungal infection, poor water quality and poor environmental conditions especially inadequate mist.

Population structure

Age and sex-class composition at the KCBF remained relatively balanced between the early (2011–2016) and late (2019–2025) monitoring

periods (Fig. 3). Adult males accounted for 39.8 % of individuals in 2011–2016 and 34.9 % in 2019–2025, while adult females increased from 30.8 % to 35.1 %. The proportion of gravid females rose slightly from 1.8 % to 2.8 %, and juveniles remained nearly constant at about 27 %. A chi-square test revealed a significant difference in class proportions between periods ($\chi^2 = 10.4$, $df = 3$, $p = 0.016$), reflecting minor but measurable shifts in demographic composition. Overall, the age structure demonstrates continued representation of all reproductive classes, indicating sustained recruitment within the captive colony.

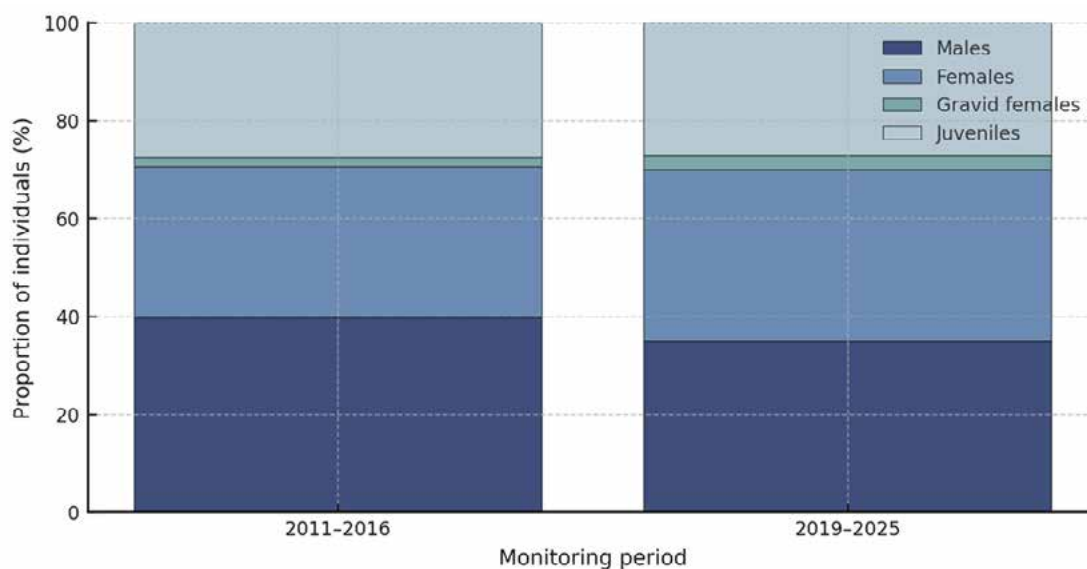


Figure 3. Proportion of KST age classes (males, females, gravid females, juveniles) at the Kihansi Captive Breeding Facility (KCBF) during 2011–2016 and 2019–2025. Bars show percentage composition per monitoring period; legend identifies age classes. Colors are print-compatible and color-blind-safe, following Zoo Biology graphical standards.

Environmental conditions within rearing rooms

Continuous logging from Rooms R01–R03 between May and July 2024 produced > 21,000 records of ambient temperature and relative humidity (RH), representing the core microclimatic parameters monitored to ensure suitable captive conditions for the KST (*Nectophrynoides asperginis*). Mean ($\pm$ SE) monthly temperature and RH are presented in Figures 4A–B, which also depict daily

fluctuations and three-day rolling trends derived from continuous sensor data (see Section 2.3, “Environmental data recording and quality control”).

Mean monthly temperatures remained slightly below the operational target range of 19–24 °C across all rooms, averaging 17.8 ± 1.47 °C (R01), 17.2 ± 1.74 °C (R02), and 16.9 ± 0.59 °C (R03). Daily values ranged between ≈ 19.8 °C and 25.6 °C (Fig. 4A), generally within the broader acceptable window for amphibian

maintenance (18–26 °C). Minor transient declines were observed in late June followed by stabilization toward July, indicating that the automated temperature regulation system effectively minimized excessive variation (daily amplitude < 2 °C).

Mean RH ranged 75.0–78.7 %, also below the desired 85–95 % target, although humidity exceeded 85 % for 26–39 % of logged intervals (Fig. 4B). The daily RH trendlines show brief humidity peaks coinciding with misting or cleaning events, while the monthly mean $\pm$ SD overlay confirms low between-room variability (< 5 %). ANOVA revealed significant room differences in both temperature ($F_{2,9} = 8.41$, $p = 0.009$) and relative humidity ($F_{2,9} = 6.72$, p

$= 0.018$), with R02 maintaining slightly higher humidity and R01 tending to be warmer.

Collectively, these results confirm that environmental parameters within KCBF rearing rooms remained generally stable through the May–July 2024 period, with daily and monthly variation captured by continuous logging and rolling averages. Despite mean values falling marginally below defined targets, both temperature and humidity were maintained within biologically acceptable limits for *N. asperginis*, demonstrating functional environmental control systems and microclimatic stability necessary for sustained captive health and reproduction.

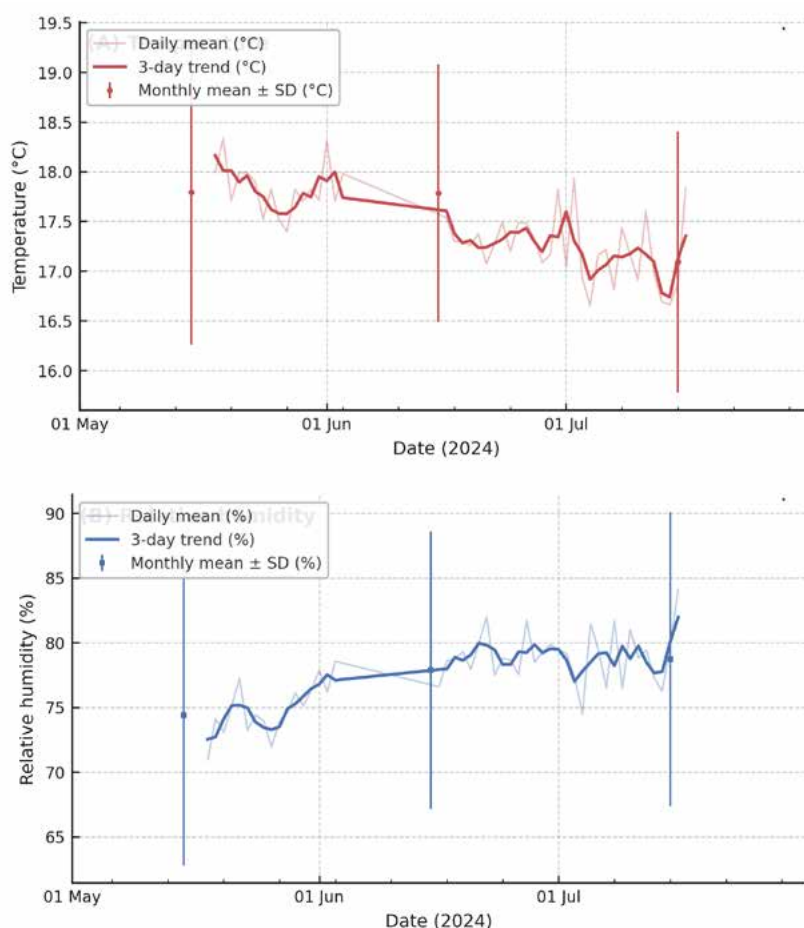


Figure 4 (A–B). Daily and monthly variation in temperature (°C) and relative humidity (%) in rearing rooms R01–R03 at the KCBF, May–July 2024. Panel (A) shows temperature; Panel (B) shows relative humidity. Lines indicate daily means with three-day rolling trends; points and error bars denote monthly mean $\pm$ SD across rooms. Target ranges: temperature = 19–24 °C; humidity = 85–95 %.

Summary of key quantitative results

Overall results confirm robust recovery and stable demography under long-term captive management (Table 4).

Parameter	Result
Study period	2011–2025
N_end range	48 – 2,570
Mean N $\pm$ SD	1,418.4 $\pm$ 1,044.9
Log-linear slope (b)	0.244
Annual growth rate (r)	27.6 % yr ⁻¹
R ² (log-linear model)	0.71
Total births	10,903
Total deaths	4,273
Total released	2,649
Median births-deaths ratio	2.39
Mean N (2011–2016 $\rightarrow$ 2019–2025)	282.0 $\rightarrow$ 2,031.3 (p < 0.01)

DISCUSSION

The Kihansi Captive Breeding Facility (KCBF) population exhibited a strong positive trajectory between 2011 and 2019, increasing from 48 to over 2,500 individuals, with an annual growth rate of 27.6% and a good model fit ($R^2 = 0.71$). This steady increase confirms the long-term viability of *Nectophrynoides asperginis* under controlled conditions and parallels the success achieved in other amphibian assurance colonies such as *Atelopus zeteki* and *Rana muscosa* (Bishop *et al.*, 2012; Mendelson *et al.*, 2019). Sustained population growth after 2015, coupled with reduced mortality, reflects refinement of husbandry practices, particularly improvements in humidity and temperature regulation and the stabilization of feeding and breeding routines under captive breeding conditions.

Births consistently exceeded deaths after 2015, producing a median births-to-deaths ratio of 2.39 and a population structure dominated by females and juveniles. These patterns demonstrate effective reproduction and recruitment comparable to those observed in other direct-developing amphibians maintained

ex situ. Reduced mortality after 2018 coincided with improved environmental monitoring and feeding standardization, suggesting a positive feedback between environmental management and demographic performance. The balanced age and sex structure observed in 2025, with sustained juvenile recruitment, indicates demographic maturity and provides a strong foundation for phased reintroduction planning.

Environmental monitoring between May and July 2024 revealed that mean temperatures and relative humidity remained broadly stable, although slightly below the operational targets of 19–24 °C and 85–95 %, respectively. Room R01 maintained warmer conditions, while R02 showed higher humidity, consistent with small differences in misting coverage and airflow. The modest deviation from target values, coupled with significant differences among rooms, underscores the need for continued calibration and automation of misting and ventilation systems. Maintaining near-natural microclimatic conditions remains critical for amphibian health and reproduction, as fluctuations in temperature and humidity are known to influence skin permeability, hydration, and breeding behavior (Browne *et al.*, 2011; Tapley *et al.*, 2015).

The demographic and environmental stability achieved at KCBF demonstrates readiness for phased reintroduction, provided that spray-wetland microclimates in Kihansi Gorge are restored and consistently monitored. These results emphasize the importance of linking ex situ and in situ conservation in line with IUCN/SSC (2013) guidelines for amphibian reintroductions. Sustaining these achievements will require stable funding for environmental control systems, live-prey culture, and facility maintenance. The integration of automated data logging, real-time alerts, and energy-efficient environmental regulation would further enhance sustainability and husbandry precision.

In summary, the fifteen-year dataset from KCBF provides a rare longitudinal assessment of amphibian recovery in a zoological setting.

The findings confirm that environmental stability and adaptive management underpin demographic resilience in *N. asperginis*. The dataset has also indicated a successive story of captive rearing of amphibians in Tanzania, which can be used to bench mark other captive facility initiatives for threatened amphibians in tropical countries. Continued coordination between captive management, habitat restoration, and hydrological rehabilitation in Kihansi Gorge remains essential to achieving a self-sustaining wild population and ensuring the long-term success of this conservation effort.

ACKNOWLEDGEMENTS

We thank the Tanzania Wildlife Research Institute (TAWIRI), the Ministry of Natural Resources and Tourism (MNRT), and partners from the University of Dar es Salaam and the Wildlife Conservation Society/Bronx Zoo for their continued collaboration. Special appreciation is to the KCBF staff for their dedication and meticulous record-keeping.

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SUB-THEME: INNOVATION AND TECHNOLOGY IN WILDLIFE CONSERVATION

REAL-TIME CLASSIFICATION OF SERENGETI WILDEBEEST BEHAVIOUR WITH EDGE MACHINE LEARNING AND A LONG- RANGE IOT NETWORK

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ABSTRACT

Globally, animal populations are facing increasing levels of environmental disturbance. Human activity, land-use change, and global warming are altering migration routes, space use, activity budgets, and the behaviour of many wildlife species. Understanding impacts on wildlife at a fine scale is essential to identify locations of increased disturbance, mitigate its effects, and predict potential population-level outcomes. In this work, we introduce a low-cost animal tracking system that integrates open-source electronics, edge machine learning, and an Internet of Things network to provide real-time information on the location and behaviour of animals. The system employs an on-board machine learning algorithm to identify distinct behaviours and then transmits classification outputs along with location data over a long-range network. We deployed the system on wildebeest (*Connochaetes taurinus* (Burchell, 1823)) in Serengeti National Park, Tanzania a highly social migratory ungulate population that is ecologically and economically vital to the region. Analysis of the transmitted data showed activity readings were consistent with location data and revealed biologically meaningful fluctuations in daily activity patterns such as reduced walking and grazing at night. Our system introduces a new dimension to studying animal behaviour and movement ecology by offering immediate insights into the behaviour and location of collared animals.

Keywords: *Animal behaviour, conservation technology, machine learning, remote tracking, wildebeest*

INTRODUCTION

Understanding the effects of environmental disturbance on the population dynamics of wildlife species is one of the most significant challenges in ecology. Environmental changes, such as resource degradation, increasing anthropogenic presence, or climate change, can significantly reduce the survival and reproduction rates of populations (Davis *et al.* 2018, Waters and Turner 2022, Turvey and Crees 2019). Direct impacts on mortality

are relatively well studied and measurable (DeCesare *et al.* 2014, Thaxter *et al.* 2017, Rentsch and Packer 2014), however changes to the climate or landscape can induce altered patterns of space use and behaviour in animals (Sih 2013). While non-lethal, these effects can create large physiological and energetic demands that will ultimately lead to increased mortality rates and reduced reproduction (Cooke *et al.* 2014). The challenge of monitoring and mitigating the indirect effects of disturbance on population dynamics is especially acute

for migratory species. Globally, animal migrations are increasingly impeded by human dominated landscapes (Tucker *et al.* 2018), they are challenging to observe, and often the landscapes they inhabit are difficult to manage, since they span regional and national borders. Wildlife species may respond to altered habitats and increasing human activity in various ways. To minimize encounters with humans, wildlife species may infrequently use habitats that are adjacent to human-dominated landscapes (Wittemyer *et al.* 2008, Veldhuis *et al.* 2019, Kavwele *et al.* 2022). Landscapes perceived to be high risk may lead to behavioural responses that involve increased walking or vigilance at the expense of other activities, such as resting or grazing (Ciuti *et al.* 2012). Moreover, altered activity patterns may indicate fear or a flight response, caused by the presence of a direct threat such as poachers or a natural predator (de Knegt *et al.* 2021).

In the context of increasing human activity and climate change, continuous remote monitoring of animals is an essential component of understanding and mitigating the impacts of ecological disturbance on wildlife populations. Remote telemetry devices have had a major impact on the field of animal ecology allowing researchers to track the movements of animals over large spatial scales and long time periods (Cagnacci *et al.* 2010, Hebblewhite and Haydon 2010). For example, using high precision GPS collars it is possible to identify key habitats selected or avoided by wildlife, map corridors used by species such as African elephants (*Loxodonta africana* Blumenbach, 1797) (Graham *et al.* 2009), or identify the kill sites of large predators (Knopff *et al.* 2009). Furthermore, GPS data have been used to identify behavioural states of individuals such as encamped versus exploratory states using measurements of step lengths and turn angles as a function of resources or risks (Hopcraft *et al.* 2014, Morales *et al.* 2004, Roever *et al.* 2010). While the ability to identify behaviours based on position data is a valuable approach, there are several disadvantages associated with the lack of identifiability of certain behaviours and the

role of observation scale in the interpretation of movement data (Plank and Codling 2009, Benhamou 2014). Remote telemetry that offers more detailed insights into an animal's behaviour along its movement trajectory will provide significant advantages over traditional GPS tracking, enabling researchers to assess complex behavioural responses to ecological disturbances, identify locations associated with alterations in patterns of behaviour, and monitor at risk species in near real time.

Advances in telemetry technology, microcontrollers, and machine learning have made it possible to deploy GPS collars with sophisticated on-board sensors for quantitative measurements of the behaviour and physiology of free-roaming animals in situations where direct observation is impractical or logistically infeasible (Brown *et al.* 2013, Tatler *et al.* 2018). Many of these high-resolution biologging devices collect large amounts of data within a short period of time (often recording at 40Hz and reaching several million records) (Ropert-Coudert and Wilson 2005, Wilson *et al.* 2008). While these advances have ushered the study of animal behaviour and movement ecology into the realm of big data (Kays *et al.* 2015, Nathan *et al.* 2022), there are now significant challenges associated with handling, transmitting, and processing the large volumes of data being generated (Williams *et al.* 2020, McCrea *et al.* 2023). Machine learning techniques have proven invaluable in addressing the challenges posed by complex, high-dimensional data given their ability to identify patterns such as distinct behavioural classes from high-frequency time series data (Valletta *et al.* 2017, Tatler *et al.* 2018). Allied with low-power, high-performance microcontrollers, these machine learning models can now be deployed using edge machine learning (Merenda *et al.* 2020) to classify animal behaviour on board telemetry devices (Yu *et al.* 2022, Kazimierski *et al.* 2023), thereby reducing the need for transmitting large volumes of data to a central server or storing on the device for analysis after the device has been recovered.

Acquiring behavioural data in near real time has several advantages. Firstly, alterations in patterns of behaviour can be an indication of heightened levels of perceived threat in the environment. Allocating management effort to regions associated with high risk will potentially improve the efficiency of surveillance or de-snaring operations. For at risk species, such as rhinoceros, alterations in behaviour such as the initiation of rapid movements may indicate the presence of a direct threat. Edge machine learning offers the possibility to detect these behaviour changes in real time as well as enabling on-device algorithms to make use of current behavioural state in their control flow logic, for example increasing fix rates and data transmission when it is inferred that an animal is at risk. For longer term studies, on board classification and transmission removes the need to recapture animals or recover devices after a fixed interval. Data analysis can therefore proceed throughout a telemetry study. Since wildlife trackers equipped with energy harvesting systems have the potential to last 5+ years, being able to collect and analyse data while collars are deployed is essential if efforts to understand the impacts of environmental disturbance are to be relevant for conservation and management policy.

In this paper, we develop a near real-time behaviour classifier using edge machine learning to better understand the fine scale behaviour of migratory wildebeest, *Connochaetes taurinus* (Burchell, 1823), in the Serengeti National Park, Tanzania. The annual migration of wildebeest in the Serengeti ecosystem represents a unique opportunity to understand movement and behaviour of animals over a diverse landscape (Torney *et al.* 2018). The annual migration influences multiple ecological interactions partially because of the sheer abundance of animals, with an estimated population of 1.3 million (Torney *et al.* 2023), and partially because of the extent of their migratory range; on average individuals will travel about 2800km each year (Thirgood *et al.* 2004). The wildebeest consume virtually all

the grass in a given area on a given day, and are forced to move to find fresh grazing which supports a natural rotational grazing that in turn promotes compensatory growth in the grasses (McNaughton *et al.* 1997). The movement of the grazers transports nutrients to other parts of the ecosystem (Subalusky *et al.* 2017). As such, wildebeest are constantly making decisions about grazing locations, departure times from patches, resting spots, and crossing points of features such as river channels or thick vegetation that conceals ambush predators. In addition, wildebeest encounter humans and infrastructure such as tourism pressure (Larsen *et al.* 2020) or land use conversion (Veldhuis *et al.* 2019, Kavwele *et al.* 2022) particularly around the edges of the ecosystem. Considering these threats, little is known about the behaviour and response of migratory wildebeest to anthropogenic pressure in the Serengeti ecosystem at a fine scale.

To better understand the response of the wildebeest to these threats, we combined an open-source, low-cost, telemetry device with a long-range wide-area network (LoRaWAN) to provide information on the location and behaviour of collared individuals. To achieve near real time behavioural monitoring, we leveraged the Tanzania National Park Authority's (TANAPA) LoRa network that has gateway towers strategically placed throughout the Serengeti ecosystem. The ecosystem has 12 gateways serving the Serengeti National Park, a protected area, the Ngorongoro Conservation Area, and adjoining game reserves, covering approximately 25,000km² (Sinclair *et al.* 2007). The distance between the two closest gateways is 8.6km. In the following sections we describe the development of the bespoke activity tracking collar, the collection of training data, and the implementation and training of a machine learning behavioural classifier suited to edge computing. Finally, we present results from a pilot study whereby the collars were deployed on migratory wildebeest in Serengeti National Park in April 2023.

MATERIALS AND METHODS

Collar design

We developed a custom-designed low-cost telemetry device that used a low-power long range wide-area network (LoRaWAN) for connectivity. The device is based on modular open-source components (Foley and Sillero-Zubiri 2020) and comprises a 6-axis inertial measurement unit (IMU) and a GPS tag for recording accelerometry and positional data, respectively, along with a microcontroller board with the capacity to run machine learning models and transmit data via LoRa. These three components were mounted on a custom printed circuit board (PCB) that enabled communication between the devices as well as providing a solar charge controller circuit that provided a consistent power supply to the components and controlled the recharging of the battery. A breakdown of the costs of a single device are shown in table S1. For the microcontroller we employed an Arduino MKR WAN 1310 board, designed with network connectivity using the LoRaWAN protocol. The board is ideal for long-range communication and has the capacity to run machine learning models on board using the TensorFlow Lite for Microcontrollers library (David *et al.* 2021). The use of the MKR family of boards from Arduino also allowed us to switch to a different communication protocol for the collection of training data without having to redesign the collar (see below). For the IMU we employed the Adafruit ICM-20649 wide-range IMU, a 6-axis sensor, which incorporates a 3-axis accelerometer and a 3-axis gyroscope that measures acceleration force and angular velocity respectively. GPS data were provided by the Adafruit Mini GPS PA 1010D module. Printing of the PCB was carried out by European circuits (<https://european-circuits.co.uk/>) and

the components were surface mounted to the PCB by the same company. The collar was powered by a 2000mAh LiPo rechargeable battery and a 0.6W solar panel from Voltaic Systems (<https://voltaicsystems.com>).

A 3D casing, consisting of a bottom and top case, was custom designed in FreeCAD (<https://www.freecad.org/>), an open-source parametric modeler, and then manufactured with a 3D printer using Formlabs' Tough 2000 Resin. We mounted a solar panel on the upper side of the case then connected the PCB solar charge controller to the panel and to a rechargeable LiPo battery. Finally, we sealed the joint between the top and bottom cases using a silicon O-ring cord to ensure that the unit was dust and waterproof. The final design is shown in Fig. 1. The casing was designed to be lightweight but robust, and to be attached to the animal using a custom made biothane collar.

Firmware for the collar was written in C using the Arduino platform and is freely available here <https://www.github.com/ctorney/active-tracker>. The key functionality of the firmware was to classify the IMU sensor data into behaviours of the animal in real time on the device using a pre-trained machine learning model, then subsequently transmit the sequence of classified behaviours to a server via the LoRa network. To train the machine learning model we collected raw accelerometer data using a second design for the collar that replaced the LoRa communication with a Wi-Fi connection. The purpose of this separate design which is described in the next section was to enable the collection of large volumes of data that would not be possible to transmit via a LoRa network.

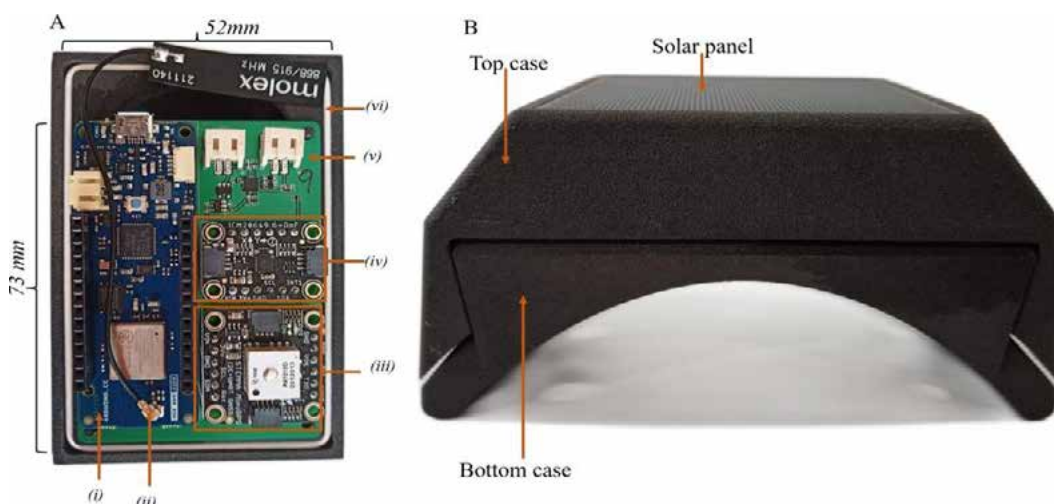


Figure 1: Panel A shows various components used to assemble LoRa collar (i) Arduino MKR WAN 1310, (ii) flexible antenna ISM 868/915MHz, (iii) GPS unit (Adafruit Mini GPS PA1010D breakout board), (iv) 6-axis inertial measurement unit (Adafruit ICM-20649 breakout), (v) solar power charge controller measuring 73mm in length and 53mm in width (vi) silicone sealant cord. The weight of the device is 256g. Panel B shows the solar panel and the top and bottom case of the fully assembled LoRa unit (without the biothane collar for attaching to the animal).

WiFi-enabled collar

The Wi-Fi collar was designed specifically for collecting training data and was able to transmit high-frequency IMU data over a range of up to 1km. We used an Arduino MKR Wi-Fi 1010 microcontroller board in place of the MKR WAN 1310 board while all other components remained the same. Firmware for the collar was developed using the Arduino platform. Since the WiFi communication involves a far greater power consumption than the LoRa protocol the firmware was designed to ensure the collar would remain in a low-power dormant state unless it was broadcasting data. Firmware would therefore periodically create a Wi-Fi network and then wait 5 minutes to see whether a receiver device connected to the network. The collar launched the network three times in a day at 07:00hrs, 12:00hrs, and 16:00hrs local time and at other times remained in its dormant state. If no device connected to the network within 5 minutes, the Wi-Fi network was terminated, and the collar switched to its low-power mode until the next scheduled high-power state. However, if a device connected, the collar would switch to a broadcast state and

the Wi-Fi collar began to transmit timestamped IMU data at 5Hz. IMU data were preprocessed by the microcontroller using a complementary filter (Higgins 1975) and the acceleration in an absolute frame of reference along with the pitch and roll angle of the collar were transmitted. The complementary filter is a data fusion technique that combines the measurements from the accelerometer and the gyroscope to estimate the orientation of the device in physical space. The filter merges the outputs of the two sensors so that the accelerometer is used to estimate the long-term average orientation (using the gravity vector) while the gyroscope provides short-term rotation information. The IMU data was transmitted to the connected device using the telnet protocol, and the device (typically a laptop) logged the data to a text file. In the event that the Wi-Fi connection was lost for more than 5minutes during data transmission, the collar went back to its low-power state until its next scheduled high-power mode.

Training data collection

To collect behavioural data for training our machine learning model, we immobilized and

collared six mature wildebeest at Kilimanjaro Golf and Wildlife Estate in Arusha, Tanzania. Data collection took place from 25th November 2022 to 8th December 2022. The facility is dedicated to the management of captive species, including populations of wildebeest, common eland (*Taurotragus oryx* Pallas, 1766), Grant's gazelle (*Nanger granti* Brooke, 1872), impala (*Aepyceros melampus* Lichtenstein, 1812), and Plains zebra (*Equus quagga* Boddaert, 1785). The facility has no predators so it is expected that the wildebeest population will display reduced levels of vigilance compared to those in the national parks, however the animals freely roam and graze the facility and are expected to otherwise behave similarly to Serengeti wildebeest. We approached the ideal candidates for collaring slowly in a vehicle and immobilized them using a combination of three drugs: Etorphine (3mg-5mg, depending on the animal's body size), Azaperone (50mg), and Medetomidine (3mg). The drugs are administered using a Dan-inject projectile system, which has a 3ml volume and 1-inch- sized dart needle. Within a time frame of 3 to 8 minutes, the darted animal was fully immobilized, we then approached the animal and attached the Wi-Fi collar. To reverse the immobilization effect, Diprenorphine (24mg) was used to reverse both Etorphine and Atipamezole, while Atipamezole (10mg) was used to reverse Medetomidine. The immobilization exercise was led by a veterinary officer approved by the Tanzania Wildlife Research Institute (TAWIRI).

Behavioural observations began the day following the collaring exercise and continued for 14 days. At the start of each day collared individuals were located then once within range we waited for the collar to become active and establish a connection with the receiver device. Once a successful connection was established, we tracked the collared individual by vehicle or on foot while receiving 6-axis IMU sensor data recorded at 5Hz, translating to 5 samples per second. At the same time, we utilized a handheld data logger synchronized with the GPS time of the Wi-Fi collar to record the initiation time of

each of four mutually exclusive behaviours: grazing, lying, standing, and walking. This allowed us to precisely match the switch points of each recorded behaviour. In addition to the IMU data and behaviour transition data, we also filmed the focal individual for potential later verification purposes. Following 14 days of observations we obtained over 40 hours of IMU data and associated behavioural classifications from five collared individuals (we were unable to connect to one collar due to a battery failure).

Behavioural classifier

To classify the IMU data into the four behaviours, we employed a convolutional neural network (CNN) model (LeCun *et al.* 1995) that could be executed on a microcontroller. We employed the CNN approach in order to be able to employ the Tensorflow Lite library, a library that has been specifically optimized for on-device machine learning, however other more traditional machine learning classifiers may perform equally well (for a comparison between deep learning classifiers and more standard machine learning classification with hand-crafted features see Otsuka *et al.* (2024)). Prior to training the CNN model, we performed several pre-processing steps. To begin with, we divided all time series data into 10 second windows and then inspected whether each unique window had 50 time observations. Due to disconnections.

packet loss, and the starting or stopping of the observation process, some windows had fewer than 50 samples and any that had less than 40 observations were removed from the dataset. To ensure all windows had equal number of samples ($n=50$), we filled missing values for windows with fewer than 50 observations but more than 40 using mean imputation. Each window was then a 50×5 array, which was a result of 5 measurements per second for 10 seconds, with 5 channels corresponding to the pitch, roll, and the acceleration in 3 axes relative to the world coordinate system.

To train our classification model, we employed

a small CNN with three convolutional layers and two fully connected layers as shown in Fig. S1A. The final layer of the CNN model was a softmax layer that provided the probability of each of the four behaviour classes. The design of the CNN was in large part dictated by the memory requirements of the neural network and three layers was found to provide reasonable accuracy while maintaining a small memory footprint. We trained the CNN model by minimizing the cross-entropy loss using the Adam optimizer, with a batch size of 256, and trained the model for 300 epochs. To evaluate the performance of the trained CNN model, we split our dataset into training and validation sets with a ratio of 90:10, giving 11,907 windows for training and 1,323 windows for validation. To counter class imbalance in the training data, we employed a weighted loss function that penalized the model more for misclassifying classes that were underrepresented in the training data. We assessed model performance and monitored training by plotting the loss and accuracy curves.

Serengeti deployment

To deploy LoRa collars on wildebeest, we conducted opportunistic searches in areas likely to be frequented by herds. Upon sighting a herd, we approached it slowly by vehicle at a speed of 5 km/h or less to minimise any potential disturbance. Ideal candidates for collaring were identified and immobilised using a combination of three drugs: Etorphine (3mg-5mg, depending on the animal's body size), Azaperone (50mg), and Medetomidine (3mg). These drugs are administered using a Dan-inject projectile system, with a 3ml by 1-inch-sized dart needle. Within 3-8 minutes, the animals were fully immobilised. At this point, we approached them slowly and attached the LoRa collars. Blood samples and tail hair samples were also taken, and the age of the animal was estimated based on tooth wear. To initiate recovery, Diprenorphine (24mg) was used to reverse both Etorphine and Atipamezole, while Atipamezole (10mg) was used to reverse Medetomidine. The animals typically recovered

and rejoined the herd within an average of 7 minutes. Subsequently, we closely monitored the collared individuals to ensure they resumed normal activities. In total, we successfully collared 8 mature wildebeest in an exercise led by a veterinary officer approved by TAWIRI.

Firmware for the LoRa collar was designed to balance the need for high frequency data collection with the need to conserve battery life. The devices were programmed to obtain a location fix every hour and to classify wildebeest behaviours for 30 minutes every hour. The devices therefore operated on an hourly cycle, beginning by switching to a high-power active state whereby the GPS is initiated and the IMU activated and behavioural classifications performed. Once an accurate location fix is obtained the GPS device is turned off to conserve battery. The behavioural classifications continue until the end of the 30 minute period at which point the device switches to a transmission mode. At this point the device attempts to contact the LoRa network by broadcasting a message to any gateways that are within range. If a gateway responds the device will transmit the location and behaviour data to the network beginning with any previously stored readings. If no gateway responds the device stores the current data for later transmission. After a maximum of 20 minutes in transmission mode, or until no gateway acknowledges receipt of transmitted data, the device enters a low-power dormant state until the start of the next hour. The device is therefore in the active high-power state for at least 30 minutes each hour. If the animal is not in range of a LoRa gateway then it will enter the low-power state shortly after the 30 minute behavioural classification period has ended. If the device is in range of a gateway it will spend up to 50 minutes each hour recording behavioural data (30 min) and then transmitting to the network (up to 20 min).

Since the LoRa protocol is designed for long-range, low volume communication, for transmission purposes, data corresponding to each behaviour was encoded as two binary

digits. Each behaviour was encoded in binary as follows: grazing: 00, lying: 01, standing: 10, and walking: 11. Four sequential behaviours corresponding to 40 seconds of IMU data were encoded as 8 bits, or 1 byte, and within 30 minutes, there were 180 behaviours amounting to 360 bits of data, totaling 45 bytes. Location data was transmitted as a separate message and included the GPS coordinates and a timestamp. Along with the timestamp associated with the measurement interval, 30 minutes of behavioural data could be encoded in a single packet of less than 51 bytes. Keeping the amount of data to be transmitted to a minimum, and below 51 bytes, allowed us to use the lowest data rate (DR0) for the LoRa protocol, which has the longest range and lowest power consumption, as well as enabling us to store over 2 years of behavioural data on the device should the collar be out of range of the network for extended periods.

RESULTS

CNN model training accuracy

The validation classification accuracy of the four wildebeest behaviours are as follows. The standing behaviour class had the lowest

accuracy of 79.93%, with approximately 13% and 5% misclassified as lying and walking, respectively. Confusion between standing and lying is likely due to the similarity in the IMU data in these two behaviours, however the confusion between standing and walking is most likely due to the frequent transitions between these two behaviours. Within a 10 second window, wildebeest frequently transition between standing and walking, and the label for the sample was determined by the most common behaviour within the window. The walking behaviour class had an accuracy of 82.86% with approximately 6% and 9% misclassified as grazing and standing respectively. The lying class had 89.22% with about 9% misclassified as standing while grazing achieved the highest classification accuracy of 94.08%, with about 5% misclassified as walking behaviour class. The confusion matrix for the four behaviour classes, displaying true positive and false positive classifications, is presented in Fig. 2. Overall, the CNN model achieved an accuracy of 84% on the validation data.



Figure 2: Confusion matrix for wildebeest (*Connochaetes taurinus*) behaviour classification showing true and predicted behaviour categories for test data.

Deployment and behavioural data analysis

Once trained, the CNN model was installed on the LoRa enabled devices ready to be deployed on wildebeest in the Serengeti National Park, Tanzania. Collars were deployed on eight mature wildebeest on the 16th April 2023 when the wildebeest were in the southern region of their migratory range and located on the nutrient rich short grass plains they occupy during the wet season. The LoRa-enabled devices operated from 16th April 2023, through 15th April 2024. However, three devices ceased to transmit data within a month of deployment, a further 4 devices ceased to transmit after 6 months. The final collar last transmitted data on the 15th April 2024, exactly one year after deployment. Improving the robustness and reliability of the devices is now the primary focus of current efforts.

To evaluate the performance of the LoRa collar we analysed data from two individuals that were consistently within range of a gateway and broadcasting data for over 30 days combined. The minutes of behavioural data analysed for individuals one and two are 3,810 and 19,350, respectively, totalling 23,160 minutes or 386 hours. As an initial sanity check on the output of the classifier we first compared the predicted behaviours with the distance covered by an animal in one hour. Specifically, we examined the proportion of time the classifier predicted the animal was walking and then compared this to the distance travelled in the same interval. The distance walked by a collared individual within one hour ranged from 0.49m (min) to 4225.99m (max), with a mean of 337m. As we would expect there is a positive correlation between fraction of time walking

and distance, even though the behaviours were only predicted for 30 minutes of each hour. Additionally, we examined the proportions of activities of an individual wildebeest over a 24-hour period. We next examined the frequency of predicted behaviours as a function of time of day to reveal the daily activity patterns of the wildebeest at this point in their migration. We employed a non-parametric Gaussian process model (Rasmussen and Williams 2006) to model the latent probability of each behaviour as a function of time of day (see supplementary material for details published version).

The results of this analysis are presented in Fig. 3 and show that the probability of a given activity is strongly influenced by the time of day. The observed patterns have biological relevance, with the probability of grazing being highest in the early morning and late afternoon, just around dawn and dusk respectively, and the probability of walking being highest during the day. The probability of lying and standing was highest during the night, with a slight peak around mid-day.

As such, the probability of a given activity is strongly influenced by the time of day. The observed patterns have biological significance; for instance, increased immobility during the night may represent a strategy to minimize encounters with predators. This activity budget extracted from the data is intended to demonstrate how animals allocate their time to various activities in typical scenarios, along with other influencing factors. It also serves as a key performance indicator for assessing the effectiveness of the LoRa collar as a tool for fine-scale animal tracking.

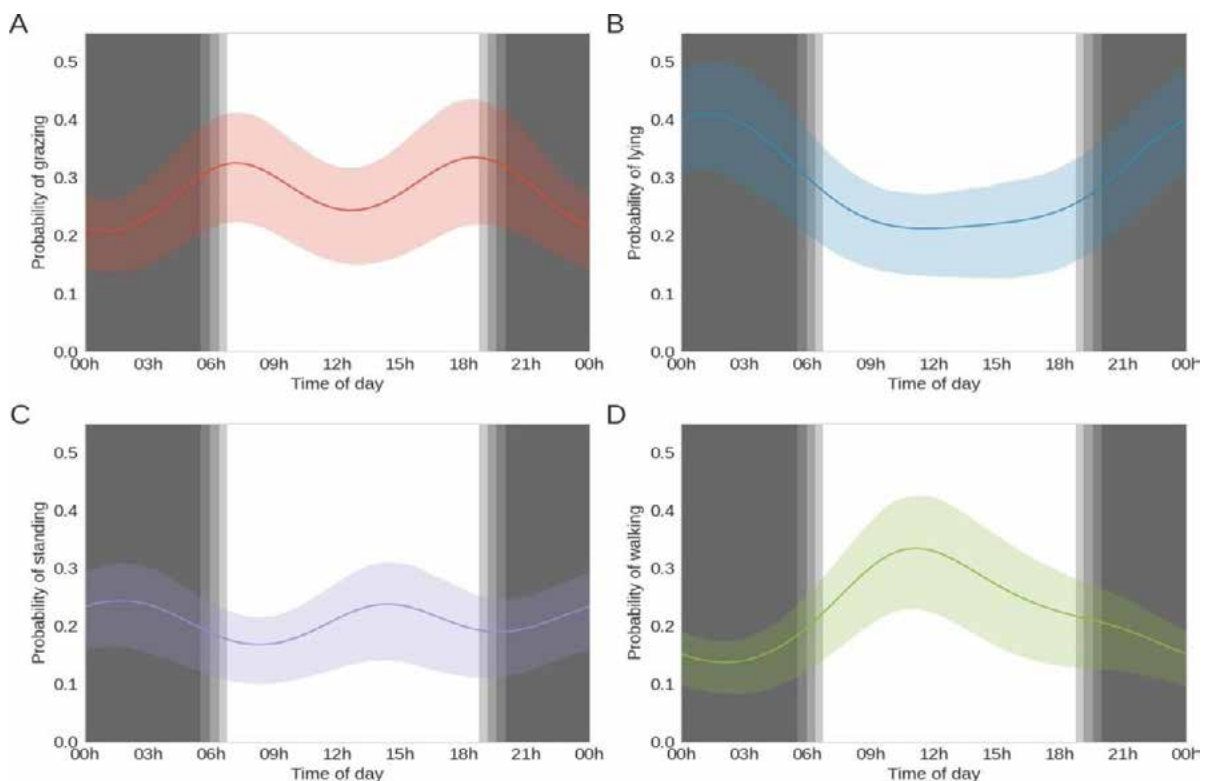


Figure 3: Probabilities of wildebeest (*Connochaetes taurinus*) activities as a function of local time of day. (A) Grazing,

(B) Lying, (C) Standing, (D) Walking. Shaded regions indicate the 50% credible interval while the grey shading indicates civil, nautical, astronomical twilight and night.

DISCUSSION

In this work we have presented a low-cost animal tracking system capable of near real-time behaviour classification using open-source electronics, edge machine learning, and an IoT network. We developed a CNN-based model specifically designed to be deployed on a microcontroller and demonstrated its ability to reliably distinguish between four distinct behaviours of wildebeest with a high degree of accuracy on our validation data set. Furthermore, field tests conducted on Serengeti wildebeest have confirmed that the LoRa collar can successfully record, pre-process IMU sensor data, predict behaviour classes, and transmit this information, along with auxiliary data such as positional and date-time data,

to a server via a LoRa network. These data can then be easily accessed by end-users for download in near real-time. Key performance indicators illustrate the strengths of the LoRa collar as a tool suitable for remote tracking of animals as they navigate heterogeneous habitats and encounter varying levels of threats. Consequently, our framework will be potentially invaluable in assisting researchers to gain new insights into how animals perceive their environment in relation to anthropogenic disturbance gradients.

Being able to obtain fine scale activity data is an essential component of relating activity budgets to perceived risk in the environment. For example, an area where an animal rest may be perceived as a safe zone, whereas areas where an animal moves quickly and ignores resources may be being perceived as high risk. Inferring the relationship between spatial locations and behavioural effects will require the collection of high-resolution data, which

the LoRa collar is capable of providing. These data may be coupled with landscape layers relating to Normalised Difference Vegetation Index (NDVI), grass nitrogen levels, distance to drainage beds, and satellite imagery relating to human-made infrastructure such as the Open Buildings dataset (Sirko *et al.* 2021). Similarly, understanding the behaviour of wild populations in relation to daylight length, meteorological patterns, and seasonal variations such as winter versus summer activities in the context of a changing environment is crucial. The behavioural data that is made accessible with the edge machine learning approach provides a valuable opportunity to investigate how rapid weather changes, including phenomena like snowfall, precipitation, temperature, humidity, among others, impact the activity budget of rare or endangered species and the mapping of their critical habitats.

Previous approaches to inferring behavioural states from remote telemetry data have frequently been limited to the analysis of GPS data. Encamped or exploratory states (Michelot *et al.* 2016) inferred from directional persistence or step lengths (Klappstein *et al.* 2023) have been used to link animal behaviour with observed movement trajectories in a variety of species, including wildebeest (Hopcraft *et al.* 2014), neotropical forest deer (Grotta-Neto and Duarte 2019), and elephants (Vogel *et al.* 2020). However, relying solely on relatively infrequent GPS locations can fail to capture fine scale behavioural transitions that occur over time intervals shorter than the time between fixes, or can lead to the misidentification of behaviours due to the scale of observation, for example, a highly tortuous path may be interpreted as resting since the displacement between fixes is low.

Beyond the importance of high temporal resolution data, on-board machine learning models make it possible to transmit classifications of animal behaviour in near real-time. By monitoring positional and behavioural data over time, researchers can establish baseline behaviours for different species

and consequently generate a comprehensive ecosystem monitoring tool. For instance, if an animal's behaviour deviates from the norm at a given time, in a given habitat and season, this can act as a real-time indicator of ecological disturbance or intrusion by poachers especially if an entire ecosystem has LoRa network coverage (de Knecht *et al.* 2021). Given that ongoing human-induced environmental changes exceed the evolutionary response rate of many species or populations, an in-depth understanding of their responses to such changes is vital (Sih *et al.* 2011, Snell-Rood 2013). In these circumstances, the framework we present here can be a highly valuable tool for collecting data at a scale commensurate with the rate of occurrence of ecological disturbances, offering insights into how species adjust their nocturnal or diurnal activities as a coexistence mechanism, as well as assessing their overall behavioural plasticity in response to human-induced disturbances. Here, it is critical to get real-time information rather than waiting until a collar is retrieved.

Behavioural data is also crucial for livestock monitoring (Arablouei *et al.* 2021, Riaboff *et al.* 2022) and for post-release monitoring of reintroduced species (Sarmiento *et al.* 2019, Mertes *et al.* 2019). Incorporating IMU devices and edge machine learning models into traditional GPS collars can provide benefits such as offering a more comprehensive understanding of the behaviour of reintroduced species in their new environment, a system for monitoring free-ranging livestock, and a potential early warning system for detecting when animal behaviour deviates from the norm, for example prolonged periods of lying or resting could signal the possibility of a sick or injured animal, prompting a timely response from conservationists or livestock managers. A further advantage of the edge machine learning approach is that it may be considered as a highly optimised form of data compression, retaining only information that is relevant for the specific application. Our algorithm compresses 10 seconds of 5Hz data which is equal to 1000 bytes, or 8000 bits, into 2 bits. When scaling

to large numbers of individuals this may lead to significant cost savings for data storage and transmission.

A key advantage of our framework is the use of programmable open-source electronics which ensures the system is flexible, customisable, and versatile. This feature means that the user has the option of reconfiguring the firmware or incorporating additional modules to collect other types of data, such as temperature, humidity, or audio, to enhance the quality and quantity of data collected. For example, the device could be modified to sample more during diurnal, nocturnal, or crepuscular hours for species known to be active at such times (Vazquez *et al.* 2019). In terms of the physical dimensions, the 3D housing can be modified to fit a range of species as long as water-tightness and robustness are maintained.

While our framework has the potential to open up new research avenues, there are a number of challenges involved in fully utilising it. Although the LoRa protocol has a low power consumption, frequent transmission of data can limit the battery life of the device, especially in scenarios where the LoRa coverage is sparse. To overcome this obstacle, we deployed a solar charging system to ensure the device remains powered for extended periods, however using it to study animal species that spend a significant amount of time in low-light conditions may severely limit its longevity. For the LoRa-enabled device, the measured sleep current was 1.7mA, while running the IMU processing and classification consumed around 18mA, with GPS acquisition increasing this to 50mA. We observed current consumption of around 70mA during data transmission. Assuming an average time of 2 minutes to obtain a GPS fix and 5 minutes of data transmission, the device will run for 5 days without charge, meaning that any energy harvesting system would need to be able to fully recharge a 2000mAh LiPo battery over the course of this time frame. Further development of the tool could explore possibilities of harvesting green energy by using vibration energy harvesters that convert

the kinetic energy of the collared animal into electrical energy (Zhang *et al.* 2021). As previously described, the devices deployed had a high failure rate and current efforts are focused on improving their robustness and field durability. There are several potential causes for the high failure rate that we are seeking to address. Firstly, the collars had an in-built breakaway mechanism that ensures the device will eventually detach itself from the animal. Tuning this mechanism so that it does not detach prematurely may well improve the longevity of the collars. Secondly, the LiPo battery used may be a cause of failures and we intend to use a higher quality, more expensive battery in future. Firmware issues may also lead to devices ceasing to transmit data. Further hardware and firmware testing will be performed before the next phase of deployment.

Customising the device for different species will require new training data and this will likely be a significant challenge. In our case, the wildebeest classifier was trained on data collected from a separate population of wildebeest and then deployed on the Serengeti population. While this enabled us to collect data at a scale that would be impossible for the migratory population given the spatial extent of the Serengeti and the density of the wildebeest herds, this does raise the issue of generalisation between populations. While behaviours such as walking and grazing are expected to be consistent across the two populations, it seems likely that the frequency of vigilance behaviours may differ between the two populations due to the lack of predators in the Kilimanjaro Golf and Wildlife Estate. The lack of vigilance behaviour (typically characterised by a raised head) in the training data set means the accuracy of the classifier in distinguishing between standing and lying behaviours may well be overestimated, given that the head position appears to be the distinguishing feature between these two behaviours.

Notably, setting up and maintaining a LoRa network in protected areas is costly and logistically challenging. Depending on the

terrain, real world transmission range can be over 70km, however even at this range the number of gateways required to fully cover a large area and enable full real-time monitoring can be substantial. In situations where real-time data transmission is not required portable gateways may be used, or gateways can be located in areas where the animal is likely to pass and data can be stored on the device until it is in range. In all scenarios, collaboration between organisations and the sharing of gateways is likely to be necessary to fully utilise the potential of a LoRa network and there will always be a trade-off between the cost of network coverage and the delay in receiving data; fully real-time monitoring is possibly but requires complete network coverage.

Our framework will enable researchers to overcome restrictions imposed by accessibility issues and observer influence, as well as eliminate the need to recapture animals for data download. The use of open-source components paves the way for others to enhance the design or customise the device to address specific needs. As such, our framework will advance our capacity for fine-scale species monitoring and enhance our understanding of animal behaviour and responses to ecological disturbances, particularly in scenarios requiring rapid and adaptive management policies. In the context of the Serengeti, the methods we propose represent a significant new tool in understanding and protecting the iconic wildebeest migration.

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ESTABLISHING ENVIRONMENTAL AND SOCIAL BASELINES FOR AI-DRIVEN CLIMATE ADAPTATION AND HUMAN-WILDLIFE CONFLICT MITIGATION IN THE RUAHA-RUNGWA ECOSYSTEM, TANZANIA

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ABSTRACT

Climate change and increasing human–wildlife conflicts (HWC) continue to threaten rural livelihoods and conservation efforts within the Ruaha-Rungwa ecosystem in southern Tanzania. Communities in this area depend heavily on rain-fed agriculture and livestock, making them vulnerable to recurring droughts, shifting rainfall patterns, and wildlife incursions from elephants and carnivores. Artificial Intelligence (AI) presents promising opportunities to enhance resilience through improved wildlife monitoring, early warning systems, and better access to markets. However, the effectiveness of such digital interventions requires a clear understanding of local environmental and social conditions, including community perceptions, vulnerability levels, and technological readiness. This study reports results from an Environmental and Social Impact Assessment (ESIA) baseline survey conducted in March 2025 across selected villages bordering protected areas. A mixed-methods approach involved a household survey of 365 randomly selected respondents and qualitative interviews with local leaders, wildlife scouts, and agricultural officers. Data were collected on climate risks, livelihood dependence, HWC experiences, and awareness and acceptance of AI technologies. The results indicate widespread exposure to climate variability and frequent wildlife encounters that heighten food insecurity. Although prior knowledge of AI is limited, many respondents expressed willingness to adopt AI-supported solutions, constrained by low digital literacy and limited smartphone access. Such insights will inform the inclusive and context-specific implementation of the AI4-Climate Change Tanzania project.

Keywords: *Artificial intelligence, climate adaptation, human-wildlife conflict, Ruaha-Rungwa, Tanzania*

INTRODUCTION

Climate change has emerged as one of the greatest threats to rural livelihoods and biodiversity conservation in sub-Saharan Africa, where dependence on climate-sensitive resources is high and adaptive capacity remains limited (IPCC, 2022; Supuku, 2024). In Tanzania, the Ruaha–Rungwa ecosystem stands out as a critical socio-ecological landscape, hosting significant populations of elephants, large carnivores, and diverse savannah wildlife while simultaneously supporting thousands of rural households reliant on rain-fed agriculture and livestock keeping (Kiffner

et al., 2020). These dual ecological and livelihood functions make the region a priority zone for understanding and addressing the compounding pressures of climate change and human–wildlife conflict (HWC). As rainfall patterns become increasingly erratic and drought frequency intensifies, communities in protected area buffer zones face reduced food production, declining water availability, and heightened vulnerability to climate-related shocks (URT, 2021; Shemweta & Kideghesho, 2023). In parallel, habitat fragmentation and scarcity of natural resources push wildlife beyond protected boundaries into farmlands and grazing zones, escalating crop raiding,

livestock predation, and property damage (Hariohay, 2019; Kidane *et al.*, 2024). These intertwined stressors amplify livelihood insecurity and erode social tolerance toward wildlife conservation, generating governance and sustainability challenges (Musalo *et al.*, 2025).

Innovative approaches are therefore urgently needed to strengthen coexistence and resilience in such human–wildlife interface landscapes. Advances in artificial intelligence (AI) and emerging digital tools offer transformative opportunities for enhancing conservation surveillance, predicting wildlife movement, and supporting informed decision-making in climate-sensitive food systems (Fergus *et al.*, 2024; Bjorck *et al.*, 2019). AI-driven systems — including automated species identification, threat detection, and agricultural advisory services — have shown strong potential across African conservation and agricultural programmes (Adom, 2024; Aliyu *et al.*, 2025). Their integration into community adaptation pathways could improve early-warning systems, strengthen livelihood planning, and reduce harmful encounters with wildlife.

However, the successful deployment of AI in rural conservation settings requires a comprehensive understanding of baseline environmental and social conditions, community needs, digital access levels, and institutional readiness (OECD, 2019; IFC, 2015). The effectiveness, acceptance, and sustainability of technology-enabled solutions depend not only on their technical accuracy, but also on their alignment with local priorities, equity considerations, and ethical constraints in wildlife and data management (Vaughan *et al.*, 2022). This article establishes baseline insights into the environmental stressors, HWC dynamics, and socio-economic vulnerabilities within the Ruaha–Rungwa ecosystem, and assesses the potential role of AI technologies in strengthening climate-resilient human–

wildlife coexistence. The findings contribute to growing regional and global discourse on technology-supported adaptation in conservation landscapes where climate threats and biodiversity values converge.

MATERIAL AND METHODS

Study Area

This study was conducted in villages located within the buffer zones of the Ruaha–Rungwa ecosystem in southern Tanzania (Fig. 1). The area forms one of the largest connected wildlife conservation landscapes in East Africa, covering over 45,000 km² and comprising Ruaha National Park, Rungwa Game Reserve, and adjacent wildlife and forest management areas that maintain essential migratory routes and predator–prey dynamics. The landscape is dominated by Miombo woodlands and savannah mosaics, with riparian corridors and seasonal wetlands supported by the Great Ruaha River and its tributaries, which are critical for wildlife dispersal and for sustaining downstream communities. The climate is semi-arid with unimodal and highly variable rainfall occurring from November to April, followed by prolonged dry seasons. Recent climatic shifts—including delayed rainfall onset and increased heat stress—have affected vegetation condition, water availability, and crop productivity. Settlements are concentrated around village centers but extend into agricultural and grazing zones near protected area boundaries. Livelihoods rely largely on rain-fed farming (maize, legumes, sunflower) and livestock keeping, making households vulnerable to climate shocks. Continued expansion of farmland into wildlife corridors has driven habitat fragmentation and increased encounters with elephants and carnivores, leading to frequent crop damage, livestock predation, and escalating human–wildlife conflict.

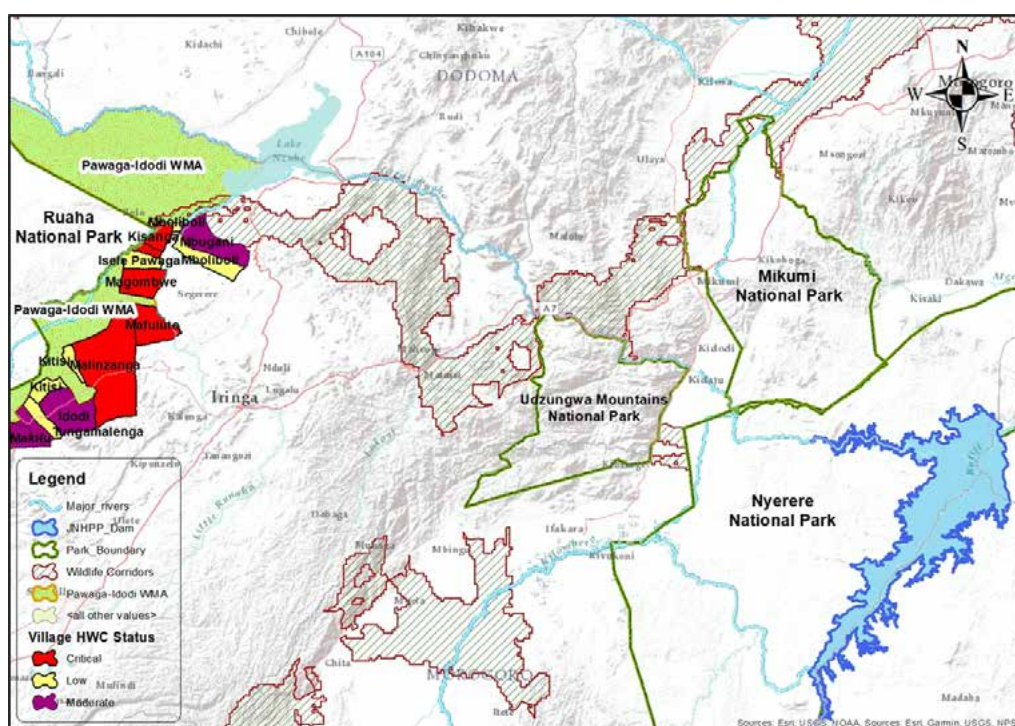


Figure 1: Map of the Project Area

Study Design and Data Collection

A mixed-methods research design was used to capture the complex interactions between climate stressors, human–wildlife conflict (HWC), and socio-economic vulnerability in communities bordering the Ruaha–Rungwa ecosystem. This design integrates quantitative evidence on exposure and risk with qualitative insights into perceptions, coping strategies, and institutional capacity (Kidane *et al.*, 2024; Musalo *et al.*, 2025). The approach consisted of four complementary data collection techniques

Household

Structured questionnaires assessed livelihood dependence, climate impact exposure, wildlife conflict frequency, and adaptation practices. This method allows quantification of vulnerability patterns across households (Supuku, 2024; Shemweta & Kideghesho, 2023).

Surveys

socio-cultural influences on risk tolerance, and community-driven solutions (Kidane *et al.*, 2024).

Key Informant Interviews (KIIs)

KIIs with village leaders, extension officers, and wildlife authorities examined institutional roles, policy gaps, and response mechanisms (OECD, 2019).

Spatial & Environmental Assessment

GIS and remote sensing analysis mapped land-use change, wildlife corridors, and settlement expansion to determine environmental drivers of HWC (Fergus *et al.*, 2024; Bjorck *et al.*, 2019).

Qualitative data were audio-recorded and transcribed, while survey responses were coded and analyzed statistically to triangulate insights across methods.

Focus Group Discussions (FGDs)

FGDs with farmers, pastoralists, and women’s representatives explored shared concerns,

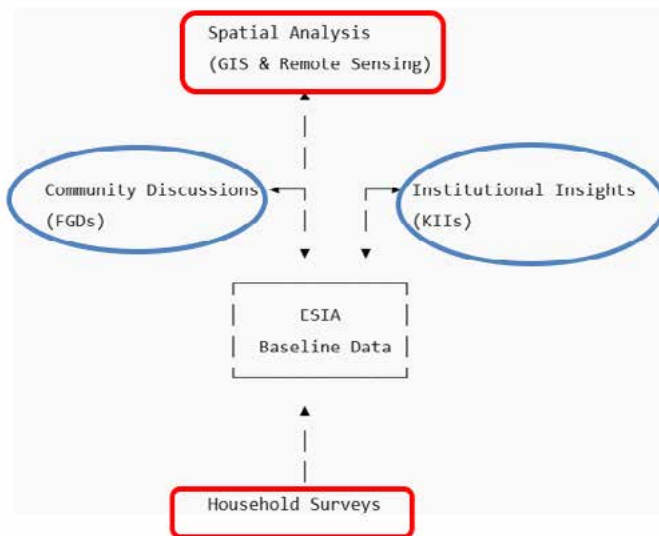


Figure 2: Mixed-Methods ESIA Research Design

Data Analysis

The analysis integrated quantitative, qualitative, and spatial datasets to generate a comprehensive understanding of environmental

pressures, human–wildlife conflict dynamics, and community vulnerability. Each dataset was processed using an approach appropriate to its structure and analytical purpose, as summarized in Table 1.

Table 1: Summary of Data Types and Analysis Approaches

Data Type	Analysis Approach	Supporting Reference
Quantitative household data	Descriptive statistics (frequencies, percentages) to determine prevalence of livelihood vulnerability and human–wildlife conflict trends	Supuku (2024)
Qualitative FGD and KII data	Thematic analysis to identify community perceptions, coping practices, and institutional challenges	Musalo <i>et al.</i> (2025)
Spatial environmental data	GIS-based land-use change detection and proximity mapping to assess environmental pressures and habitat interface	Fergus <i>et al.</i> (2024)

RESULTS

Demographic Characteristics

A total of 392 respondents participated in the household survey conducted across selected villages in the Ruaha–Rungwa ecosystem. The surveyed population is predominantly young, with 60–66.4% aged between 18 and 35 years, suggesting a labor force that is highly dependent

on physically demanding and climate-sensitive livelihood activities such as farming and livestock keeping. Most households are located within approximately 60 km of protected areas, placing community members in zones of direct exposure to wildlife incursions and associated risks.

The sex distribution of the sample shows a male dominance, with 63.8% male and 36.2%

female respondents. Educational attainment remains low, with primary education being the most common level achieved (69.6%), while only 5.6% of respondents reported having completed tertiary education. Household size patterns indicate relatively small family units, as the majority (58.4%) reported between 1–4 household members. These demographic characteristics collectively highlight a youthful but vulnerable rural population with limited formal education, high dependence on natural resources, and significant exposure to human–wildlife conflict pressures in the study area.

Community Vulnerability, Risk Exposure, and Pathways for AI-Enabled Climate and Human–Wildlife Resilience

This section presents a comprehensive analysis of the socio-economic conditions, climate risks, human–wildlife interactions, digital readiness, and institutional capacities that collectively shape community vulnerability and resilience pathways within the Ruaha–Rungwa ecosystem.

Livelihoods and Exposure to Human–Wildlife Conflict (HWC)
Agriculture remains the dominant livelihood source across surveyed households, often complemented by livestock keeping, artisanal activities, and small-scale trade to diversify income streams. Most families cultivate between 1–20 hectares of land, reflecting typical rural smallholder farming systems in Tanzania. However, their proximity to protected areas heightens vulnerability to HWC. Frequent crop raiding by elephants, especially during crop-maturing seasons, and livestock predation by lions, hyenas, and other carnivores were consistently reported as major shocks to household economies. These impacts disrupt food production cycles, increase financial burdens, and contribute to seasonal food insecurity, particularly during prolonged droughts when both wildlife and communities compete for diminishing natural resources.

Digital Access and AI Readiness

Smartphone ownership remains limited, with only about 35% of surveyed households reporting access to internet-enabled devices, and network coverage being inconsistent across villages. Despite these constraints, communities demonstrated a strong willingness to adopt digital solutions that can improve livelihood security. Respondents highlighted a clear interest in AI-driven early warning systems for wildlife movement (72%), climate hazard alerts (69%), market price updates (64%), and localised seasonal forecasts (58%)—all seen as practical tools for reducing vulnerability and enhancing productivity. However, readiness gaps persist. Key challenges include limited digital literacy (54%), high data and device costs (47%), and concerns regarding data privacy and trust in automated systems (32%). These barriers emphasize the need for inclusive digital capacity-building programmes, investments in rural connectivity and charging infrastructure, and community-centered governance models to ensure equitable and sustainable adoption of AI technologies in climate-impacted landscapes.

Market Connectivity,

Market participation remains a major constraint in the surveyed communities, with **over 90%** of respondents reporting difficulties accessing reliable and profitable markets for their agricultural and livestock products. The most frequently cited barriers included **poor and seasonally impassable roads (78%)**, **high transportation costs (65%)**, and **weak value chain coordination (57%)**, which collectively force smallholder farmers to depend on intermediaries who typically offer lower prices. Moreover, **71%** of households indicated that the **absence of real-time information** on market trends, demand levels, and input availability negatively affects their bargaining power and strategic decision-making around when and where to sell. These structural limitations reduce household income and amplify livelihood risks, particularly during periods of **climate stress and human–wildlife conflict**,

when production and mobility are further disrupted. Notably, a majority of respondents (67%) showed strong interest in **AI-enabled market platforms** capable of providing timely price information, facilitating direct linkages with buyers, and improving aggregation efficiency. Such digital innovations present a strong opportunity to **increase transparency**, reduce transaction losses, and strengthen rural market systems as an integral component of resilience-building strategies.

Climate Risk and Vulnerability Exposure

Communities reported high exposure to climate-driven shocks, particularly **droughts (82%)**, **unpredictable rainfall patterns (74%)**, and **extreme heat events (61%)**, all of which directly threaten agricultural yields and water availability. These climate stresses compound existing livelihood vulnerabilities, where over **68%** of households depend primarily on rain-fed agriculture, resulting in increased production losses, food insecurity, and pressure to expand cultivation into wildlife habitats. Moreover, **59%** of respondents identified climate change as a factor intensifying **human–wildlife conflict**, with wildlife increasingly moving into farms and settlements in search of water and forage. Limited access to timely weather information (**66%**) and low use of climate-smart farming practices (**less than 40%**) heighten households' sensitivity to climate variability and reduce their adaptive capacity. These findings underscore the urgency for **localized climate advisories**, integrated **AI-driven hazard monitoring**, and the promotion of **resilient land-use practices** that safeguard both community livelihoods and ecological stability. Without such interventions, climate impacts will continue to erode household resilience and escalate coexistence challenges within the Ruaha–Rungwa ecosystem.

Social Structures and Institutional Capacities

Local governance structures, including Village Councils, natural resource committees, and WMA leadership, play a central role in mediating resource use, resolving conflicts, and coordinating community-based conservation. A majority of respondents (73%) recognized these bodies as primary institutions for decision-making and reporting wildlife incidents. However, capacity constraints remain evident. Over 62% of participants indicated that response to human–wildlife conflict (HWC) incidents is often delayed or inadequate, largely due to limited financial and logistical resources. Community awareness programmes on climate risks and environmental management are variably implemented, with only 41% reporting participation in any form of structured sensitization activity during the previous year. Additionally, traditional knowledge networks continue to inform land-use and risk coping strategies, particularly among elders and livestock keepers. Yet, institutional linkages between these local systems and external actors remain fragmented. Collaboration with key agencies—including TANAPA, wildlife rangers, extension officers, and district authorities—was reported as inconsistent by 57% of respondents.

Despite these shortcomings, communities expressed strong willingness (79%) to engage in joint planning approaches that integrate conservation efforts with livelihood priorities. Strengthening institutional partnerships, enhancing conflict-response mechanisms, and incorporating community voices in governance processes will be critical for scaling AI-enabled resilience interventions within the Ruaha–Rungwa landscape.

Likelihood	5				Drought	Huma-Wildlife Conflict (HWC)
	4			Digital Divide	Market Access Barriers	
	3			Weak Institutional Responses		
	2					
	1					
		1	2	3	4	5
Impact						

Overall, the responses highlight multidimensional vulnerability across the surveyed communities. Yet, the demonstrated willingness to adopt technological innovations and engage in collaborative governance indicates a strong foundation for implementing AI-enabled interventions that can strengthen resilience, improve livelihood security, and

promote sustainable coexistence in the Ruaha–Rungwa landscape. Figure 3 further illustrate the relative severity and likelihood of key challenges faced by communities, a risk matrix is developed to categorize and visually position the major vulnerability factors identified in the baseline assessment.

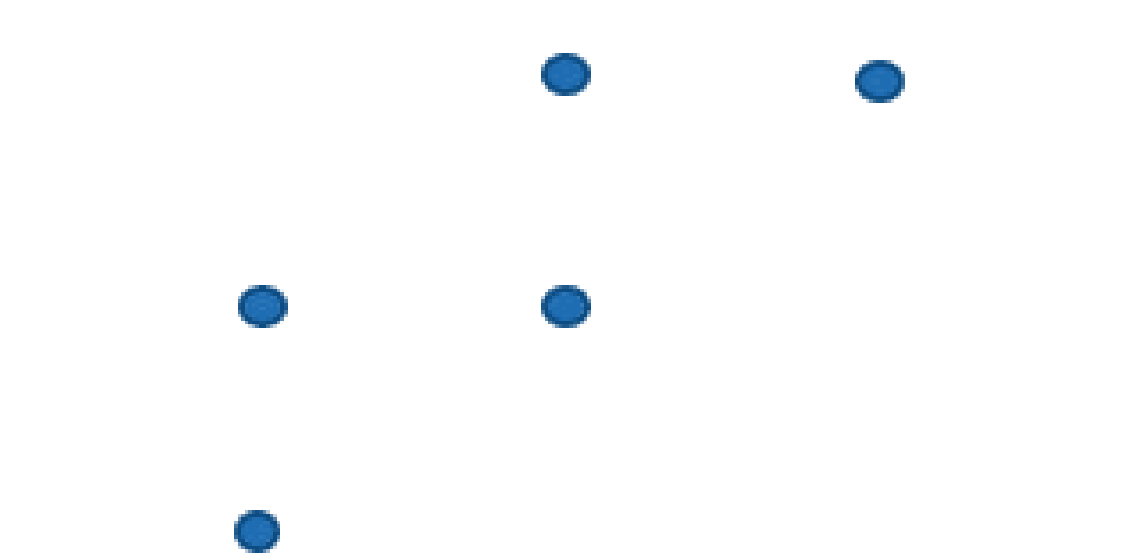


Figure 3: Risk Matrix- Climate & Community Vulnerability Factors

Overall, Table 2 summarizes the key socio-economic, environmental, and institutional indicators identified in the baseline assessment,

highlighting the principal drivers of community vulnerability and opportunities for strengthening resilience.

Table 2: The Synthesis of ESIA Key Indicators

Thematic Area	Key Indicators	Findings / Values	Implications
Demographics	Youth population (18–35 yrs)	60% of respondents	High potential for skills development and digital innovation uptake
Human–Wildlife Conflict (HWC)	Households exposed to HWC	70% within 60 km of protected areas	Frequent crop and livestock losses; heightened safety risks
Livelihoods	Land ownership and dependence on agriculture	68% rely on rain-fed farming; 1–20 ha typical farm size	Vulnerability to climate variability and reduced food security
Digital Access & AI Readiness	Smartphone ownership; interest in AI tools	35% device access; ≥58% interest in digital alerts	Strong adoption willingness if literacy and affordability gaps are addressed
Market Connectivity	Barriers to reliable markets	>90% experience challenges; 78% cite road issues	Reduced income and increased reliance on exploitative intermediaries
Climate Vulnerability	Exposure to droughts/ extreme weather	82% drought exposure; 74% rainfall variability	Increased livelihood shocks; encroachment into wildlife habitats
Institutional Capacities	Effectiveness of local governance structures	73% rely on local committees; 62% report delayed responses	Requires stronger coordination and resource empowerment

CONCLUSION

The baseline assessment reveals that communities living adjacent to the Ruaha–Rungwa ecosystem face intertwined pressures of climate variability, human–wildlife conflict, limited market access, and digital and institutional capacity gaps, all of which contribute to sustained livelihood insecurity and environmental vulnerability. However, the results also demonstrate significant readiness and willingness to adopt innovative, AI-enabled technologies that can enhance early warning systems, improve market visibility, and strengthen co-existence strategies. Collectively, these findings underscore both the urgency and the opportunity to implement locally grounded, technology-supported adaptation measures that build resilience, protect livelihoods, and support more sustainable stewardship of the Ruaha–Rungwa socio-ecological landscape.

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SUB THEME: HUMAN-WILDLIFE CONFLICTS

PERCEPTIONS AND AWARENESS OF CHICKEN FARMERS ON NEWCASTLE DISEASE AND INDIAN HOUSE CROW (*CORVUS SPLENDENS*) INTERACTIONS IN DODOMA, TANZANIA

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ABSTRACT

Newcastle disease (ND) is among the most serious threats to poultry health in Tanzania. Despite efforts to promote vaccination campaigns, outbreaks continued to be reported, particularly among smallholder farmers. This study aimed to assess chicken farmers' perceptions and awareness of ND, linked on the perceived role of Indian house crow (*Corvus splendens*)-chicken interaction as a potential transmission pathway. About 150 Household Questionnaire Surveys (HQSs), 3 Focus Group Discussions (FGDs) and 3 Key Informant Interviews (KIIs) were conducted in three wards in Dodoma region. The results showed that 75.3% (n = 113) of respondents reported that their chickens were affected by ND, while the rest had no reported outbreaks. Common symptoms reported included coughing, sudden weakness and sudden death, and the distribution of ND signs varied significantly among wards ($\chi^2 = 43.39$, $p = 0.018$). When asked about the role of Indian house crow in ND transmission, 31.8% (n = 14) of respondents in Kikuyu, 43.6% (n = 17) in Makulu and 35.8% (n = 24) in Ng'ong'ona believed the species contributes to the spread, while most disagreed ($\chi^2 = 1.271$, $df = 2$, $p = 0.530$). To reduce crow-chicken interaction, 38.6-47.8% kept chickens inside enclosures, 29.9-36.4% actively chased crows, while 20.5-25% covered feeding and watering points. These practices showed no significant variation across wards ($\chi^2 = 1.061$, $df = 4$, $p = 0.900$). Access to veterinary support was reported by 23-30% ($\chi^2 = 0.592$, $p = 0.744$), and training on ND by 44-51% ($\chi^2 = 0.55$, $p = 0.530$). Local perceptions and awareness reflect concern about ND, though understanding of transmission routes remains inconsistent. Awareness campaigns and improved access to vaccines should be prioritized to prevent the effects caused by ND to chicken farmers in Tanzania.

Key words: Chicken farmers; crow–chicken interaction; Indian house crow; Newcastle disease

INTRODUCTION

Newcastle disease (ND), caused by the virulent strains of avian paramyxovirus type 1 (APMV-1) (Ganar *et al.*, 2014), represents one of the deadly viral infections affecting poultry worldwide, particularly in developing countries where smallholder farming

systems predominate (Otiang *et al.*, 2021). In Tanzania, ND poses a significant economic threat to rural livelihoods, leading to high mortality rates in unvaccinated flocks, reduced productivity, and substantial financial losses (Hugo *et al.*, 2017). Smallholder chicken farmers, who predominantly keep indigenous breeds under free-range systems, remain particularly susceptible to disease outbreaks.

Their vulnerability is largely attributed to limited access to veterinary services. Despite national vaccination campaigns and extension initiatives, outbreaks continue to recur, driven by weak disease surveillance systems, inadequate farmer training, and potential transmission from wildlife reservoirs (Mramba *et al.*, 2025).

Emerging evidence suggests that wild birds, including corvids, may play a role in ND transmission through direct or indirect contact with domestic poultry (Johan *et al.*, 2022). The Indian House Crow (IHC), an invasive species widespread in East Africa, has been implicated in disease ecology due to its scavenging behavior, urban adaptability, and frequent interactions with human-modified environments. In Tanzania, particularly in semi-arid regions like Dodoma, where agriculture and livestock rearing intersect with expanding urban interfaces, crow-chicken interactions could facilitate pathogen spillover. For instance, crows may access poultry feed, water sources, or carcasses, potentially disseminating the virus via mechanical transmission. However, farmer perceptions of this interaction remain underexplored, hindering targeted interventions. Previous studies in Africa have focused primarily on epidemiological aspects of ND, with limited attention to socio-behavioral factors influencing disease management. This knowledge gap is critical, as perceptions shape the adoption of preventive practices and compliance with control strategies.

This study aims to assess chicken farmers' awareness and perceptions of ND, with a specific focus on the perceived role of the IHC in transmission. Using a mixed-methods approach that includes household surveys, focus group discussions, and key informant interviews in three representative wards (Kikuyu, Makulu, and Ng'ong'ona), we sought to: (1) evaluate farmers' knowledge of ND symptoms and impacts; (2) explore beliefs regarding crow-chicken interactions as a transmission route; and (3) identify adopted mitigation strategies. The findings from this research can inform

targeted awareness campaigns, improve biosecurity measures, and support sustainable poultry health management in resource limited settings.

MATERIALS AND METHODS

Geographical location

This study was carried out in rural and peri-rural areas of Dodoma, Tanzania, specifically in three wards which were purposefully selected, namely; Kikuyu (6°11'33"S, 35°45'35"E), Makulu (6°03'00"S, 35°19'59"E) and Ng'ong'ona (6°14'17"S, 35°50'04"E) (Fig. 1). These areas were selected due to their high concentration of smallholder chicken farmers and the common presence of IHC. Dodoma lies at the heart of Tanzania and functions as its political capital (6.17°S, 35.74°E). The region sits on broad upland plateaus, with elevations typically between **1,100–1,500 meters** above sea level, interspersed with rolling hills and rocky outcrops (Awoke *et al.*, 2023). The climate is semi-arid, featuring one rainy season from November/December to April/May during which average annual rainfall ranges from **500 to 600 mm**; outside that period, the dry season prevails with high temperatures (up to ~35°C) and occasional drought (Brüssow & Grote, 2017).

Vegetation is dominated by savanna types; wooded grasslands, bushlands, scattered dry woodland (notably *Acacia* and *Baobab*), with soils that are sandy, loamy or mixed, often low in fertility and with limited water retention. Agriculture is the mainstay of the economy: over half the population engages in farming, growing crops such as maize, sorghum, millet, beans, sunflower, and grapes (Awoke *et al.*, 2025). Livestock (including poultry) also play a central role. Land use changes have been significant in recent decades: built-up areas and bare land have expanded substantially, while vegetation cover has declined (Mbilingi, 2020)

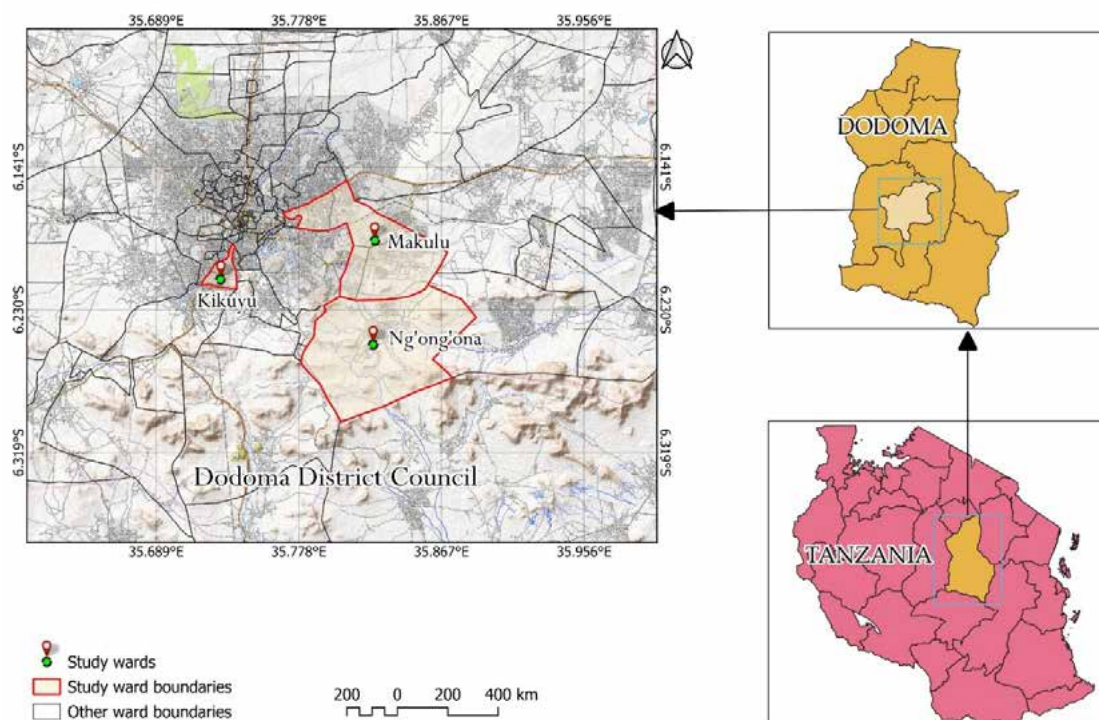


Figure. 1. The three study wards selected for this study (Makulu, Kikuyu and Ng'ong'ona; Prepared by authors using QGIS)

Individuals recruited for data collection

This study adopted a cross-sectional study design, a method in which data are collected at a specific point in time. This design is useful for generating preliminary information that can guide the development of future, more advanced studies (Chebby *et al.*, 2024). Participants were recruited from three wards – Makulu, Ng'ong'ona and Kikuyu. With permission and assistance from local government leaders, a list of poultry keepers was compiled for each study ward. Thereafter, respondents were selected using the random function in Microsoft Excel “= RAND” from the sampling frame prepared for each ward. A proportionate sampling approach was used to ensure that the number of respondents from each ward reflected the relative distribution of poultry farmers. At the end, a total of 150 respondents were obtained for this study, distributed as follows: Makulu = 39, Ng'ong'ona = 67 and Kikuyu = 44.

Data Collection

During the survey, three data collection techniques were employed to enhance the validity of findings through triangulation (Donkoh & Mensah, 2023). Firstly, semi-structured questionnaires (SSQs) were administered to the selected respondents. Through this tool, we sought to assess poultry keepers' awareness of the signs and modes of transmission of ND, their perceptions of the IHC as a potential carrier of the disease and the self-imposed measures they adopt to prevent the interaction between IHCs and their chickens. Secondly, one Focus Group Discussion (FGD) was conducted in each study ward to obtain more detailed insights on the topic under investigation. Each FGD consisted of six to eight discussants, as recommended by Chebby *et al.* (2023). The guiding questions were adapted from SSQs but were slightly modified to encourage deeper discussion and gather additional perspectives. To avoid repetition of

ideas, participants in FGDs were different from those who took part in the questionnaire survey. Lastly, Key Informant Interviews (KIIs) were conducted with three Livestock and Extension Officers, using a checklist of guiding questions related to the study topic. These interviews provided expert perspectives and contextual information that complemented data obtained from the questionnaires and FGDs.

Data analysis

The collected data were classified, organized, cleaned and arranged using Microsoft Excel (Microsoft Corporation, Washington, USA). Statistical analyses were then performed using RStudio (Version 2025.09.1 Build 401) at a significance level of 0.05 ($\alpha = 0.05$). The distribution pattern of the data was examined using the Kolmogorov–Smirnov (K-S) test, which is appropriate for sample sizes greater than 50. Statistical analyses were chosen based on the type of variable (categorical, continuous or ordinal) and the normality of the data. Categorical variables were analyzed using the Chi-square test to determine associations, while continuous variables were summarized using descriptive statistics and compared between groups using Kruskal-Wallis tests. Correlation analyses (e.g. flock size vs number of affected) were performed using Spearman’s rank test where appropriate. These methods ensure robust evaluation of socio-demographic factors, flock characteristics, ND occurrence, seasonal patterns and crows-chicken interaction, to name a few. To enhance clarity and interpretability, the results were

presented using appropriate tables, graphs and figures, including bar charts to visualize trends and associations among variables. Moreover, qualitative data from FGDs and KIIs were analyzed using thematic content analysis. Responses were systematically coded and organized into key themes and sub-themes to identify patterns, similarities and differences across participants (Chebby *et al.*, 2024). The results were presented in the form of quotes to illustrate participants’ perspectives and support the identified themes.

RESULTS

Socio-demographic Profile of the Respondents across Study Wards

The socio-demographic characteristics of respondents differed slightly across the three study wards (Kikuyu, Makulu and Ng’ong’ona). However, none of this variation was statistically significant at $P < 0.05$ (Table 1). Gender distribution was relatively balanced across wards, with no significant difference observed ($\chi^2 = 0.712$, $p = 0.700$). Similarly, the distribution of respondents by age category did not differ significantly between wards ($P = 0.324$). Differences in educational attainment were also not statistically significant ($\chi^2 = 6.814$, $P = 0.338$), although respondent exhibited a range of formal and informal education levels. Likewise, the proportions of respondents engaged in various occupations such as micro-enterprises and farming did not vary significantly among the wards ($\chi^2 = 4.685$, $P = 0.585$) (Table 1).

Table 1: Socio-demographic profile of the respondents from three study wards (Kikuyu, Makulu and Ng’ong’ona)

Sn	Variable	Category	Kikuyu, n (%)	Makulu, n (%)	Ng’ong’ona, n (%)	Test	P
1	Gender	Female	19 (43.2)	19 (48.7)	27 (40.3)	0.712	0.700
		Male	25 (56.8)	20 (51.3)	40 (59.7)		
2	Age	18–40 (Youth)	23 (52.3)	29 (74.4)	40 (59.7)	NA	0.324
		41–60 (Adult)	17 (38.6)	9 (23.1)	23 (34.3)		
		>60 (Old)	4 (9.1)	1 (2.6)	4 (6)		

Sn	Variable	Category	Kikuyu, n (%)	Makulu, n (%)	Ng'ong'ona, n (%)	Test	P
3	Education level	College	16 (36.4)	19 (48.7)	18 (26.9)	6.814	0.338
		Informal	9 (20.5)	3 (7.7)	12 (17.9)		
		Primary	8 (18.2%)	7 (17.9)	15 (22.4)		
		Secondary	11 (25)	10 (25.6)	22 (32.8)		
4	Occupation	Micro-enterprise	14 (31.8)	19 (48.7)	27 (40.3)	4.685	0.585
		Casual work	8 (18.2)	3 (7.7)	10 (14.9)		
		Farming	9 (20.5)	5 (12.8)	14 (20.9)		
		Employed	13 (29.5)	12 (30.8)	16 (23.9)		

Flock Size and Composition

The mean flock sizes vary among wards, with Kikuyu recording the largest average flock size (85.2 ± 113.3 chickens), followed by Makulu (74.1 ± 116.3 chickens) and Ng'ong'ona (69.3 ± 80.8 chickens) (Fig. 2). Considerable variation in flock size was observed within each ward, however a Kruskal – Wallis test showed no significance difference in flock size among the three wards (Kruskal-Wallis, $H =$

3.36 , $P = 0.186$). The composition of chicken types differed slightly among wards. Broilers accounted for the largest share of flocks across all wards (ranging from 43 – 51%), followed by local chickens (29 – 37%), while layers constituted approximately one-fifth of all chickens (20 – 21%), although the difference was not statistically significant across three wards ($\chi^2 = 0.805$, $p = 0.938$) (Fig. 3).

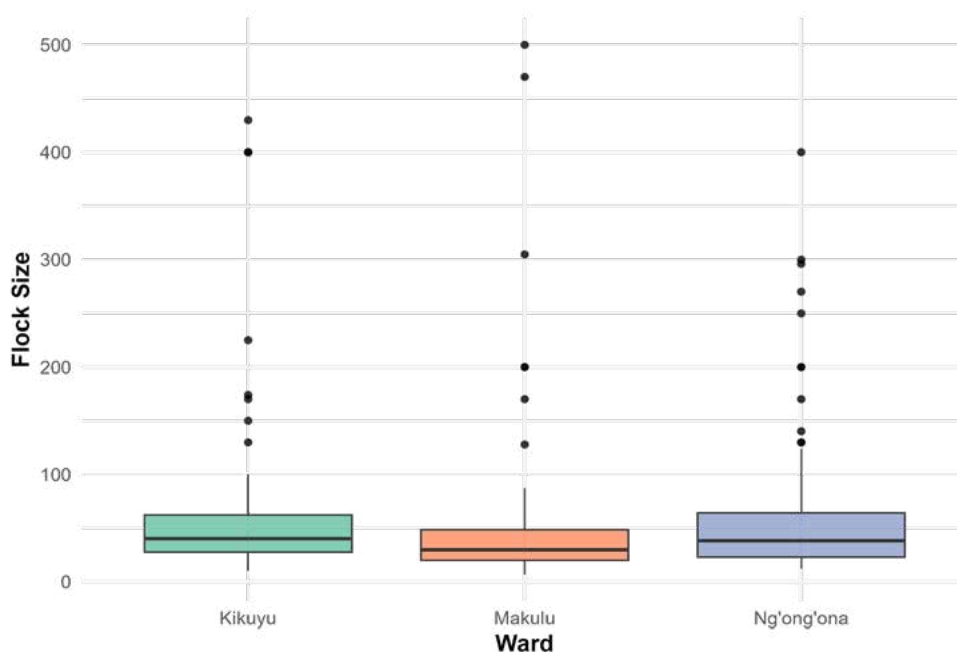


Figure. 2. Distribution of chicken flock sizes cross three study wards (Kikuyu, Makulu and Ng'ong'ona)

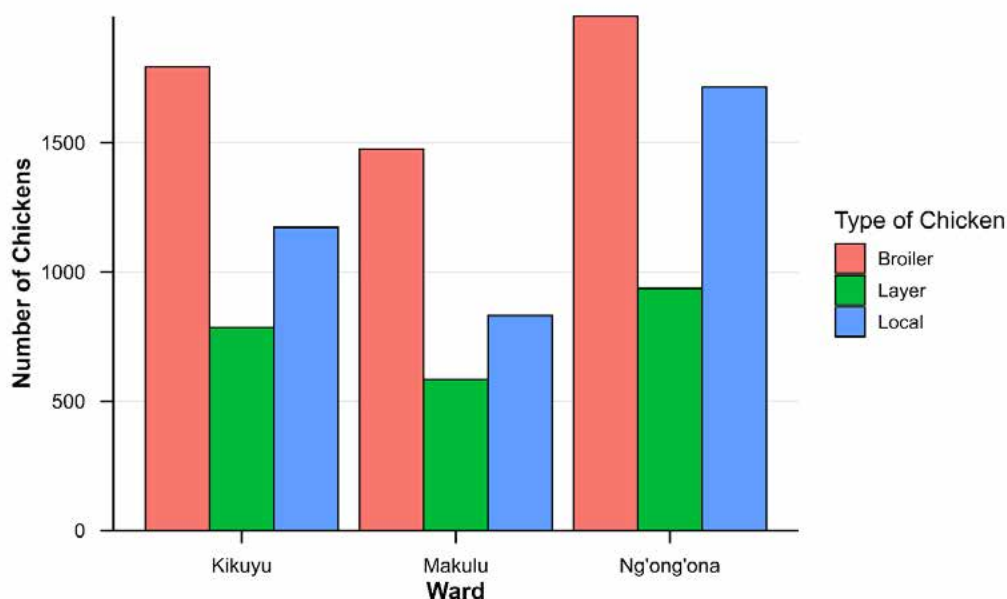


Figure 3. Distribution of chicken types (i.e., Broiler, Layer, and Local chickens) across three study wards (i.e., Kikuyu, Makulu and Ng'ong'ona)

Prevalence of Newcastle Disease, Flock Impact and Observed Signs

A total of 113 respondents reported that their chickens were affected by ND. The mean number of chickens affected by ND did not differ significantly among wards ($P = 0.400$) (Table 2). Regarding chicken categories, there was no significant difference across wards for the number of hens ($P = 0.163$), roosters ($P = 0.159$), cockerels ($P = 0.096$), or pullets ($P = 0.103$) affected, based on Kruskal-Wallis tests. However, significant differences were observed for chicks ($P = 0.036$) and total deaths ($P = 0.019$), with Ng'ong'ona having higher

numbers of chicks affected (4 ± 8.6) and deaths (6.5 ± 6.3) compared to Kikuyu and Makulu (Table 2).

Moreover, the distribution of ND signs varied significantly among wards. Broader examination of reported signs indicated that symptoms, such as coughing and sudden death, were more prevalent in specific wards. A Chi-squared test confirmed that the differences in the frequency of observed ND signs across wards were statistically significant ($\chi^2 = 43.39$, $p = 0.018$), suggesting that the pattern of ND manifestation is not uniform across locations (Fig. 4).

Table 2: Newcastle disease impact on chicken flocks across three study wards

Ward	Number of respondents	Affected chickens	Hens	Roosters	Cockerels	Pullets	Chicks	Died
Kikuyu	33	12.6 ± 12.3	3.6 ± 6.7	2.3 ± 3.9	1.6 ± 2.4	2 ± 4.6	1.1 ± 3.4	2.6 ± 2.3
Makulu	29	9.8 ± 12.9	1.5 ± 1.8	1.2 ± 1.5	2.6 ± 4.4	2 ± 2.2	2.8 ± 7.5	3.5 ± 3
Ng'ong'ona	51	13.9 ± 16.4	3.6 ± 8	1.2 ± 1.4	1.7 ± 2.6	2.5 ± 4.2	4 ± 8.6	6.5 ± 6.3
	P	0.4	0.163	0.159	0.096	0.103	0.036	0.019

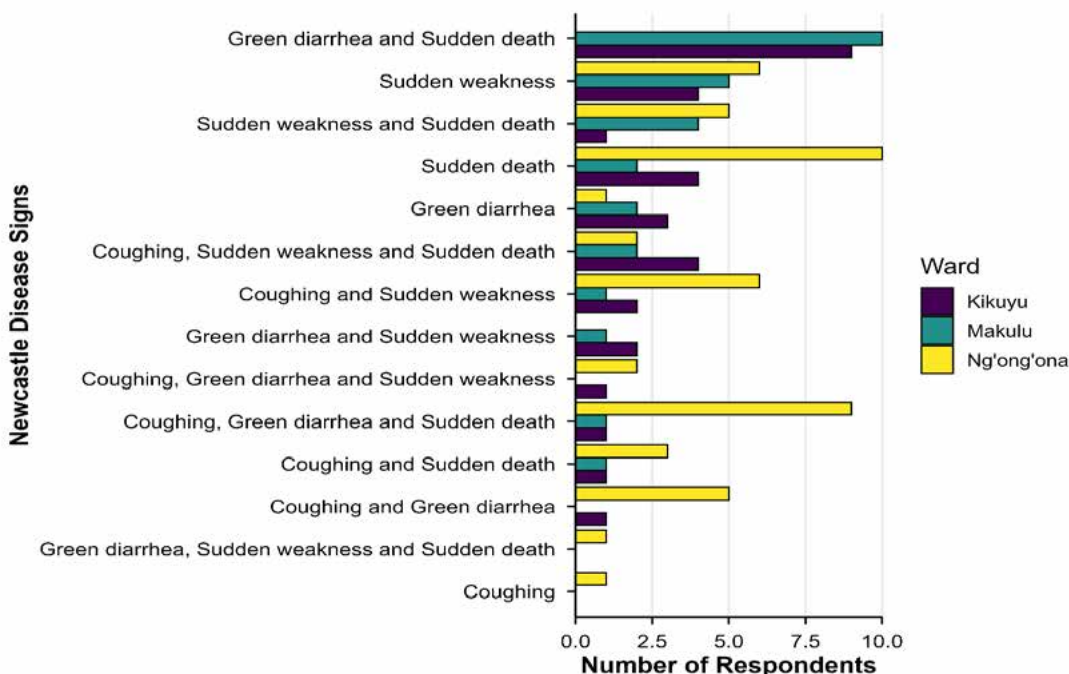


Figure 4. Frequency of Newcastle disease signs observed per ward

Seasonal Outbreak of Newcastle Disease across the Study Wards

The occurrence of ND outbreaks varied across seasons in all three wards. In Kikuyu, the highest number of respondents reported ND between April – June (n = 13), while Makulu exhibited similar trend, with peaks in July –

September (n = 14) and April – June (n = 12). Ng'ong'ona experienced the highest outbreaks in April – June (n = 28) and July – September (n = 18). Statistical analysis showed that the differences in outbreak counts among wards were not significant ($F = 0.27$, $P = 0.77$) (Fig. 5).

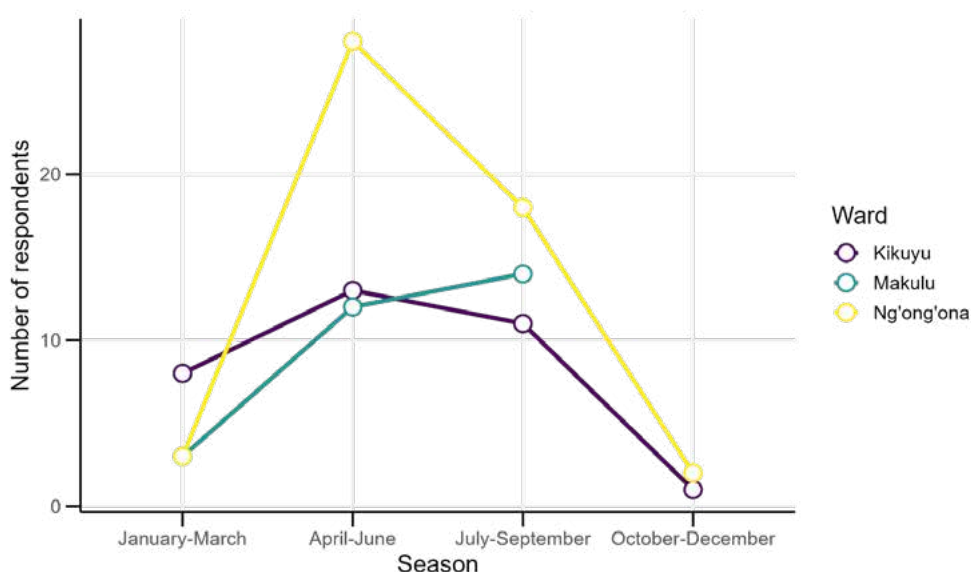


Figure 5. Seasonal trends of Newcastle disease outbreaks by ward

Perceived Contribution of Indian House Crows to Newcastle Disease Transmission

31.8% (n = 14) of respondents in Kikuyu, 43.6% (n = 17) in Makulu, and 35.8% (n = 24) in Ng'ong'ona agreed that IHC spread ND, while most disagreed: 68.2% (n = 30), 56.4% (n = 22), and 64.2% (n = 43), respectively, with no significant difference across wards ($\chi^2 = 1.271$, df = 2, p = 0.530). These findings were supported by one of the key informants from Makulu who asserted that; *"Honestly, I don't think the crows are the main cause of ND. I've seen my chickens fall sick even when*

there are no crows around. Maybe the disease comes from other birds or when we buy new chickens. But still, I chase the crows away and keep my chickens enclosed, just to be safe" Preventive measures against ND as associated with chicken-crow interaction were broadly consistent across wards, with most respondents relying on enclosures to protect chickens, followed by active chasing of crows and covering of feeding or watering points (Table 3). Although slight numerical differences were observed between wards, the variation was not statistically significant ($\chi^2 = 1.061$, df = 4, p = 0.900) (Table 3).

Table 3: Preventive strategies against Newcastle disease associated with chicken–crow interactions

Preventive strategies	Kikuyu	Makulu	Ng'ong'ona	Test statistic	P
Actively chased crows	16 (36.4%)	13 (33.3%)	20 (29.9%)	1.061	0.900
Covered feeding and watering points	11 (25%)	8 (20.5%)	15 (22.4%)		
Kept chickens inside enclosures	17 (38.6%)	18 (46.2%)	32 (47.8%)		

Access to Veterinary Support and Training

Access to veterinary support and training on ND were generally similar across the three wards. For veterinary support, 29.5% (n = 13) of respondents in Kikuyu, 23.1% (n = 9) in Makulu and 23.9% (n = 16) in Ng'ong'ona reported having access, with no significant

difference between wards ($\chi^2 = 0.592$, p = 0.744) (Fig. 6). Regarding training, 50% (n = 22) of respondents in Kikuyu, 43.6% (n = 17) in Makulu and 50.7% (n = 34) in Ng'ong'ona reported receiving training on ND, also showing no significant variation across wards ($\chi^2 = 0.55$, p = 0.530) (Fig. 6).

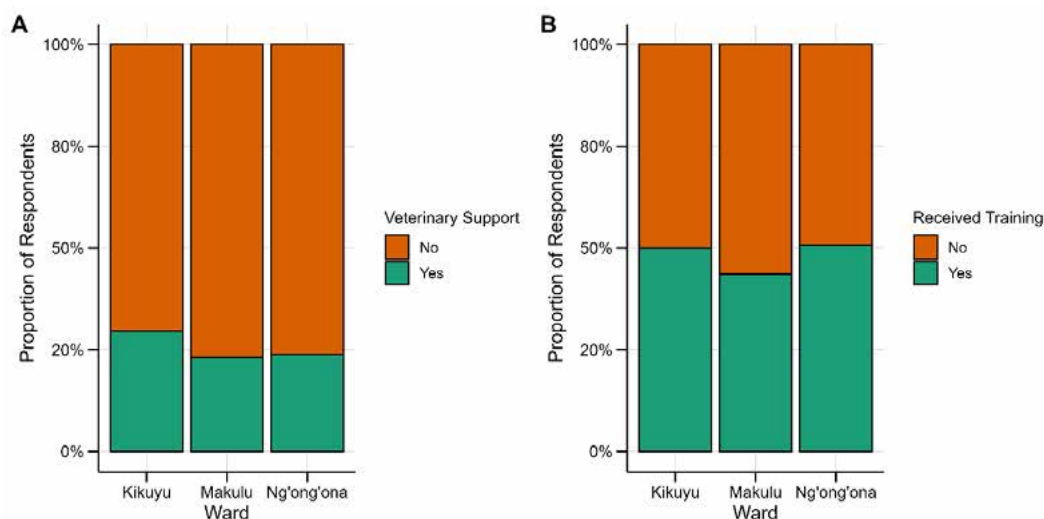


Figure 6. (A) Access to veterinary support by ward and; (B) Training in Newcastle disease by ward

DISCUSSION

The results of this study highlight a moderate level of awareness among smallholder chicken farmers in Dodoma regarding Newcastle disease (ND), with most respondents recognising the disease and reporting recurrent outbreaks. This awareness aligns with previous reports from Tanzania, where ND is widely regarded as a major constraint to poultry production (Mramba *et al.*, 2025). However, a smaller proportion of farmers still lacked clear understanding, reflecting persistent knowledge gaps likely linked to limited access to extension services and veterinary support. Commonly reported symptoms such as sudden weakness and unexpected death were consistent with classical ND presentations (Ipara *et al.*, 2024). This suggests uniform disease patterns across wards, though variations in observed signs may reflect ward-specific environmental or management factors, such as flock density or seasonal stress (Abdisa & Tagesu, 2017) with viruses of the other/night serotypes (avian paramyxovirus 1-9).

Most flocks were moderate size and largely composed of broiler chickens, indicating a gradual shift toward commercial-oriented production even among smallholders. This implies that husbandry practices are relatively similar across wards, yet ND impacts remained evident. Higher chick mortality and total deaths in some wards could be attributed to denser populations or weaker biosecurity, conditions that are common in semi-arid areas facing water scarcity and heat stress (Dinka *et al.*, 2010; Sesay, 2022) lack of proper health care, poor feeding and poor marketing information. Replacement of indigenous chickens by exotic chicken breeds is also a major threat in eroding and dilution of the indigenous genetic resources. Newcastle Disease (NCD. Seasonal disease patterns appeared to peak during the post-rainy and dry seasons, when nutritional and environmental stressors tend to increase susceptibility (Molia *et al.*, 2017). This pattern highlights the influence of climate and supports the need for a well-timed vaccination programs.

Farmers' perceptions of IHC as ND transmitters were mixed. While some farmers believed crows could spread the disease, others regarded them as harmless nuisances. This limited awareness contrasts with ecological studies that identify corvids as potential carriers of ND and other zoonotic pathogens (Johan *et al.*, 2022; Ryall, 2016). Qualitative findings indicated that farmers were aware of crows scavenging on dead birds, yet many did not link this behavior to disease transmission. Preventive strategies such as confining poultry and discouraging crow contact were adopted differently across areas, reflecting differences in available resources, although cultural practices appeared generally similar among wards. Socio-demographic characteristics, including gender, age, education, and occupation, did not significantly influence awareness, although farmers with higher education tended to demonstrate better understanding, consistent with Joshi *et al.* (2020).

Furthermore, access to training was moderate, indicating that while some farmers had received exposure to poultry health education, other still lacked access to structured learning and technical support. This pattern aligns with observations from other African context, where farmer education has been shown to influence awareness and adoption of vaccination practices (Waweru *et al.*, 2023).

CONCLUSION

This study shows that while awareness of ND is relatively high among chicken farmers in Dodoma, Tanzania, there are still misunderstandings about transmission routes, especially involving IHC, which contribute to ongoing outbreaks. Key findings include moderate flock impacts, seasonal peaks in disease occurrence, and varying adoption of preventive measures like enclosures and crow deterrence. These insights highlight the need for targeted interventions such as community awareness campaigns, better vaccine access and management, control and eradication of IHC to alleviate crow-chicken interactions. To improve poultry health and farmer livelihoods,

policymakers should focus on extension services in semi-arid regions, incorporating socio-cultural perceptions to promote sustainable practices. Limitations of the study include the cross-sectional design, which captures snapshots rather than longitudinal trends, and reliance on self-reported data, potentially subject to recall bias. Future research should incorporate serological testing and crow surveillance to validate their transmission roles. Overall, integrating local perceptions into policy could strengthen ND control, reducing economic burdens Tanzania's poultry sector. Future research could investigate molecular epidemiology to confirm crow involvement in ND transmission. By addressing these gaps, Tanzania can move toward effective ND control and improve food security.

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A POISONING INCIDENT INVOLVING A WHITE-BACKED VULTURE (WBV) IN MIKUMI NATIONAL PARK, TANZANIA: A CASE REPORT AND REVIEW OF AN EMERGING CRISIS

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ABSTRACT

We report a poisoning event involving a Critically endangered white-backed vulture (*Gyps africanus*) linked to an elephant (*Loxodonta africana*) carcass in Mikumi National Park, Tanzania. Following an alert, a veterinary team discovered a decomposed elephant carcass surrounded by six dead white-backed vultures. One live vulture was found exhibiting severe clinical signs including dull demeanor, weakness, flaccid neck paralysis (astasia), white diarrhea, hypoventilation, delayed palpebral reflex, and oral regurgitation. Based on the scene and clinical presentation, a tentative diagnosis of pesticide poisoning was made. The individual was transported to the Sokoine University of Agriculture Animal Hospital for intensive care. Immediate treatment included atropine (4 ml), intravenous fluid therapy (900 ml Dextrose Normal Saline), and activated charcoal was given to animal. The vulture showed significant improvement within 24 hours, regaining alertness, the ability to stand, and normal respiratory function. It was successfully released after six days of supportive care. This case is not isolated; it forms part of a severe and escalating wildlife poisoning crisis in the Eastern and Southern Highlands of Tanzania. A review of incident data from 2015 to 2025 reveals that over 562 terrestrial and aquatic animals have been killed in similar events across Wildlife Management Areas, game reserves, and National Parks. We conclude that there is an urgent need for a multi-faceted response involving community education on the dangers of pesticides, engagement with agrochemical firms on responsible product distribution, and enhanced law enforcement and veterinary surveillance to mitigate this critical threat to Tanzania's biodiversity.

Keywords: *Gyps africanus*, Carbofuran, Pesticide poisoning, Vulture decline, Tanzania

INTRODUCTION

Vultures provide critical ecosystem services by rapidly consuming carrion, which helps control the spread of zoonotic diseases and recycle nutrients (Ogada *et al.*, 2012). However, vulture populations are experiencing catastrophic declines across Africa, primarily driven by poisoning, both intentional and incidental (Ogada *et al.*, 2016). The white-backed vulture (*Gyps africanus*) is listed as Critically Endangered on the IUCN Red List, with poisoning representing the single greatest threat to its survival (BirdLife International, 2021).

In Tanzania, poisoning incidents have emerged as a severe conservation crisis. About 562 animals have been reported dead in poisoning incidences in this landscape from 2015 to 2023. The incidences involved 13 lions, 100 cat fish, 312 vultures, 100 pigeons, 30 guinea fowls, 3 warthogs, 1 impala, 1 hippo, 1 hyena and 1 elephant from Wildlife Management Areas (WMAs), open areas, game-controlled areas, game reserves, conservation areas and national parks. Both terrestrial and aquatic environments have been involved in events. These events typically fall into two categories: (1) retaliatory poisoning, where livestock predators such as lions, hyenas, or leopards are targeted by lacing

a livestock carcass or bait with pesticides, also in Human Elephant conflict farmers use poison for revenge after crops being destroyed by Elephant; and (2) sentinel poisoning, where elephant carcasses are laced with highly toxic pesticides to eliminate vultures, whose circling flight patterns can reveal poachers' activities to law enforcement agencies (Odino & Ogada, 2021; Nyirenda *et al.*, 2024). In both scenarios, the poison leads to mass mortality of non-target scavengers, particularly vultures.

The Eastern and Southern Highlands of Tanzania, encompassing a network of protected areas including Nyerere National Park, Mikumi National Park, Udzungwa National Park, Ruaha National Park, Selous Game Reserve, and associated Wildlife Management Areas (WMAs), have become a hotspot for such incidents. The widespread availability of highly toxic agricultural pesticides, such as carbamate (e.g., Carbofuran) and organophosphate compounds, facilitates this illegal practice (Odino & Ogada, 2021).

This paper presents a detailed case report of the successful rescue, treatment, and release of a white-backed vulture poisoned at an elephant carcass in Mikumi National Park. Furthermore, we contextualize this single event within a decade-long pattern of wildlife poisoning in the region, highlighting the scale of the problem and the urgent need for a coordinated national response and to include poison management protocol in National action plan for vulture conservation.

MATERIAL AND METHOD

Study Area

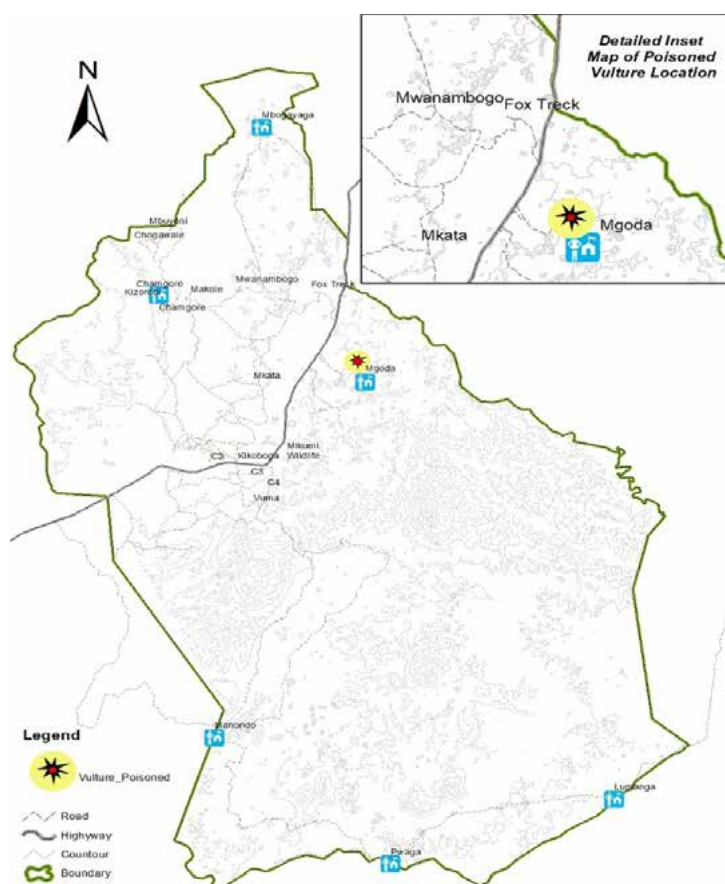
The incident occurred at Korongo la Mgoda, located in the northern sector of Mikumi National Park in eastern Tanzania at 7.24290°S,

37.24163°E (Map 1). Mikumi National Park is one of the country's most accessible protected areas and receives a high volume of domestic tourism due to its proximity to major road networks and the visibility of iconic large mammals such as giraffes, elephants, lions, buffaloes, zebras, wildebeest, and impalas. Owing to its location, the park is bordered by several villages to the north, west, and south, where human–wildlife interactions are frequent. These interactions occasionally escalate into human–wildlife conflict, with wildlife poisoning emerging as one of the documented outcomes in the surrounding landscape.

Incident Alert and Field Investigation

On 15 August 2024, the Tanzania National Parks (TANAPA) Eastern zone veterinary unit received an alert from anti-poaching rangers regarding an elephant carcass at Korongo la Mgoda in Mikumi National Park. A field response team, comprising an animal health technician and conservation rangers, was dispatched to the site.

Upon arrival, the team discovered a decomposed elephant carcass, estimated to be 3–4 days old. A rapid perimeter survey revealed six carcasses of white-backed vultures scattered within a 1–5-meter radius from the elephant's carcass discovered. During the assessment, a single live white-backed vulture was observed on the ground, unable to fly or escape. Alive vulture was moved to Sokoine University of Agriculture teaching animal hospital for treatment. The remain carcasses of both elephants and the six birds (WBV) were buried underground.



Map 11: Location of the poisonous incidence occurred in Mikumi National Park forming this case study report.

Clinical Presentation

The live vulture was apprehended for clinical examination and exhibited the following signs: Neurological (Dull demeanor, profound weakness, flaccid paralysis of the neck (astasia), and delayed eye-opening (palpebral reflex)), Gastrointestinal (clear fluid regurgitation from the mouth), Respiratory (Marked hypoventilation (shallow, slow breathing)). The combination of neurological signs (muscle weakness, paralysis) and gastrointestinal distress in a scavenger found at a carcass site is highly indicative of acute pesticide poisoning, most commonly from carbamate or organophosphate compounds (Eaton *et al.*, 2008; Gupta, 2019).

Treatment and Supportive Care.

A tentative diagnosis of acute pesticide poisoning was made this was based with clinical symptoms presented. Symptomatic and tentative diagnosis treatment were administered to the Vulture as following; firstly provision of Atropine Sulfate: A total of 4 ml was administered intramuscularly at right breast muscles given as five 0.8 ml administration volume at 8 hours interval for an antidote to counteract the effects of acetylcholinesterase inhibition, which is the mechanism of toxicity for carbamate and organophosphate pesticides, A total of 900 ml of Dextrose Normal Saline (DNS) was administered intravenously through the wing vein (Brachial vein), given as three 300ml administered volume at 8hour intervals, to combat dehydration and support renal function, Secondly Decontamination: Two tablets of medicinal-grade activated charcoal were dissolved in water and administered

orally by using 20ml syringe to adsorb any remaining toxin in the gastrointestinal tract. Supportive care included housing the bird in a quiet, dark enclosure and providing fresh water and easily digestible food (pieces of goat meat) was provided.

RESULT

According to the symptomatic and tentative treatment it was hence proved that the animal was either poisoned by carbamate or organophosphate poison. This is due to the fact that the preliminary drug used to counteract the effect of poison in the acetylcholinesterase receptor. Hence, the observed progress within 12 hours of initial treatment, the vulture showed marked improvement. Breathing normalized, and the bird became more alert. By 24 hours, it was able to stand and maintain an upright posture. The atropine and supportive care were continued for three days, after which the bird was self-feeding and displaying normal behavior. After six days of management and observation, the vulture was deemed fit and then released back into the wild. The time taken to release the animal differ based on treatment, dosage and level of exposure as the similar case happen in Isele Village took about 14 days for animal to be released back into the wild.

DISCUSSION

The clinical signs observed in this vulture flaccid paralysis, respiratory depression, and clear fluid vomitus are classic manifestations of acetylcholinesterase inhibitor poisoning which includes Organophosphates (OPs) and carbamates (Friend *et al.*, 1999); Eaton *et al.*, 2008; Gupta, 2019). The rapid positive response to atropine strongly supports the diagnosis of poisoning by a carbamate (e.g., Carbofuran, Carbaryl) or organophosphate compound. Atropine acts as a competitive antagonist at muscarinic receptors, reversing the life-threatening respiratory and cardiac effects (Friend *et al.*, 1999). The use of activated charcoal is a critical step in gastrointestinal

decontamination, though its efficacy diminishes significantly if administered long after toxin ingestion. DNS fluid therapy was administered IV for boosting energy as bird rescue, since the bird was much lethargy also to help renal activities on toxic clearance (Antoniou *et al.*, 2021).

It is important to know that, the lethal dose of toxicity depends on the amount of food consumed, which is directly related to the vulture's arrival time at the carcass (Friend *et al.*, 1999). Vultures that arrive early to the poisoned carcass consume larger quantities than the one come later. Therefore, the rescued vulture was probably arrived late compared to others, hence reduced its exposure to the toxin. The success in this case underscores the importance of rapid response and the application of standard veterinary toxicology principles to wildlife.

Whilst several case report in Africa such as Kruger, Namibia and Kenya protocol used for treating the bird did not differ significantly from the authors choice. It has been observed that the universally recommended immediate actions for suspected organophosphate/carbamate poisoning in vultures are: rapid triage and scene recording, administration of atropine as an emergency antidote, fluid therapy (IV or subcutaneous Ringer's/lactate or saline) to support circulation/renal clearance, activated charcoal if the bird is stable and ingestion is recent, minimization of stress/handling, and rapid transfer to a veterinary facility when possible. On the other hand, some national protocols and NGO manuals also include use of oximes (e.g., pralidoxime/2-PAM) for organophosphate poisoning and guidelines for repeated atropinization until muscarinic signs resolve (Moretto & Lotti, 2001; 4Vultures, 2013; Hirschauer *et al.*, 2019; Odino & Ogada, 2021; Antoniou *et al.*, 2021; Gupta & Doss, 2022). This single case is a symptomatic event of a much larger epidemic, as our review of records from TANAPA, TAWIRI, and other conservation entities from 2015 to 2025 documents over 562 wildlife

mortalities attributed to poisoning across the Eastern and Southern Highland landscapes. The scale and diversity of affected species are alarming; for instance, one incident recorded in August 2023 in Ruaha National Park, when eighty-nine (89) deaths suspected of poisoning observed. The incident involved 1 adult Hippopotamus, 81 adult vultures (74 white-headed, six (6) Ruppel's and one (1) hooded), 1 adult hyena and 6 lions (two adult female, 1 adult male, 2 subadult male and 1 juvenile) at Mhawa area (Photo 1 A-G). The killing of

these animals was suspected to be caused by poaching through poisoning. As the poachers killed and poisoned the carcass of a Hippo that was later scavenged by Lions, vultures and the Hyena was suspected to be poisoned by scavenging on dead carcasses of lions that died of primary poisoning. This was a typical sentinel poisoning, since out of six lions only one was found intact, while the rest five lions their heads, feet, palms, and tails were missed (amputated) (Photo 1 D-G).



Photo 1 A-G: Carcasses of animals died of poison in Ruaha National Park in August 2023. Photo A-B: Vultures, Photo C: Hippopotamus, Photo D, E & F: Lions amputated their body parts, and Photo G: Intact Lion

This incident is correlated OPs and carbamate toxicity, similar to other incident occurred at Mafiga Matatu swamps in the park in July 2023. The incident involved the poisoning of water which led to death of some animals including impala. From the bottle collected in the scene, it was revealed to have 155.4mg/l of Diazinon as Organophosphate (Orthophosphate

residual) while the water was detected with organophosphate chemical at 2.23 mg/l. These situations not only result into killing of wildlife, but also contaminating the environment.

The two primary drivers are retaliatory and sentinel poisonings. The retaliatory poisoning is when predation on livestock incites farmers

and pastoralists to lace carcasses with easily accessible pesticides, leading to the death of lions, hyenas, and wild dogs, and subsequently, any scavengers that feed on the bait. While the sentinel poisoning is as evidenced in these cases, poachers targeted elephant for ivory, lions for their body parts and fish for food deliberately poison the carcasses to eliminate vultures, whose presence in the sky is a reliable indicator of a kill site for rangers and their population is declining globally (BirdLife International, 2021; Westrip *et al.*, 2022).

In wildlife poisonings, both the environmental contaminants, natural chemicals, microbial and other biotoxins are commonly cause wildlife mortality (Friend *et al.*, 1999). Direct poisoning and mortality of wildlife are the results of exposure to chemical toxins. However, Friend *et al.* (1999) further described that, the indirect effects of chemicals may have significantly greater impacts on wildlife populations than the direct effects. The examples of indirect chemical toxicity include behavioral changes that affect survival, reproductive success and the survival of young, and that impair the functioning of the immune system.

Our study identified a widespread availability of highly toxic pesticides like Carbofuran (sold as Furadan), Carbaryl (sold as Sevin), OPs like Diazinon, synthetic pyrethroids and avicides has been associated with various wildlife poisoning in the landscapes. Some of these pesticides available in local agro-vet shops and others not, for instance, the controlled avicides used against grain eating birds in particular *Quelea quelea*. has been associated with wildlife poisoning in Mvomero district. The poachers used the smuggled avicides to soak grains like maize and rice before baiting them into areas with high number of doves/pigeons, guinea fowls and aquatic birds especially wild ducks. Misbranded pesticides like CANINE PLUS DUST 113DP containing Fenitrothion 1g/Kg + Deltamethrin 1.3g/Kg, registered for control of larger grain borer (*Prostephanus truncatus*) and weevils has been used for lion poisoning in

agropastoral communities. The name and lion image on the container have influenced the users to believe on its manufacturer as for killing of lions. Therefore, it is clear that in Tanzania, the misuse of pesticides is often with inadequate regulatory oversight, is a key enabling factor for their applications on wildlife poisoning (Odino & Ogada, 2021).

CONCLUSION AND RECOMMENDATIONS

The rescue of a single white-backed vulture, while a conservation success, highlights a devastating and ongoing threat to Tanzania's wildlife. The mass poisoning events documented over the past decade represent a significant drain on the ecological integrity of the country's protected areas and a direct threat to its tourism economy.

To effectively combat this crisis, we recommend an urgent, multi-pronged strategy; Community Engagement and Awareness: Launch targeted education campaigns in villages bordering protected areas to highlight the ecological role of vultures and carnivores, the legal consequences of poisoning, and the extreme dangers these pesticides pose to human health, livestock, and water sources, Regulatory Action and Corporate Responsibility: Strengthen government regulation and monitoring of the import, sale, and use of Class I pesticides. Engage pesticide manufacturers and distributors in promoting safer alternatives and implementing traceability systems for high-risk products, Enhanced Law Enforcement and Surveillance: Increase rapid-response capabilities for poisoning incidents, including dedicated veterinary and forensic teams. Integrate poisoning alert networks into existing anti-poaching efforts and utilize intelligence-led operations to disrupt the supply chains of pesticides used for wildlife crime. The time for action is now. Without a concerted effort, the continued loss of keystone scavengers and predators will have irreversible consequences for Tanzania's world-renowned ecosystems.

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CROP PREFERENCES OF AFRICAN ELEPHANTS: A COMPARATIVE STUDY OF CROP DAMAGES TO FARMERS ADJACENT TO MIKUMI NATIONAL PARK, TANZANIA

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ABSTRACT

The elephant crop raiding behavior has been reported to be influenced among other factors by seasonality and crop preferences in villages adjacent to Mikumi National Park. However, while some studies suggest that elephant crops preference exacerbate crop damage, more research is needed to enhance understanding of attractiveness and preferences of crops to elephants and potential alternative crops for farmers. This paper seeks to conduct a comparative assessment of crops' preferences and damage by elephants towards establishing sustainable mitigation measures. Data on the extent for crop damages and preferences was collected from July to August 2023 to 405 affected households through household questionnaire surveys from 10 villages adjacent to MNP. Descriptive analysis was run in R to produce bar graphs on crop preferences and damages. The results indicated that Maize was most highly preferred by elephants followed by Pigeon peas, Sorghum and Tomato, while Sesame was not preferred by elephants. Kendall's Coefficient of Concordance for agreement among respondents on ranking crops elephant preferences ($W=0.758$) indicating a strong agreement among respondents. Crop losses recorded in Maize, Pigeon peas, Sorghum, Tomato and Paddy were 102; 56; 36; 34; and 28 acres respectively. The spearman's rank test for the relationship between elephant preferences and size of damaged crops (Coefficient $\rho = 0.066$, $p=0.1866$), indicating a strong relationship between preferred crops and damaged crops. Understanding elephant crop preferences is crucial towards crafting effective mitigation measures to promote co-existence between humans and elephants.

Key Words: Crop damage, Crop preference, Crop raiding, Human–elephant conflict, Mitigation

INTRODUCTION

Human–elephant conflict (HEC) is a pervasive challenge in human–elephant dominated landscapes with expanding human settlements, infrastructural development and agricultural practices (Hoare, 2015; Songhurst *et al.*, 2016; Hariohay *et al.*, 2020). Crop raiding by elephants forms the most widespread and economically damaging dimension of all human–wildlife conflicts (Graham *et al.*, 2010; Hariohay *et al.*, 2020). When agricultural fields are within or near their traditional movement routes, elephants often exploit crops as concentrated, energy-dense food sources, leading to substantial losses for farmers and heightened tensions between communities and

conservation authorities (Rode *et al.*, 2006; Chama *et al.*, 2025). Studies in Uganda and Zambia have demonstrated that elephants exhibit selective feeding habits on specific crop types which are often targeted because of their high sugar, starch, and moisture content, providing greater nutritional payoff compared to natural wild browses (Sitati *et al.*, 2005; Chama *et al.*, 2025). Elephants also prefer crops that are easy to uproot or consume rapidly, especially those planted in contiguous plots near protected areas' boundaries or along migratory corridors (Ortega, and Eggert, 2004). However, seasonal availability of wild forages strongly influences crop raiding frequency and timing with raids peaking during the dry season when natural vegetation is scarce or during crop maturation stages when palatability is

highest (Hoare, 2015). In the MNP context, elephant crop raiding has been reported as the most serious human–wildlife conflicts affecting adjacent communities (Gunn *et al.*, 2013; Mayengo *et al.*, 2017). Farmers cultivate diverse crops, but not all are equally susceptible to elephant damage. Yet, comparative data on the magnitude of losses across crop types underlying these preferences remain scantily documented. Understanding such crop raiding specific patterns is vital for developing targeted and cost-effective mitigation strategies. Understanding the crop preferences of African elephants around MNP is crucial for several interrelated motives including; identifying which crops are most susceptible to elephant damage enables targeted and cost-effective interventions. This study is expected to provide insights into the human dimensions of HEC and the trade-offs between agricultural practices and wildlife conservation, by providing a comparative and crop-specific assessment of elephant crop damage around MNP and contribute to more sustainable human–elephant coexistence mitigation strategies.

MATERIAL AND METHOD

Study area

Mikumi National Park lies in south-eastern Tanzania in Morogoro Region (Fig. 1), forming part of the complex Selous–Nyerere-Mikumi ecosystem, with Bimodal climate whose wet season spans from December -May and a pronounced dry season from June to November (Gunn *et al.*, 2013). Communities bordering Mikumi are predominantly smallholder farmers practicing rain-fed subsistence agriculture supplemented by petty trade, casual labor and small livestock (Mayengo *et al.*, 2017). Common food and cash crops reported in the literature and government baseline assessments include maize, paddy rice (in lowland/wet plots), sorghum, cassava, beans, groundnuts, and some horticultural/cash species like sesame and sunflower (Kahimba *et al.*, 2015).

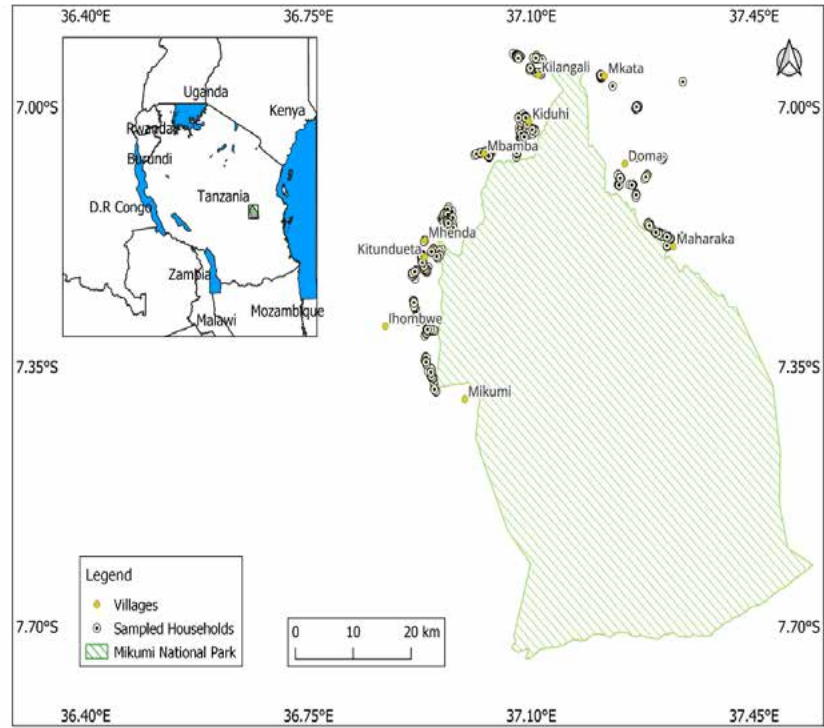


Figure 1: Locations of sampled households in ten surveyed villages adjacent to Mikumi National Park (July-August 2023)

Study design

The study employed a cross-sectional observation design, using household questionnaire surveys during wet season of 2023 to 405 households from 10 villages from July-August 2023 immediately after the main cropping season when farmers could recall the impacts of elephant crop-raiding. Villages were purposively selected considering the history of presence of human-elephant conflicts, accessibility, and directly sharing borders with MNP. Similar study designs have been used in Tanzania, Botswana and Cambodia to understand the extent of crop loss and the community perceptions of communities towards crop preferences by elephants (Webber *et al.*, 2011; Hariohay *et al.*, 2020; Matsika *et al.*, 2023). The sample size was reached out based on Cochran's formula (1977); $= (Z^2 * P * Q) / e^2$ (Uarkarn *et al.*, 2021). Where N_0 is the sample size, Z is the value corresponding to the required confidence level ($Z=1.96$ at 95% confidence), P is the estimated proportion of the population (0.5) to maximize the sample size, ($Q = 1 - P$), e is the margin of error (0.05) for 5% margin error. Households' selection followed systematic sampling, by starting from nearest household at the village office, following sampling interval determined by dividing the number of households by 40 households per village. Questionnaires were translated into Kiswahili and pretested to 20 village leaders' households in Mikumi, Maharaka and Mhenda villages. Household heads spouses or another adult over 18 years old residing within household were interviewed.

Data collection methods

The study collected primary data on elephant damages on questionnaire surveys, gathered respondents and household demographics, and

respondents were asked to state if they had crop damage from elephants. Those who experienced crop damage from elephants were asked about the crops they cultivated and further estimate the size of the area damaged by elephants during the intended study period. Respondents were also asked to rank the crops in the order from the most destroyed to the least destroyed, a list of crops cultivated in the areas was made created for farmers to choose and rank based on the perceived levels of preferences by elephants. Elephant crop preference ranking was done on a 5-point Likert scale from 1 – Highly preferred by elephants to 5 – Not preferred at all.

Data analysis

Descriptive analysis was performed using R studio version 4.3.1 (McGuinness and Sheppard, 2021), to produce a visualization frequency chart for all crop damage and crop preferences by elephants. Shapiro-Wilk test was run to confirm the normality assumption of data distribution. The Spearman's Correlation test was used to test the relationship between the cumulative area damaged and elephant crops preferences. Kendall's coefficient test was run to confirm agreement of perceptions among respondents on elephant preferences.

RESULTS

Elephant crop damage

Figure 2 shows the quantitative estimates from household assessments of the crop fields destroyed by elephants in Mikumi villages. Maize suffered the most damage by 102.52 acres, followed by Pigeon peas 56.73 acres, Sorghum 36.25 acres, and Paddy 28.35 acres. Comparatively the least damage was observed within Mangoes and Papaya with 2.04 and 2.94 acres respectively, while Sesame did not record any damage. The spearman's coefficient relationship between the cumulative area damaged and elephant crops preferences ($p = 0.066$, $P = 0.187$), indicating a weak positive and statistically non-significant relationship.

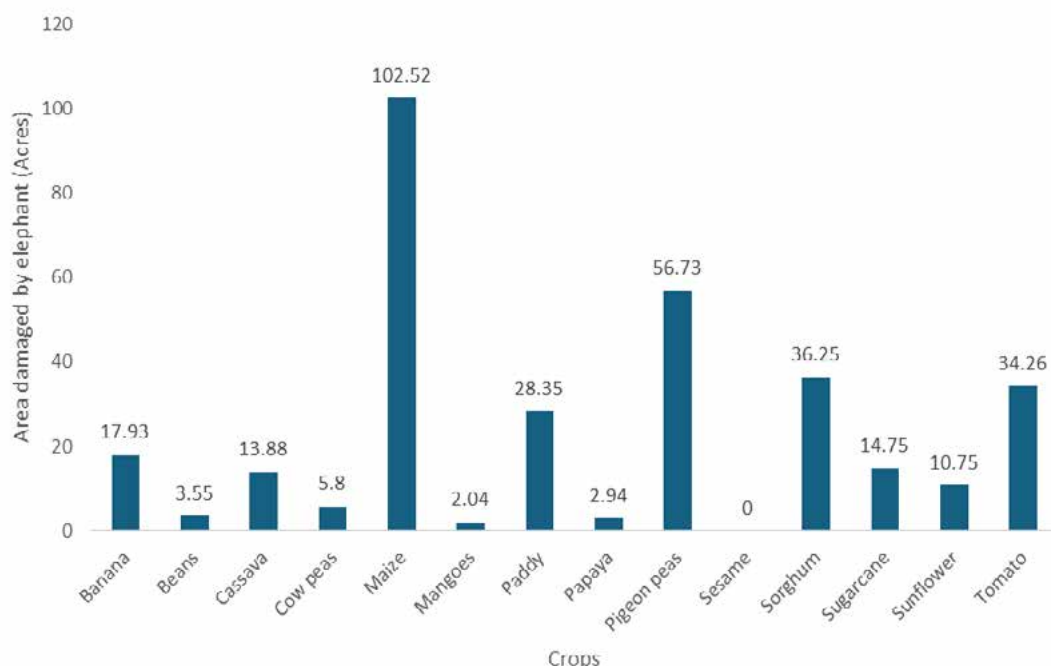


Figure 2: Extent Crop damages in villages adjacent to Mikumi National Park (July-August 2023)

Elephant crop preferences

Figure 3 shows farmers perceived crop preferences by elephants in Mikumi villages. Maize was the most highly preferred crop by elephants (Rank 1), followed by a group of crops—including paddy, tomato, banana, and millet that are classified as “preferred” (Rank 2). Bambaranuts, mango, and sugarcane, fall into the “moderately preferred” category (Rank 3). While sunflower, beans, peas, papaya, and cassava were perceived “less preferred” (Rank 4) by elephants. Sesame stands out between least preferred crop and/or not edible by elephants.

Kendall’s Coefficient of Concordance for assessing the level of agreement on crop preferences among respondents ($W = 0.758$, $X^2=3700$, $P = 0$), indicates a strong agreement among the participants on ranking of elephant crop preference suggesting strong consensus among themselves. The Chi-square test further confirms that this agreement is statistically significant, with a p-value of 0, suggesting that the observed agreement among respondents is very unlikely to be due to chance.

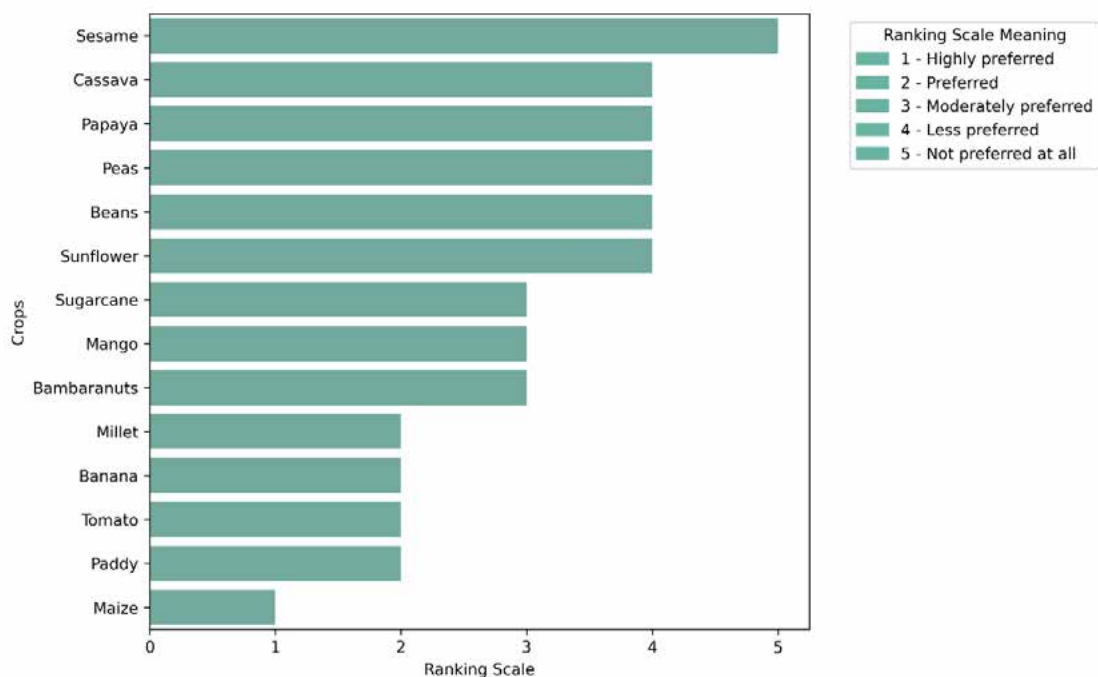


Figure 3: Farmers perceived elephant crop preferences in Mikumi villages.

DISCUSSION

Elephant crop damage

Observations suggest that Maize, Pigeon peas, Sorghum and Paddy fields are particularly vulnerable to elephant damage as preferred food source for elephants. Most households reported significant reductions in harvests and consequently household food outputs with substantial decline on households' welfare economic and social dimensions. The findings of this study are consistent with extensive literature across Africa showing that crop damage by elephants as the most visible and economically significant forms of human–elephant conflict (Hoare, 2015; Pozo *et al.*, 2017; Hariohay *et al.*, 2020). The scale of crop damage reported by farmers adjacent to MNP further aligns with findings from other East African sites, where households experience significant annual losses on financial and food stocks due to elephant raids (Sitati *et al.*, 2005; Mckenzie and Ahabyona, 2012).

Such **severity** of damage varies by season

and crop type raids often peak during the late dry season when natural forage is limited, and cultivated crops such as maize and paddy are at or near harvest maturity (Ortega and Eggert, 2004; Pozo *et al.*, 2017). Further studies in southern Tanzania show that annual elephant-related crop losses can reduce household incomes, significantly shortening households' food stocks with economic and social implications (Mabula, 2017; Mutanga *et al.*, 2022).

The elephant crop preference patterns underscore the role of forage scarcity and crop phenology as major triggers of elephant incursions into agricultural fields. Over time, repeated crop raiding has consequences for declining agricultural productivity, lowering household food outputs and financial gains which could be obtained from selling agricultural products.

Elephant crop preferences

Rating of elephant crop preferences demonstrates how most people share a similar understanding or experience about which crops

are preferred by elephants. Farmers' responses and field assessments indicated that elephants show distinct crop preferences that influence damage patterns. However, statistical results suggest that crop damage may not necessarily depend on preference alone, other factors such as crop location, availability, or seasonal needs could be influencing elephant behavior.

These findings are consistent with previous research in Uganda and Zambia indicating that elephants exhibit strong and selective preferences for certain crops, particularly those that are energy-rich, palatable, and easily accessible (Rode *et al.*, 2006; Chama *et al.*, 2025). In Villages around MNP as in other elephant ranges, crops such as maize and paddy were found to experience the most frequent and severe damage compared to others. These preferences are probably driven by the crops' high carbohydrate content, moisture availability, and low fiber structure, which make them nutritionally rewarding and easier to process compared to wild forages (Rode *et al.* 2006; Chama *et al.*, 2025).

Studies in Kenya's Laikipia (Graham *et al.*, 2010) and Botswana's Chobe District (Songhurst *et al.*, 2016) also report that elephants selectively target maize and banana fields, especially during the late dry season when natural forage is scarce and unpalatable. The observed preference for crops in Mikumi therefore aligns with a pan-African pattern of foraging optimization, to maximize energy intake with minimum effort and risk (Chiyo *et al.*, 2012). Elephant crop preferences could be shaped by seasonality, crop phenology, and farms' location with respect to elephant habitats. During the dry season, elephants are forced to extend their foraging ranges outside the core protected boundaries as water and natural forage decline (Ortega and Eggert, 2004; Hoare, 2015). Differential crop preference has profound implications for livelihoods and local food security. Households cultivating most preferred crops like maize, pigeon peas and paddy for both subsistence and market income bear disproportionate economic and nutritional losses (Mutanga *et al.*, 2022; Nyhus, 2016).

Elephant induced crop losses, this translates into direct seasonal food shortages, increased dependence on purchased food, and reduced resilience to other shocks such as drought. Recognizing and integrating these preferences into local land-use planning, crop selection, and mitigation strategies offering the most promising future toward sustainable human–elephant coexistence.

CONCLUSION AND RECOMMENDATIONS

This study demonstrates that African elephants crop preferences, and these preferences strongly shaped by seasonality, crop phenology, location with respect to elephant habitats and human factors. Crops vulnerability mirrors the possibility of the increased crop losses resulting in economic loss and food security which threatens farmers' local livelihoods and conservation objectives. The damage intensifies community resentment toward wildlife conservation and can potentially threaten long-term coexistence between humans and elephants if unmanaged. We therefore recommend promotion and encouragement to diversify and integrate less-palatable crops such as chili, ginger, citrus, sunflower, and sesame around field perimeters or buffer strips and/or prioritizing high-risk areas for such deterrent measures alongside adjusting farming calendars. This would reduce hostility toward conservation efforts and enhance local cooperation with wildlife authorities and promote human–elephant coexistence.

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EXPLORING LOCAL BELIEFS, ATTITUDES, AND PRACTICES TOWARD THE WHITE-BACKED VULTURES IN NORTHERN TANZANIA: IMPLICATIONS FOR CONSERVATION AND TOURISM

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ABSTRACT

This study examined local attitudes, beliefs, and practices regarding the White-backed Vulture (WBV) in selected villages of Loliondo and Sale Divisions in Northern Tanzania, using a mixed-methods approach. Data were collected through household surveys and key informant interviews. The findings indicate that respondents generally held positive beliefs and attitudes toward the species, recognising its ecological, economic, and cultural significance. Although most respondents acknowledged the importance of the species, they also reported a perceived population decline, attributing it to factors such as habitat loss linked to climate change, human activities, and land-use changes. Cultural practices involving the killing of vultures for traditional medicine, rituals, and superstition were largely dismissed as major causes of decline. Importantly, there was growing interest in community participation, particularly in conservation initiatives and the development of bird-watching tourism projects. These findings underscore the importance of engaging local communities as active partners in White-backed Vulture conservation and in tourism planning to enhance local livelihoods.

Keywords: Attitudes and practices, birdwatching tourism, conservation, traditional beliefs, White-backed vulture

INTRODUCTION

Countries rich in natural resources, including Tanzania, often face difficulties in balancing biodiversity conservation with the livelihood needs of local communities (Cortés-Avizanda *et al.*, 2018). This challenge arises from conflicting interests, particularly when critical community values, such as beliefs, attitudes, and indigenous knowledge, are ignored in conservation programmes (Kideghesho *et al.*, 2007; Martín-López & Montes, 2015). Many communities, especially in developing countries, rely heavily on natural resources for their survival, while conservationists seek to protect these resources (Persha *et al.*, 2010). This tension largely stems from conservationists' focus on saving wildlife species with little or no regard for nearby communities. Consequently, neglecting these human factors has led many communities to

adopt practices that harm biodiversity (Bennett *et al.*, 2016).

In order to attain sustainable natural resource management, the importance of integrating people's needs into conservation has received significant attention in conservation research and practices over the recent decades (Kideghesho *et al.*, 2007). As Cortés-Avizanda *et al.* (2018) argue, conservation is, in reality, as much about people as it is about species survival. Accordingly, a growing body of research supports the use of traditional beliefs and attitudinal studies to understand people's interests, goals, and motivations towards conservation, which in turn helps to inform the design of inclusive and new interventions. (Kideghesho *et al.*, 2007). Such an integration has therefore become a critical priority for policymakers, conservation practitioners, and

scholars (Kideghesho *et al.*, 2007; Lyamuya *et al.*, 2014; Rastegar, 2019). One promising approach is to incorporate non-consumptive initiatives, such as nature-based tourism, in conservation programmes to enable communities participate in conservation while also deriving livelihood benefits (Igoe & Croucher, 2017; Ouled-cheikh *et al.*, 2023; Rastegar, 2019). To emphasise this, other research demonstrates that conservation programmes actively involving local communities - through this approach - achieve more effective and equitable integration outcomes while those that disregard local communities frequently fail to meet their objectives (Berkes, 2017). Given this, many studies demonstrate that traditional beliefs, attitudes, and practices play a vital role in fostering meaningful community engagement in such initiatives (Mohamad *et al.*, 2012; Thi Khanh & Phong, 2020). Therefore, understanding local beliefs, attitudes, and practical aspects of those communities is essential, as it reveals how communities perceive, value, and interact with their natural environment and helps identify opportunities for collaboration in designing sustainable conservation programmes.

Framing the problem statement, research objectives, and research questions

Tanzania, one of the most biodiverse countries in sub-Saharan Africa, supports a wide variety of biodiversity, including a remarkable array of bird species. The northern region, in particular, stands out for its exceptional biological and cultural richness. This diversity provides opportunities for ecological recreation and nature-based tourism, which contribute to conservation and human development through economic, psychological, and intellectual benefits. Birds in general play a vital role in ecosystem functioning and serve as important subjects for tourism, scientific research, advancing ornithological knowledge, and informing evidence-based conservation policies.

Among many bird species of Tanzania, the White-backed Vulture (WBV) has drawn the interest of many researchers due to its rapidly declining population. Various studies have documented a sharp decline in WBV populations, driven by a combination of human and ecological factors, some of which remain poorly understood, leading to the classification of these birds as an endangered species (Ibrahim *et al.*, 2023; Safford *et al.*, 2019; Shiferaw *et al.*, 2022). Taking the perspective of the Sub-Saharan Africa, studies report that between 2017 and 2018 alone, over 387 vultures of various species were found dead in Kenya, Mozambique, Zambia, and Tanzania due to poisoning (Safford *et al.*, 2019). Other research highlights that traditional beliefs and community practices, including unsustainable hunting for consumption and the use of vulture body parts for rituals and superstitions, are also contributors to the species' decline (Agunbiade *et al.*, 2024; Murn *et al.*, 2016; Ogada & Buij, 2011). Empirical evidence indicate that in Africa, approximately 60% of vulture deaths are linked to poisoning, 29% to belief-driven killings, and the remaining 11% to factors such as illegal trade, bush-meat hunting, habitat alteration, and collisions with high-tension power lines (Mfilinge, 2023).

In northern Tanzania, the decline of WBVs was reported by Mfilinge (2023) in the BirdLife International web page. Based on Kendall & Virani (Kendall & Virani, 2017), “loss of vultures in East Africa is likely to have significant ecological and economic consequences, and thus conservation of this important scavenging assemblage is critical” (p.136). Despite this, in northern Tanzania, the causes of the decline of vultures remain underexplored, particularly at the community level (Agunbiade *et al.*, 2024). In this area, the dominant indigenous communities are the Maasai (Melubo, 2020), Wasonjo, Barbaig, Hadzabe, Iraqw, Chagga, Pare and many others. In today's world, the Maasai, and Wasonjo remain dominant tribes that possess strong cultural beliefs and attitudinal values known to influence conservation, enabling

people and wildlife to co-exist. These belief-based systems demonstrate how cultural norms can complement formal conservation efforts by promoting environmental stewardship in everyday life (Bennett *et al.*, 2016; Berkes, 2017; Parvin, & Begum, 2025). They are often rooted in cultural rules and taboos that limit the extraction of resources from sacred places, thereby preserving the ecological integrity of those areas where formal regulations may be weak, non-inclusive, or non-existent (Parvin, & Begum, 2025). For example, sacred sites in some parts of northern Africa have been found to support higher biodiversity and better structural integrity than nearby non-sacred areas because spiritual beliefs restrict harvesting of biodiversity (Hounkpati, 2023; Sinthumule, 2024). However, in northern Tanzania, factors associated with voluntary socio-cultural transformations, religious influences, and modernisation are increasingly undermining these customary mechanisms, eroding adherence to traditional norms and weakening their role in biodiversity protection (Darko & Issifu, 2015; Melubo, 2020; Western *et al.*, 2019). This erosion poses significant threats to local ecosystems essential for the survival of WBVs. The decline of WBVs in Tanzania may indicate a failure of both formal and traditional mechanisms to effectively foster conservation. This situation suggests the emergence of new human–nature relationships that are generally less favourable to the survival of this bird species. Notably, no research in this area has examined the extent to which community beliefs, attitudes, and practices contribute to the survival or mortality of WBVs. This knowledge gap presents significant ecological, economic, and conservation challenges, particularly if new conservation interventions that incorporate local communities are to be designed. Understanding these factors is therefore essential for developing context-specific, community-based conservation programmes, such as bird-watching tourism, that promote both community livelihoods and vulture survival (Virani *et al.*, 2010).

Birdwatching is increasingly being recognized as a non-consumptive approach that sustainably promotes the conservation of avifauna while providing socio-economic benefits to local communities (Liu *et al.* 2021; Padilla-Delgado *et al.* 2025; Randler (2023). When developed as community-based initiatives, bird-watching programs can create both financial and social incentives for long-term conservation (Hall, 2010; Mbaiwa & Kolawole, 2013). Such initiatives serve as a link between conservation and livelihoods by encouraging local communities to protect bird species from which they derive direct or indirect benefits. Evidence from Namibia and Nepal, for example, shows that community-based tourism has supported wildlife recovery and improved local livelihoods simultaneously (Jones *et al.*, 2015). Implementing similar models in northern Tanzania, focusing on WBVs, could significantly contribute to both species conservation and socio-economic development. Although many attitudinal studies have been conducted in northern Tanzania, most do not focus on WBVs; instead, they focus on carnivores (e.g. Lyamuya *et al.*, 2014; Masenga *et al.*, 2019). Accordingly, this study was conducted in northern Tanzania, a region rich in biodiversity, where the Maasai and Sonjo remain dominant communities with strong cultural traditions yet are relatively understudied regarding local perceptions of WBVs.

The purpose of this study was twofold, first to examine traditional beliefs, attitudes, and practices related to the conservation of the White-backed Vulture in northern Tanzania; and to draw implications for the development of new conservation strategies and the introduction of bird-watching tourism initiatives targeting WBVs. By addressing these core objectives, the study aimed to generate insights to inform policies and programs that integrate indigenous knowledge with tourism-based conservation strategies, thereby promoting sustainable coexistence between people and biodiversity. Specifically, the study seeks to answer the following research questions:

- i. What are the traditional beliefs that favour or limit the survival of White-backed Vultures?
- ii. What are the people's attitudes toward the current and future presence of White-backed Vultures in the study area?
- iii. What existing practices enhance or threaten the survival of White-backed Vultures?
- iv. How do community members perceive the introduction of bird-watching tourism aimed at promoting vulture conservation?

MATERIALS AND METHODS

Study area

The study was conducted in northern Tanzania, particularly in the Division of Loliondo and Sale in Ngorongoro District. Loliondo and Sale Divisions form an integral part of the greater Serengeti ecosystem, encompassing the Loliondo Game Controlled Area and several indigenous communities, primarily the Maasai, Wasonjo, and other ethnic groups. Other ethnic groups represent community members who migrated to Loliondo from other parts of the country and who are not guided by any traditional beliefs about nature. While the climate, vegetation, and landforms of the area have been described in previous studies (e.g., Bartels, 2016; Lyamuya *et al.*, 2014; Masenga *et al.*, 2019), ecologically the region serves as a critical migratory corridor for wildlife particularly wildebeest, zebra, and other ungulates that move seasonally between the Ngorongoro Conservation Area (NCA) and Serengeti National Park (SENAPA) in Tanzania and the Maasai Mara and Amboseli National Parks in Kenya (Masenga *et al.*, 2019).

Loliondo and Sale were chosen as the study site for several reasons. Historically, the region has supported key nesting habitats, such as the Gol Mountains, for the white-backed vulture (*Gyps us*) and Rüppell's griffon vulture (*Gyps rueppellii*), both now classified as endangered (Masenga *et al.*, 2019). It also provides habitat for endangered carnivores, including the wild dogs (*Lycaon pictus*), as well as threatened

species like the cheetah (*Acinonyx jubatus*) (Bartels, 2016; Masenga *et al.*, 2019), which sometimes prey on livestock, leading to retaliatory practices and increased human-wildlife conflicts, which may indirectly harm WBVs. The area also frequently experiences land-use conflicts and human-wildlife interactions that may exacerbate conservation challenges.

Research methods and data collection

This research adopted the mixed method approach, involving both qualitative and quantitative data. The data for this research were collected between 2023 and 2024 through a survey involving 186 household heads and 12 key informants selected by using a convenience, making a total of 198 respondents (i.e., $N = 198$). With the assistance of village leaders, the households were conveniently drawn from six purposively selected villages: Sendui ($n = 35$), Maalon ($n = 33$), Pinyinyi ($n = 31$), Ololosokwan ($n = 33$), Oloipiri ($n = 32$), and Arash ($n = 34$). Data collection followed established procedures described in earlier studies by Bethlehem (2009) and Christensen *et al* (2011). Respondents were selected based on convenience, age (18 years and above), and gender (ensuring that both males and females were represented). A questionnaire that combined a five-point Likert scale with open-ended questions was used to collect the data. The questionnaire focused mainly on capturing the respondents' attitudes, beliefs, and practices related to conservation and tourism.

Data analysis

The Stata software (version 18) was used to analyse quantitative data, while content analysis was applied to qualitative data. Content analysis helped to identify patterns in social phenomena, particularly where convergence of ideas was observed. Descriptive statistics, including frequencies and percentages, were calculated, and inferential statistics such as the Chi-square (χ^2) test and associated p-values were used to make comparisons of data between villages. Qualitative data were analysed thematically,

and percentages of responses were reported to reflect convergence of ideas and the frequency of similar responses among participants.

RESULTS AND DISCUSSION

Local beliefs

Table 1 presents a list of traditional beliefs that emerged during the interviews. Among the Maasai, the power of *Enkai/Engai* (God) as the creator of all living things is the dominant belief. Although some Maasai respondents acknowledged a lack of specific belief in recognising vultures as a cultural symbol in their area, other respondents indicated that a belief to *Enkai* (God) presents their respect to all living things and their habitats, including the vultures. This respect indirectly supports the survival of WBVs, and discourages the killing of them, or destroying their habitats. The respondents consistently indicated that harming wildlife, including birds, or destroying vegetation that forms important breeding sites for the birds a sinful and likely to invite divine punishment in the form of drought, disease, or famine. This belief positions God as the mediator of human-nature relations and promotes an important intrinsic positive moral behaviour that controls human practices towards bird species. Similar beliefs have been widely documented in other studies, with spiritual taboos and religious norms functioning as informal conservation institutions (Berkes, 2017; Dudley *et al.*, 2009). However, while this belief encourages wildlife protection in general, the absence of species-specific cultural recognition for vultures may weaken targeted protective actions, as general tolerance does not necessarily translate into

active conservation of less charismatic or threatened species such as WBVs (Dickman *et al.*, 2011).

A second dominant belief among the Maasai is that livestock, rather than wild animals, culturally define wealth, and that vultures play only a functional role in locating lost cattle. Interestingly, the cultural rejection of hunting and consumption of wild animals in favor of livestock indirectly benefits WBVs by reducing deliberate hunting and human disturbance (Homewood *et al.*, 2009; Western & Wright, 2019). However, such beliefs can indirectly influence conservation outcomes for WBVs, yet they do not inherently guarantee cultural protection on these species. When livestock predation attacks occur, pastoralists may resort to using poisons to control predators, with vultures becoming unintended victims of these practices (Ogada *et al.*, 2016; Manqele *et al.*, 2023). Research indicates that the majority of reported vulture deaths from poisoning arise from retaliatory or opportunistic measures linked to livestock protection (Ogada, 2016). Therefore, the perception of livestock as primary wealth, alongside the low economic or cultural value attributed to wildlife, may undermine targeted WBV conservation unless educational programs, eco-friendly conflict mitigation strategies, and incentive-based approaches are implemented to manage livestock–wildlife conflicts and reduce indirect harm such as poisoning (Mfilinge, 2023). Addressing these socio-cultural and livelihood dimensions is critical for fostering coexistence and mitigating threats to WBVs in pastoral landscapes.

Table 1: Community beliefs

Community	Traditional beliefs	Some key responses from interviewees
Maasai	Belief in God's power (<i>Enkai/Engai/Eng'ai</i>)	a) "God is the creator of our mother earth, where all creatures found on it, including forests, livestock, and wild animals, including the birds you are referring to". b) "I don't think we have specific beliefs about vultures, all what I know, God manifests himself to human beings through the natural resources we see and use". c) "In our community we believe that any act to kill any wild animal (including vultures) or cut trees is a disgraceful action that deserves God's punishment". w d) "We believe that consuming any wild animals is a disgrace to God". e) "God's punishments as a result of wrongdoing against any wild animal manifest through droughts, floods, tsetse flies, and diseases, which consequently result in famine and hunger affecting both people and wildlife". f) All misfortunes are a result of God's anger, which is caused by ill-treatment of natural resources, including trees and wildlife". g) God brings water, which is a source of life for all living things, including trees, big and small animals, and birds such as vultures, ostriches, and others.
	Belief in wealth from cattle for money (<i>ingohola/mpesai</i>) and livelihoods	a) "We believe that true wealth originates from livestock such as cattle, goats, and sheep and not from wild animals". b) "A family that eats meat from cattle, fish, goats or sheep in a meal or ceremonies symbolise wealth, while eating wild animals such as eland, antelopes, dik-dik, even fish, local chicken or wild birds is a curse and a person committing it is regarded as poor and an outcast in the Maasai community". "However, we use vultures to track lost livestock to determine whether it has already been preyed upon; if there are no signs of predation, we continue searching for the livestock".
Sonjo	Water and agriculture for survival	a) "In our community, water is considered a treasure and a source of life as it enhances agriculture, and all living things (including WBVs) depend on water for survival". b) "Agriculture is our main economic activity, which depends on rainwater availability." c) "Our cultural values respect all plants and animals, which, like human being, thrive when there is sufficient water".
Other communities	Belief in science	a) "Most of us have been brought up in traditional societies with the misconception that conserving wild animals is not an option, even though scientific evidence shows that it is essential, and we now believe in it". b) "Vultures play no big deal here apart from consuming in dead animals yet we don't harm them, and believe their survival is essential".

On the other hand, the Sonjo community believes in water and Agriculture as the foundations of life. This belief indirectly fosters an ecosystem-wide conservation ethic, even though vultures are not a primary focus of this community. Based on respondents, water is as an essential resource for agriculture, plants, animals, and human survival, encouraging conservation of landscapes that indirectly support vulture habitats. Such findings align with studies demonstrating that traditional ecological knowledge often promotes biodiversity conservation through recognition of ecosystem interdependence (Berkes, 2017). For WBVs, such values are ecologically significant, as they foster the survival of a wide range of species, thereby enabling reliable carcass availability for this scavenger. (Houston, 1985). Nevertheless, the adoption of agricultural intensification practices by the community may lead to habitat degradation, thereby affecting the survival of vultures (Millennium Ecosystem Assessment, 2005).

Other community members (the immigrants from other regions) expressed their belief in scientific understandings of conservation, recognising the ecological role of vultures as scavengers. This finding extends the widely acknowledged evidence that science can foster positive conservation attitudes among local communities (Inskip & Zimmermann, 2009; Jacobson *et al.*, 2015). However, the perception that vultures “play no big role” apart from helping to dispose of the dead animals reveals a persistent undervaluation of WBVs among the community members and this can be a key factor contributing to weak conservation of these bird species.

Community attitudes

The findings presented in Table 2, reveal generally positive community attitudes toward

the presence of African White-backed Vultures (AWBVs) across the surveyed villages. This represents an essential factor for the success of conservation initiatives targeting WBVs if introduced in the study area. Based on the analysis, a combined 74.3% of respondents expressed either very positive or somewhat positive attitudes, while only 5.5% reported negative perceptions and 20.7% remained neutral. These findings suggest a relatively high level of social tolerance toward vultures within the study area, which is a critical precondition for the success of community-based conservation strategies. Similar patterns of positive or tolerant attitudes toward scavenging birds is reported in other African rangeland systems, where communities recognise the ecological role of vultures in carcass disposal and disease regulation (Ogada *et al.*, 2012; Kendall *et al.*, 2017).

The village-level analysis shows that positive attitudes were consistently dominant, with the majority of respondents from Pinyinyi village exhibiting the highest level of favorable attitudes (90.3%) and no respondents expressing very negative views. This aligns with findings by Margalida and Colomer (2012), who reported that communities exposed to environmental education and awareness of ecosystem services tend to demonstrate higher acceptance of scavenger species. In contrast, there are relatively higher levels of negative attitudes reported in Maalon and Arash, although these remained low overall. Such localised negativity is consistent with studies suggesting that negative perceptions of vultures often arise from cultural symbolism, fear of livestock predation, or associations between vultures and death or misfortune (Asso *et al.*, 2024).

Table 2: Community attitudes towards WBVs

Village	Extremely positive	Somewhat positive	Neutral	Somewhat negative	Extremely negative
Sendui (<i>n</i> = 35)	4 (11.4%)	23 (65.7%)	7 (20.0%)	1 (2.9%)	0 (0.0%)
Maalon (<i>n</i> = 33)	2 (6.1%)	20 (60.6%)	9 (27.3%)	2 (6.1%)	1 (3.0%)
Pinyinyi (<i>n</i> = 31)	3 (9.7%)	25 (80.6%)	3 (9.7%)	0 (0.0%)	0 (0.0%)
Ololosokwan (<i>n</i> = 33)	1 (3.0%)	25 (75.8%)	5 (15.2%)	1 (3.0%)	1 (3.0%)
Oloipiro (<i>n</i> = 32)	1 (3.1%)	20 (62.5%)	9 (28.1%)	2 (6.2%)	0 (0.0%)
Arash (<i>n</i> = 34)	2 (5.9%)	21 (61.8%)	8 (23.5%)	3 (8.8%)	0 (0.0%)
Total (<i>n</i> = 198)	13 (6.6%)	134 (67.7%)	41 (20.7%)	9 (4.5%)	2 (1.0%)

Chi-square = 16.12, *df* = 20, *p* = 0.709 → *Not significant*

The chi-square test ($\chi^2 = 16.12$, *df* = 20, *p* = 0.709) demonstrates the absence of statistically significant variations in attitudes among villages, indicating a relatively homogeneous pattern of perceptions across the study landscape. This spatial uniformity suggests that shared socio-cultural and ecological contexts, rather than village-specific conditions, shape community attitudes in the region. Comparable homogeneity in attitudes has been reported in the pastoral landscapes literature, where livelihoods, belief systems, and patterns of wildlife interaction are broadly similar (Bennett, 2016). At the same time, this lack of variation contrasts with studies from South Asia, which have indicated sharp spatial differences in attitudes toward vultures, driven by religious practices and carcass management traditions (Pain *et al.*, 2003).

The substantial proportion of neutral responses (20.7%) is particularly noteworthy. From a conservation perspective, neutral responses represent a socially dynamic group whose attitudes may shift depending on information exposure and future expectations. They can be considered an unreliable indicator, as they suggest that the community may support conservation at some times but not exhibit conservation behavior at others. Nevertheless, previous research indicates that individuals with neutral or ambivalent attitudes toward wildlife are more responsive to targeted awareness campaigns and participatory

conservation initiatives (Waylen *et al.*, 2010). Therefore, while overt hostility toward WBVs appears limited in the study area, sustained conservation outcomes will likely depend on reinforcing existing positive perceptions and converting neutral attitudes into active support.

Overall, these findings are congruent with broader conservation literature emphasizing the importance of favourable community attitudes in enhancing the persistence of threatened vulture species (Ogada *et al.*, 2016). However, they contrast with reports from parts of Africa and Asia where vultures are heavily killed due to belief-based uses in traditional medicine and superstition practices (McKean *et al.*, 2013). This divergence suggests that the study area currently represents a comparatively supportive community for WBV conservation, opposing the thesis that the killing of WBVs in northern Tanzania is influenced by transformations of traditional values caused by religion, modernization, and other factors among the indigenous people. Nevertheless, proactive engagement remains essential to prevent the emergence or intensification of negative perceptions in the future.

Community practices

Table 3 presents the list of statements used to assess community practices toward WBVs and the corresponding analysis results, as measured using a five-dimensional Likert scale. The

findings reveal generally low levels of practices involving the direct exploitation of WBVs in the study area but highlight significant indirect anthropogenic threats linked to land-use practices such as agriculture, wildfires, or the intentional burning of vegetation to clear land or encourage pasture regrowth. An overwhelming majority of respondents strongly disagreed with the assertion that WBVs are hunted for human consumption (83.8%), traditional medicine (77.3%), or economic purposes (51.0%). These findings suggest that intentional, utilitarian

killing of WBVs is not socially normalised in the area. The literature (Ogada *et al.*, 2012; Virani *et al.*, 2010) reports similar patterns in other parts of sub-Saharan Africa, where vultures are more often threatened by incidental human activities than by direct subsistence hunting. This low level of agreement also aligns with Campbell (2014), who observed that many rural African communities do not traditionally regard vultures as a primary food or commercial resource.

Table 3: Community practices towards WBVs

Statements	Strongly agree (Freq, %)	Agree (Freq, %)	Don't know (Freq, %)	Disagree (Freq, %)	Strongly disagree (Freq, %)
In this area, WBVs are hunted for human consumption.	0 (0.0%)	0 (0.0%)	9 (4.5%)	23 (11.6%)	166 (83.8%)
In this area, WBVs are hunted for traditional medicines.	0 (0.0%)	17 (8.6%)	18 (9.1%)	10 (5.1%)	153 (77.3%)
In this area, WBVs are hunted for economic reasons.	0 (0.0%)	0 (0.0%)	18 (9.1%)	79 (39.9%)	101 (51.0%)
In this area, WBVs are hunted for traditional rituals.	0 (0.0%)	8 (4.0%)	34 (17.2%)	19 (9.6%)	137 (69.2%)
Slashing and burning fields for agriculture destroys habitats for WBVs	100 (50.5%)	15 (7.6%)	3 (1.5%)	25 (12.6%)	55 (27.8%)
Burning fields to encourage fresh pasture destroys habitats for WBVs	125 (63.1%)	15 (7.6%)	1 (0.5%)	40 (20.2%)	17 (8.6%)
Use of poison to kill predators also kills WBVs	55 (27.8%)	15 (7.6%)	78 (39.4%)	34 (17.2%)	16 (8.1%)
In this area, WBVs are hunted for superstition practices.	2 (1.0%)	11 (5.6%)	72 (36.4%)	55 (27.8%)	58 (29.3%)

However, uncertainty remains regarding ritual and superstition-related use of WBVs. Although 69.2% of respondents strongly disagreed that WBVs are hunted for traditional rituals, a notable proportion of respondents either agreed (4.0%), or reported not knowing (17.2%). Similarly, responses to superstition-related hunting were more mixed, with 36.4% indicating “don’t know” and 29.3% strongly disagreeing. These patterns suggest the persistence of secrecy or limited knowledge surrounding occult practices, which is consistent with findings by

McKean *et al* (2013), who documented that the use of vulture parts in traditional belief systems is often covert and socially sensitive, leading to under-reporting in surveys.

In contrast to the low reported levels of direct hunting, respondents strongly acknowledged the destructive effects of agricultural burning on WBV habitats. A majority strongly agreed that slashing and burning fields for agriculture destroys WBV habitats (50.5%), and an even higher proportion strongly agreed that burning

fields to promote the regrowth of fresh pasture is harmful to the birds (63.1%). These findings underscore that habitat degradation, rather than targeted killing, represents the more visible and socially recognised threat to WBVs in the area. This is consistent with global and regional evidence that habitat loss and landscape transformation are among the primary drivers of vulture population declines (Ogada *et al.*, 2016).

The use of poison to kill predators presents a more ambiguous trend. While 27.8% of respondents strongly agreed that poison use kills WBVs, a substantial proportion reported uncertainty (39.4%). This uncertainty suggests limited awareness of the secondary poisoning pathway by which scavenging vultures ingest toxic carcasses. Moreover, this uncertainty may reflect respondents' reluctance to disclose information about the indirect killing of vultures through the poisoning of carnivores, as many community members are aware that such practices are prohibited under conservation regulations. Similar knowledge gaps have been documented across East and Southern Africa, where poisoning remains a leading cause of vulture mortality despite low community awareness of its ecological impacts (Ogada, 2016). The data thus point to a critical need for targeted environmental education to improve understanding of indirect poisoning risks.

Overall, the findings indicate that local practices toward WBVs are characterised less by deliberate harm to vultures and more by indirect threats linked to land-use change and predator control. This supports the argument that conservation interventions in such landscapes should prioritise reducing habitat destruction, regulating burning practices, and promoting non-lethal predator management strategies (Virani *et al.*, 2010). Community-based awareness programs that address both overt and covert practices could play a vital role in improving the long-term survival prospects of WBVs.

Implication for conservation and tourism

The assessment of people's opinions regarding their willingness to fully engage in WBV conservation and associated bird-watching tourism activities was done by using five thematic issues: willingness to participate in conservation and bird-watching tourism programs; willingness to provide financial support for conservation and bird-watching tourism initiatives; willingness to devote substantial time each day to conservation activities; and perceptions of the potential financial benefits of conservation through bird-watching tourism.

Table 4 shows the results of this assessment. The findings reveal a complex picture of community willingness to engage in the conservation of WBVs and bird-watching tourism. A substantial majority of respondents (76.26%) strongly agreed that they are willing to participate in activities that conserve WBVs, while 100% indicated willingness to participate in bird-watching tourism activities. These results suggest that local communities generally hold positive attitudes toward both vulture conservation and nature-based tourism initiatives, consistent with studies showing that awareness and perceived ecological value often drive community support for conservation initiatives (Brooks, *et al.*, 2012; Mbaiwa, 2004).

In contrast, communities' willingness to provide financial support for conservation and tourism development was markedly low. Only 0.51% of respondents were willing to contribute financially to WBV conservation, and only 1.52%–7.58% supported financial investment in bird-watching tourism. The majority strongly disagreed with providing financial support (91.41% for WBV conservation and 44.95% for tourism development), suggesting that economic constraints or competing livelihood priorities may limit actionable engagement (Waylen *et al.*, 2010). These findings indicate a notable gap between attitudinal support and the capacity or willingness to provide financial backing for conservation initiatives.

Table 4: Respondents' opinions on their willingness to participate in conservation and tourism

Statement	Strongly agree	Agree	Neutral/ Don't know	Disagree	Strongly disagree
I am willing to participate in activities that conserve White-backed Vultures (WBVs).	151(76.26)	10(5.05)	14(7.07)	23(11.62)	0(0.0)
I am willing to support financially efforts that protect WBVs in my community.	0 (0.0)	1(0.51)	6(3.03)	10(5.05)	181(91.41)
I would volunteer time for conservation activities related to WBVs.	34(17.17)	8(4.04)	70(35.35)	56(28.28)	30(15.15)
I would report illegal killing or poisoning of vultures if I witnessed it.	137(69.19)	8(4.04)	34(17.17)	19(9.60)	0(0.0)
I am willing to participate in bird-watching tourism activities in my area.	198(100)	0(0.0)	0(0.0)	0(0.0)	0(0.0)
I would support financially the development of bird-watching tourism in my community.	3(1.52)	15(7.58)	50(25.25)	41(20.71)	89(44.95)
I believe that conserving WBVs can bring tourism benefits to my community.	12(6.606)	67(33.84)	88(44.44)	26(13.13)	5(2.53)

Similarly, willingness to volunteer time for conservation was moderate: only 17.17% strongly agreed and 4.04% agreed, while 35.35% were uncertain and 43.43% expressed disagreement. This moderate level of engagement highlights potential barriers to practical involvement, such as competing responsibilities, lack of incentives, or limited awareness of how to participate effectively (Sillero-Zubiri & Macdonald, 1998). Encouragingly, a majority of respondents (69.19%) indicated that they would report illegal killing or poisoning of vultures, suggesting awareness of legal and ecological imperatives and a readiness to uphold conservation norms at the community level (Ogada *et al.*, 2012).

Moreover, the findings reveal mixed perceptions regarding the potential benefits of WBV conservation through bird-watching tourism. While 33.84% agreed that participation in vulture conservation through tourism could generate significant benefits,

44.44% were uncertain. This variation may reflect limited understanding of ecotourism opportunities or a lack of visible examples demonstrating financial gains from tourism-based conservation (Mbaiwa, 2004). These results suggest that promoting community-based bird-watching tourism alongside vulture conservation, coupled with targeted awareness campaigns, could enhance both participation and financial support.

Overall, the findings indicate that while communities are generally supportive of WBV conservation and bird-watching tourism in principle, converting positive attitudes into tangible financial, voluntary, or practical contributions requires deliberate interventions. Conservation strategies should therefore focus on building capacity, demonstrating economic benefits, and providing incentives to bridge the gap between willingness and actionable engagement. Such integrated approaches are critical for ensuring both the ecological

sustainability of WBVs and the socio-economic benefits of community-based wildlife tourism.

CONCLUSION AND RECOMMENDATIONS

This study examined the relationship between community beliefs, attitudes, and practices and the decline of White-backed Vulture (WBV) populations within the broader context of regional population trends, as well as the implications for introducing targeted conservation programmes and birdwatching tourism initiatives. Contrary to the initial hypothesis, most respondents expressed beliefs, attitudes, and practices supportive of vulture conservation, suggesting that local value systems may facilitate rather than undermine conservation outcomes. Overall, the findings indicate the coexistence of traditional and scientific belief systems that currently promote passive coexistence with WBVs but do not necessarily guarantee active protection. The absence of strong species-specific cultural significance and the persistent risk of indirect poisoning remain critical threats to vulture survival. Consequently, belief systems that offer some degree of cultural tolerance toward vultures may simultaneously expose them to severe indirect risks especially when agricultural practices and livestock protection are prioritised.

Although many respondents were generally unwilling to bear direct financial costs for conservation, they expressed a strong willingness to participate in tourism-based initiatives, reflecting livelihood considerations and a preference for benefit-linked conservation strategies. However, the limited awareness among some respondents of the potential financial benefits of conservation through birdwatching tourism suggests the need for community-wide education prior to introducing such initiatives.

Similarly, the continued decline of vulture populations despite favorable community attitudes indicates that broader structural

factors, such as habitat loss driven by climate variability and land-use change, poisoning associated with human–carnivore conflict, weak law enforcement, and external development pressures, play a more decisive role. Accordingly, policymakers and conservation agencies should prioritise integrated conservation approaches that emphasise co-management, equitable benefit-sharing, and inclusive community-based tourism as pathways toward sustainable vulture conservation.

FUTURE RESEARCH DIRECTIONS

The low frequency of reported cases of direct killing of vultures for rituals, superstitions or traditional medicinal purposes, or indirect killing through carnivore poisoning, may reflect respondents' reluctance to disclose their involvement in such activities for fear of legal prosecution. This observation suggests the need for further critical anthropological research to examine how such practices shape traditional conservation beliefs and behaviours toward WBVs.

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LOCAL COMMUNITY COPING AND ADAPTATION STRATEGIES ON HUMAN-WILDLIFE CONFLICT AT KWAKUCHINJA WILDLIFE CORRIDOR, CONNECTING TARANGIRE-MANYARA NATIONAL PARKS

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ABSTRACT

Human–wildlife conflicts (HWC) remain a persistent and complex global conservation challenge, and a deeper understanding of the human dimensions of coping and adaptation is still needed. This study was conducted in the Kwakuchinja Wildlife Corridor to assess how local communities cope with and adapt to persistent human–wildlife conflict. It focused on three villages within the corridor: Mwada, Vilima Vitatu, and Mswakini. A mixed-methods approach was employed, combining literature reviews, interviews, and closed-ended questionnaires to collect data from 78 households, 6 key informants, and 18 focus group discussion participants. Communities in all three villages reported experiencing crop raiding, livestock predation, human injury, and property destruction. However, the level of conflict varied significantly among the three villages ($\chi^2 = 18.1$, $N = 78$, $df = 6$, $p = 0.006$). Despite these challenges, communities demonstrated resilience through a range of coping and adaptation strategies. These included organized patrols, government support, collaboration with stakeholders, the use of local knowledge of wildlife behavior, and the application of various conflict mitigation techniques. Such techniques included constructing predator-proof livestock enclosures (bomas), using chili bombs and chili fences, and employing scare tactics such as horns, Roman candles, high-intensity torches, and scarecrows. In addition, communities adapted by diversifying their income sources through tourism-related activities and small businesses, and by cultivating crops less palatable to wildlife, such as sunflowers, onions, ginger, and chilies. The Burunge Wildlife Management Area (WMA), which manages the corridor, supports these efforts through a rapid-response ranger team for dangerous animals such as elephants and lions, and by distributing dividends from WMA revenues to ten member villages. Overall, the combination of these strategies has enabled communities to cope with and adapt to HWC. Consequently, compared to the past decade, there is now a higher level of community tolerance toward wildlife, reflected in increased reporting, formal complaints, and the use of non-lethal deterrence methods rather than killing, poisoning, or threatening wildlife.

Keywords: Adaptation, Coping, Human-wildlife conflict, Wildlife corridor, Villages.

INTRODUCTION

Human-wildlife conflict (HWC) occurs when the needs and behaviour of wildlife affect negatively humans or when humans affect negatively the needs and behaviour of wildlife (Mekonen, 2020) causes and mitigations of human wildlife conflict, the coexistence between human and wildlife assessment

was conducted around Bale Mountains National Park. Data were collected by means of household questionnaires, focus group discussion, interview, field observation and secondary sources. The nature and extent of human wildlife conflict in the study area were profoundly impacted humans, wild animal and the environment through crop damage, habitat disturbance and destruction, livestock

predation, and killing of wildlife and human. The major causes of conflict manifested that agricultural expansion (30%). Apparently, HWC is becoming a worldwide and complex problem in conservation (Saj, Sicotte and Paterson, 2001; Nyahongo, 2007). This is particularly true for communities living adjacent to Protected Areas, who often bear the brunt of negative interactions with wild animals. According to the United Nations Environment Programme UNEP (2018), societies near protected areas experience persistent conflicts arising from competition with wildlife over limited resources and space (Mudimba, 2019). These conflicts manifest as crop damage, livestock depredation, and human injury or death (Expósito-Granados *et al.*, 2019) however, it is unclear how the research community here integrates social and ecological factors to examine human-carnivore relations. Therefore, strategies promoting human-carnivore coexistence are urgently needed. We conducted a systematic review on human-carnivore relations in the American West covering studies between 2000 and 2018. We first characterized human-carnivore relations across states of the American West. Second, we analyzed similarities and dissimilarities across states in terms of coexistence, tolerance, number of ecosystem services and conflicts mentioned in literature. Third, we used Bayesian modeling to quantify the effect of social and ecological factors influencing the scientific interest on coexistence, tolerance, ecosystem services and conflicts. Results revealed some underlying biases in human-carnivore relations research. Colorado and Montana were the states where the highest proportion of studies were conducted with bears and wolves the most studied species. Non-lethal management was the most common strategy to mitigate conflicts. Overall, conflicts with carnivores were much more frequently mentioned than benefits. We found similarities among Arizona, California, Utah, and New Mexico according to how coexistence, tolerance, services and conflicts are addressed in literature. We identified percentage of federal/private land, carnivore family, social actors, and management actions, as factors explaining how coexistence,

tolerance, conflicts and services are addressed in literature. We provide a roadmap to foster tolerance towards carnivores and successful coexistence strategies in the American West based on four main domains, (1. For example, in parts of Asia, communities near protected areas have reported losses of 10–15% of their agricultural net profit due to wildlife damage (Madhusudan and Sankaran, 2010). In Africa, countries like Tanzania, Botswana, Uganda, and Kenya, frequently reported series of HWC are that involving crop damage, property destruction, livestock depredation, as well as human attack (Kangwana, 1993; Newmark *et al.*, 1994; Hill, 1997). A wide range of species, from large carnivores and herbivores to smaller animals, can instigate these conflicts (Saj, Sicotte and Paterson, 2001; Rao *et al.*, 2002). It is common for wildlife damage to occur throughout the year (Ntalwila *et al.*, 2011), but often peaks during harvest seasons when crops are mature (Malima, Hoare and Blanc, 2005; Jackson *et al.*, 2008). The impact on household livelihoods varies depending on the security level at the time of the event (Mekonen, 2020) causes and mitigations of human wildlife conflict, the coexistence between human and wildlife assessment was conducted around Bale Mountains National Park. Data were collected by means of household questionnaires, focus group discussion, interview, field observation and secondary sources. The nature and extent of human wildlife conflict in the study area were profoundly impacted humans, wild animal and the environment through crop damage, habitat disturbance and destruction, livestock predation, and killing of wildlife and human. The major causes of conflict manifested that agricultural expansion (30%). It is important to note that these conflicts are not solely initiated by wildlife; human activities are also a major driver. Rapid human population growth increases the demand for resources such as grazing land and settlements (Kideghesho *et al.*, 2006). Leading to encroachment into wildlife habitats. This reduces available grazing land for wildlife and exacerbates negative interactions, particularly in critical wildlife corridors (Klein, 2013).

Wildlife corridors are passages that facilitate the movement of animals between protected areas (Kegamba *et al.*, 2023). Tanzania particularly, has sixty-one identified corridors connecting its protected area network. A notable example is Kwakuchinja wildlife corridor, which connects Tarangire and Lake Manyara National Park, facilitating wildlife movement and promoting gene flow (Lyimo, 2019; Lyimo *et al.*, 2023). However, this corridor is also inhabited by native communities, primarily the Maasai, Barbaig, Datoga, Mang'ati, and Sandawe, who practice pastoralism and agriculture (Kiffner *et al.*, 2015). These are Maasai, Barbaig, Datoga, Mang'ati, and Sandawe, just to mention a few (Msofe, Sheng and Lyimo, 2019). Growing human populations in these areas increase the demand for land, leading to further encroachment and intensifying interactions between people and wildlife (Hariohay and Røskft, 2015)

Human-wildlife interactions in the Kwakuchinja corridor pose significant threats to local livelihoods and safety (Kegamba *et al.*, 2023). In response, communities experiencing HWC have developed various coping and adaptation mechanisms (Chamba, 2018; Noga *et al.*, 2018). These strategies aim to promote coexistence and include methods such as chili fences, bright lights for scaring animals, and reinforced livestock enclosures (bomas). However, some communities also resort to less conservation-friendly methods like poisoning, spearing, or directly killing problem animals (Hoare, 2015). Despite these efforts, HWC continues to negatively affect local people, yet they show little inclination to abandon the area (Shaffer *et al.*, 2019). This underscores the need to explore how the communities of Mwada, Vilima Vitatu, and Mswakini villages cope with and adapt to persistent HWC.

Many HWC studies focus on quantifying the impacts of wildlife on communities and identifying the problematic species. However, the mechanisms through which local communities cope with and adapt to HWC

are less documented. This study addresses this knowledge gap by specifically assessing the coping and adaptation strategies of local communities in the Kwakuchinja wildlife corridor. The study aims to determine the magnitude of HWC in the area and explore the techniques communities employ for survival. The findings will enhance awareness of the losses communities face and can inform strategies to mitigate conflicts along protected area boundaries.

METHODOLOGY

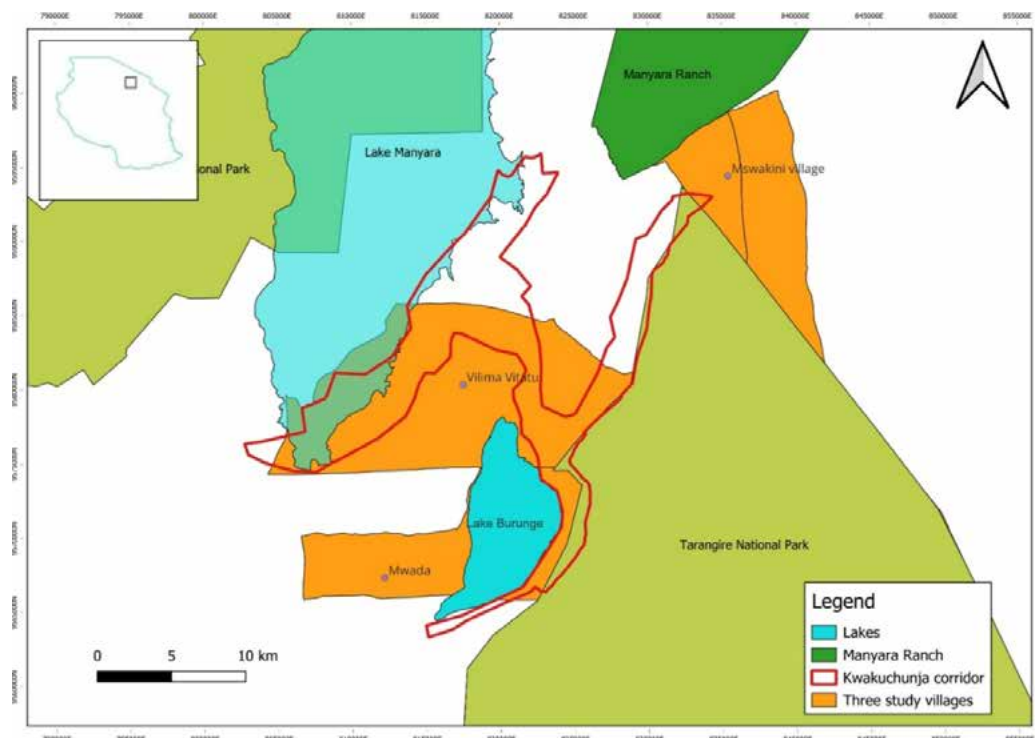
Description of the study area

The study was conducted in three villages Mwada, Vilima Vitatu, and Mswakini, bordering the Kwakuchinja wildlife corridor within the Tarangire-Manyara ecosystem. The villages are located between latitudes 3°40'S and 4°35'S and longitudes 35°45'E and 36°30'E, at altitudes ranging from 1000 to 1600 meters above sea level. The broader ecosystem covers approximately 35,000 including Tarangire National Park (2,600km²). The ecosystem is largely located in the southern Maasai steppe and includes Lolkisale and Simanjiro plain Games Controlled Area in the East, Mto wa Mbu Game Controlled Area to the north, Kwakuchinja open area in West and Mkungunero Game Reserve in the south (Galanti *et al.*, 2006).

The area is rich in biodiversity, including numerous bird species and mammals such as the (*Loxodonta africana*), Zebra (*Equus quagga*), African Buffalo (*Syncerus caffer*), Blue Wildebeest (*Connocahetes taurinus*), Lion (*Panthera leo*), Caracal (*Caracal caracal*), Spotted hyena (*Crocota Crocuta*) and Leopard (*Pantera pardus*) just to mention few. The area has array vegetations forming a wide range of habitats including Riparian woodland, *Vachellia tortilis*, wetlands and *Vachellia-Commiphora* woodland, Riverine grassland, Combretum-dalbergia woodland, *Vachellia drepanolobium* woodland, and kopjes.

Kwakuchinja wildlife corridor is a vital component in the Tarangire-Manyara ecosystem because it connects the Tarangire National Park and Lake Manyara National Park, and allowing

for wildlife movement. The corridor is also home to communities practicing pastoralism and agriculture (Map 1) (Kiffner *et al.*, 2015).



Map 12: Map shows Vilima Vitatu, Mwada, and Mswakini villages in Babati District.

Research design

This study employed a mixed-methods approach, combining qualitative and quantitative techniques. Qualitative data were collected through Key Informant Interviews (KII) and Focus Group Discussions (FGDs) to explore the nature of HWC. Quantitative data were gathered through a questionnaire survey to determine the extent and variation of the problem. A cross-sectional research design was used to assess the prevalence of the issue at a single point in time, without repeating measurements on the study population (Kothari, 2004).

Sample size

78 households from three villages were chosen purposively comprised of 26 households from Mwada, 26 households from Vilima Vitatu,

and 26 households from Mswakini. 24 key informants from the three villages were chosen purposively based on familiarity of an area and how long has a person stayed in the area. 6 key informants aired their views through KII, and 18 participants conducted FGD.

Data collection techniques

Questionnaire survey

The questionnaire was self-administered to the head or senior member of the family above 18 years old for each selected household. Questionnaires were prepared in English and translated into Swahili language. In the place where there is a language barrier, a translator for the local language was used. Closed and open-ended questions were used to allow respondents to provide their ideas and views (Kaiser, 2009).

Focus Group Discussion (FGD) and Key Informant Interview (KII)

FGD is the qualitative research method that involves a group of people who are brought together to discuss a particular issue of interest. The group involved 6 participants who were brought together in a comfortable and relaxed place to discuss a specific issue in each village. Participants were chosen based on their knowledge, experience, and people who were affected by the conflict and the discussion took a maximum of one hour (1 hour). Furthermore, a semi-structured interview was applied in this study. It involves a list of questions to cover about a specific issue whereby there is room for open-ended questions and follow-up questions to develop a better understanding of the issue and questions which help to answer the objectives of the research topic.

Data analysis

Data collected from the questionnaire were cleaned, coded, and imported to the Statistical Package for Social Science (SPSS) software version 25 for analysis. Thematic analysis was further used to analyze data collected from FGD and KII. Following Content analysis pattern, matching of themes was applied to segregate data into similar content and the pattern-matched data were coded and analyzed to obtain tables. Pearson Chi-square test was used to test the significant differences of HWC responses in Mwada, Vilima Vitatu, and Mswakini village.

RESULTS

Socio-demographic characteristics of respondents

A total of 78 (N=78) respondents were administered with a questionnaire in three

villages Mswakini, Mwada, and Vilima Vitatu with 33.3% (n=26) each village. Of all respondents, 61.5% (n=48) were males, while 38.5% (n=30) were females. Majority of respondents had an age class between 41-50 years with 37.2% (n=29), followed by 51-60 years 23.1% (n=18), 31-40 years 19.2% (n=15), over 60 years 16.7% (n=13), and lastly 18-30 years 3.8% (n=3). Agriculture was the primary economic activity practiced, whereby the majority of respondents 48.7% (n=38) were farmers, 26.9% (n=21) were agro-pastoralists, and 24.4% (n=19) were pastoralists. Furthermore, the results showed that about 66.7% (n=52) of respondents have stayed in their village for 31 and above years, followed by 14.1% (n=11) staying for 11-20 years, 12.8% (n=10) staying for 21-30 years, and 6.4% (n=5) staying for 6-10 years.

Magnitude of HWC

Crop raiding was reported by majority of respondents to be the major type of HWC in the area with 43.9% (n=18) Vilima Vitatu, 36.6% (n=15) Mwada, and 19.5% (n=8) Mswakini. Followed by livestock predation 68.4% (n=13) Mswakini, 26.3% (n=5) Mwada, and 5.3% (n=1) Vilima Vitatu. However, some of the respondents experienced both crop raiding and livestock predation, 41.2% (n=7) Vilima Vitatu, 29.4% (n=5) Mwada, and 29.4% (n=5) Mswakini, where else respondents who experienced both livestock predation and property destruction were all evidenced in Mwada 100% (n=1). The results depicted significant difference in the type of HWC incurred by a particular village ($\chi^2=18.1$, $N=78$, $df_{(1,6)}$, $P=0.006$). (Fig. 1)

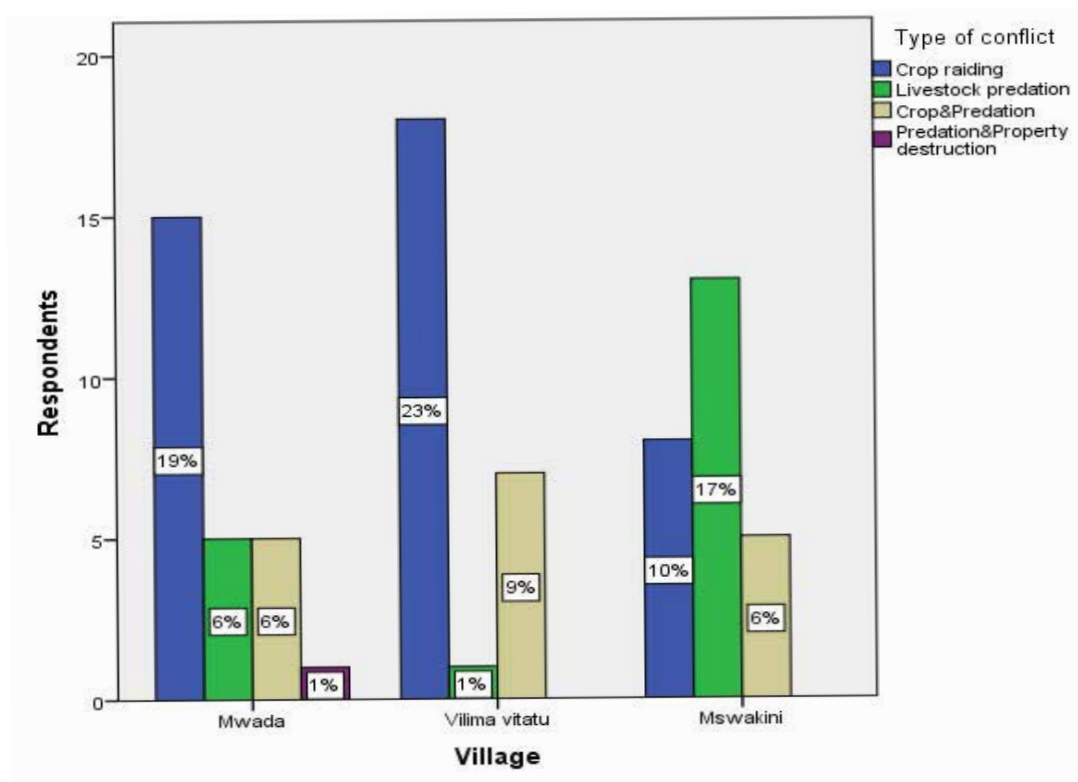


Figure 1: Types of HWC occurred in three villages.

Coping with HWC by local communities

It was evidenced that, community cooperation in preventing wildlife damage was a major coping mechanism, reported most in Mwada (22%, n=17), followed by Mswakini (14%, n=11), and Vilima Vitatu (9%, n=7). Adequate knowledge of wildlife behavior was also important, cited in Mswakini (14%, n=11), Vilima Vitatu (9%, n=7), and Mwada (5%, n=4). Diversifying income sources was another key strategy, notably in Vilima Vitatu

(12%, n=9), with lower reliance in Mwada and Mswakini (1% each, n=1). Resource availability (e.g., fertile land) was supportive in Mwada (5%, n=4) and Mswakini (1%, n=1). Government support (consolation) played a role in harmonizing coexistence, particularly in Vilima Vitatu (4%, n=3) and Mswakini (3%, n=2). Therefore, there was a significant variation in the local community coping with HWC that was reported by the three villages ($\chi^2=27.3$, $N=78$, $df_{(1,5)}$, $P=0.001$). (Fig. 2)

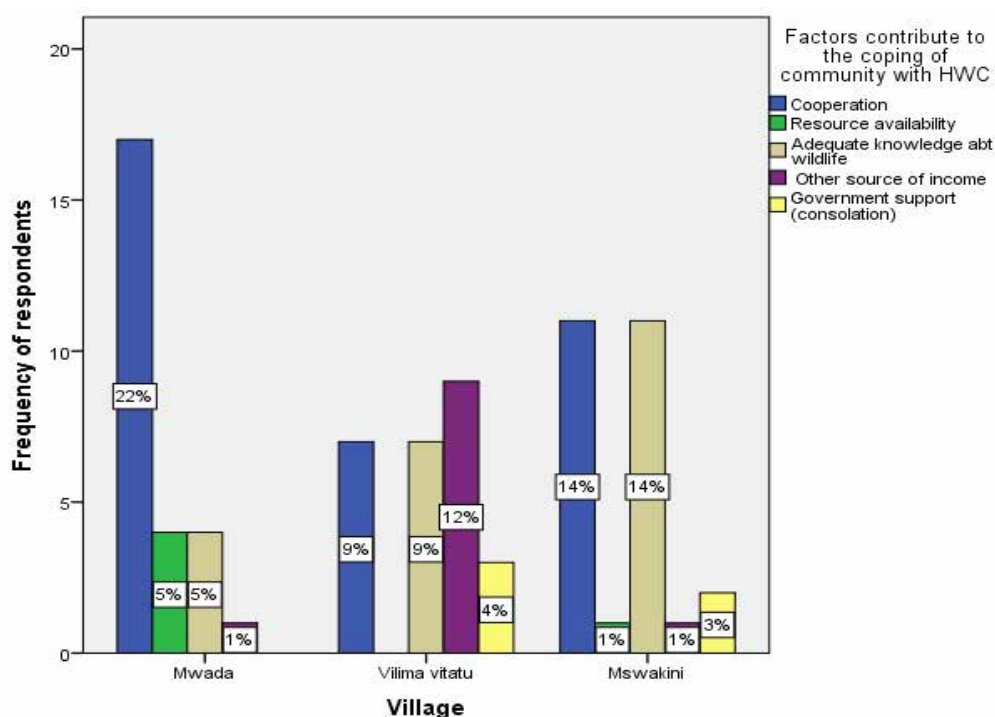


Figure 2: Factors contributing to the coping of local community with HWC.

Key informant interviews were conducted with six individuals: three Village Executive Officers (VEOs), one Burunge WMA Officer, one Tanzania People and Wildlife (TPW) officer, and one District Game Officer (DGO). Their insights are summarized in Table 1.

Table 1. Coping with HWC by local communities

THEME	RESPONDENT	QUOTE
Native	All VEO from three villages responded	“They have a lot of experience concerning HWC in their areas and they know the behaviors of wild animals” (VEO).
Dividend	DGO, VEO of all three villages, and Burunge WMA officer	“Burunge WMA provide dividends (50% is distributed to the villages) which are used in community development projects like schools, dispensaries, offices, road constructions, and students’ scholarships, so villagers know the benefit that comes from wildlife ” (WMA officer).
Employment	TPW officer	“Local communities are employed in WMA and other places around these areas which help them to improve their living standards” (TPW officer).
Partial damage	VEO from three villages	“Not all farms or livestock are destructed or killed all at the same time ” (VEO).
Low income	All VEO	“People have low income, so they have no other choices to move out because they cannot afford to migrate to other places” (VEO).

Freedom	Babati District Game Officer (DGO)	“People have the freedom to defend their properties such as crops, livestock, and house against problematic animals and they are allowed to kill when necessary, also have the freedom to shift into other means of income generation ” (DGO).
Availability of land	TPW officer and all VEO	“People get land for agriculture and pasture for their livestock” (VEO).
Consolation	Babati DGO and Burunge WMA officer	
Cooperation among community members	All VEO and TPW officers	“Government is providing money for the consolation of all affected people by wildlife and only those who are qualified by the consolation regulation which implies the sharing of problems with the community ” (DGO).
Problems do not occur to one person or in one place every day	All VEO and WMA officer	“Villagers tend to cooperate when they prevent problematic animals in their areas” (VEO).
Note that; Those coping mechanisms factors make these communities have a high tolerance level toward HWC or wildlife damage occurred and changing on the way they may react to wildlife damages because conservation is improved, so they are aware of conservation and fearing of laws that’s why end up reacting as follows,		“These problems occur to everyone found in these areas and does not choose the place to occur because of the size of areas ” (WMA officer).
Reporting	All VEO of three villages	“They provide information to the responsible authorities for the consolations and the assistance because they know about conservation” (VEO).
Complaining	WMA officers and VEOs	“Villagers complain much to the government compared to before where they just killed animals ” (VEO).
Scaring and chasing	All VEO and TPW officers	“Villagers scaring and chasing problematic animals away from their farms and bomas” (VEO).

Killing of animal	Burunge WMA officer	“Previously some people from some communities were used to killing problematic animals like a lion when reached to their bomas or around their areas but this depends on the behavior of a certain community (WMA officer).

Adaptation of HWC by local communities.

FGD results in three villages Mwada, Vilima vitatu, and Mswakini (Table 3).

Table 5. Mitigation measures used by local communities to adapt to HWC

THEME	SUMMARY	QUOTE
Mitigation measures for Mwada village	Different ways are used by the villagers to prevent wildlife damage such as for crop raiding may use touch, whistles, scaring, guarding, horn, chill fence, chill bomb, and planting unpalatable species And for livestock predation may use fire, construction of strong boma, solar-powered light, guarding, and horn.	“To reduce crop raiding villagers most use guarding, scaring, horn, touch, and planting unpalatable species, although chill bombs are used by village game scout only (VGS)....” (M2). “Many villagers use local boma and few manage to use strong boma and others use fire, touch, and guarding by using dogs...” (M5).
Mitigation measures for Vilima Vitatu village	Different ways are used by the villagers to prevent wildlife damage such as for crop raiding may use touch, whistles, chill fence, horn, scaring, guarding, and planting unpalatable species And for livestock predation villagers may use solar-powered lights, horns & whistles, use of fire, and the construction of boma.	“Villagers use touch, whistles to remove elephants and fences to prevent zebra...” (V3). “For the predators we use fire, touch, and construction of boma using thorn plant species of Vachellia and Dalbergia...” (V4). “Few villagers have managed to have strong boma using wire mesh but mostly use local methods...” (V1). “Planting unpalatable species does not help, you need to guard yourself and put things that scared animals like monkeys are very difficult animal...” (V6).
Mitigation measures for Mswakini village	Villagers use scaring & chasing, guarding, fencing, chill bomb, horn, and planting unpalatable species to prevent crop raiding And also, people use boma, guarding dogs, touching, and using fire.	“Thorn plant species are used to construct boma, the fire used to protect livestock by scaring predators, also dogs are used for guarding...” (S2). “Most local methods are used to protect our crops such as guarding, scaring the animal away, chill fence applied by few, and chill bomb can be applied by the village game scout only VGS...” (S3). “Also, planting unpalatable species help to reduce these conflict...” (S5).

DISCUSSION

Magnitude of HWC

The results of this study revealed that there was a significant variation in the type of conflicts that were experienced for three villages. Crop raiding was noted to be a big problem in all three villages starting from Vilima Vitatu, followed by Mwada, and lastly, Mswakini. Livestock predation was noted to be a big problem in Mswakini and Mwada villages. This variation is likely influenced by village location within the Kwakuchinja corridor, a critical pathway for wildlife movement in the Tarangire-Manyara ecosystem. Other contributing factors may include climate change and the spread of invasive species in the Burunge WMA, which can drive animals out of protected areas in search of food and water, increasing contact with communities. These conflicts directly threaten local livelihoods, which are heavily dependent on crop cultivation and livestock keeping. This is in line with findings by Mudimba and Tichaawa (2019), who reported wildlife attacks humans, destroying crops, and livestock predation in Zimbabwe.

Coping with human-wildlife conflict by local communities in the study area

The study found variation on how communities cope with HWC. The reason for this scenario could be due to the understanding of the local community towards wildlife conservation efforts and trust. Key factors enabling coping included inter-community cooperation in preventing damage, extensive native knowledge of wildlife behavior, and income diversification. However, access to government consolation payments after wildlife damage was inconsistent. While some respondents from Vilima Vitatu and Mswakini reported receiving consolation, many from Mwada and other villages did not, often citing bureaucratic hurdles that made the process seem futile. This inconsistency supports coping only for those who receive compensation and discourages others who feel their losses are unacknowledged.

This finding contrasts with recommendations by Nyhus (2016), who emphasized that timely consolation payments are crucial for increasing local tolerance towards wildlife.

Adaptation of HWC by local communities

The results indicated little variation in the mitigation measures used across the three villages to adapt to HWC. Communities predominantly relied on local methods such as fencing, chasing, scaring, using fire, guarding with dogs, and constructing traditional bomas. For instance, one participant noted, "Most local methods are used to protect our crops..." (S3), while another mentioned using "...fire, touch, and construction of boma using thorn plants..." (M5). This reliance on traditional methods may be due to resistance to change, an unwillingness to abandon familiar practices, or a lack of understanding and trust in newer techniques, often stemming from a perception that change is inherently risky. This observation is supported by Nyhus (2016), who noted that the perceived ability to manage risk and conflict often depends on individual actions rooted in established practices, which can reduce the feeling of risk and the perceived level of conflict.

CONCLUSION AND RECOMMENDATION

The local communities in all three villages demonstrated similar strategies for coping and adapting to HWC, with little variation in the mitigation measures employed and their perceived effectiveness. Many local methods were reported to be insufficiently effective, indicating a need for improvement and a shift away from over-reliance on traditional practices to enhance community safety. While all villages experienced both crop raiding and livestock predation, Vilima Vitatu and Mwada were more affected by crop raiding, whereas Mswakini faced greater challenges with livestock predation. The capacity to cope with HWC varied among the villages and was influenced by the community's level of understanding and

trust. A combination of factors including native status, dividends from the WMA, community cooperation, the partial nature of damage, investment in alternative income sources, and government consolation contributed to community resilience and a higher tolerance for wildlife damage. To strengthen this tolerance, management authorities should foster positive relationships with communities by increasing the tangible benefits derived from wildlife and ensuring timely consolation payments where needed.

RECOMMENDATIONS

This study assessed the coping and adaptation strategies of local communities in Mwada, Vilima Vitatu, and Mswakini villages towards HWC. Communities with higher tolerance levels are better equipped to manage wildlife damage calmly and effectively, which can also lead to improved livelihoods. Based on the findings and conclusions, the following recommendations are proposed for consideration and further research:

- i. Local communities should cooperatively protect their properties using a combination of proven local methods and the most effective modern techniques available in their areas to minimize HWC.
- ii. Management authorities should enhance cooperation with communities in managing HWC and ensure the consistent and timely provision of consolation fees for damages caused by wildlife. This will increase local tolerance and promote more conservation-friendly reactions to conflict, thereby improving overall conservation outcomes.
- iii. Relevant stakeholders should work to ensure that local communities receive adequate benefits from conservation initiatives (e.g., WMA dividends, employment). This will increase awareness of the value of wildlife and its implications for community development.

- iv. Authorities should provide education on the best available mitigation strategies to protect people and property, and to foster a positive relationship between communities and the management of adjacent protected areas.
- v. Further research should be conducted to understand the resistance to adopting modern HWC mitigation methods among local communities.

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SOCIO-ECONOMIC AND PSYCHOLOGICAL IMPACTS OF ELEPHANT INCURSION IN RHOTIA-KARATU, ARUSHA REGION, NORTHERN TANZANIA

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ABSTRACT

This study examined the socio-economic and psychological impacts of elephant incursions in Rhotia Ward, Karatu District, Arusha Region, northern Tanzania. The objectives were to assess socio-economic effects, evaluate psychological consequences of repeated elephant encounters, and identify community coping mechanisms related to elephant-associated risks. A cross-sectional design with a mixed-methods approach was employed. Data was collected from 141 respondents using household questionnaires and key informant interviews, applying purposive and simple random sampling techniques. Analysis involved descriptive statistics, thematic analysis, and chi-square tests to assess relationships among variables. Results showed significant differences in socio-economic impacts between Chemchem and Kilimatembo villages ($P = 0.031$). Crop damage was severe in both villages and resulted in government support. Psychological impacts did not differ significantly between villages ($P = 0.571$), suggesting comparable levels of stress, although anxiety was slightly higher in Kilimatembo (98.3%) than in Chemchem (91.6%). Coping mechanisms varied significantly between villages ($P = 0.001$). Beehive fences were used by 19.3% of households in Chemchem and 6.9% in Kilimatembo, indicating significant variation in mitigation effectiveness ($P = 0.023$). Overall mitigation effectiveness was higher in Chemchem (59.0%) than in Kilimatembo (39.7%). The study concludes that elephant incursions in Rhotia Ward cause substantial socio-economic and psychological challenges, including crop and property damage, financial losses, anxiety, fear of family safety, and sleep disturbances. It recommends prioritizing practical, community-based mitigation strategies to sustainably reduce human–elephant conflict.

Keywords: Elephant, incursion, socio-economic, psychological consequences, coping mechanisms.

INTRODUCTION

Human-elephant conflict (HEC) is defined as a situation where the needs and behavior of wildlife negatively impact human interests or where human activities adversely affect wildlife (Ridwan *et al.*, 2023). It commonly occurs when growing human populations encroach into wildlife habitats, leading to competition for limited resources such as food, water, and land (Western & Mose, 2021). Human-wildlife conflict (HWC) is a pressing issue that occurs globally, affecting both humans and wildlife (Naha *et al.*, 2019). Bürli *et al.* (2024) defines HWC as situations where human activities

and wildlife need overlap, leading to negative interactions that harm one or both parties. Elephants are frequently cited among the diverse range of wildlife involved due to their size, migratory behavior, and dietary needs (Smith, 2024). This conflict often manifests in elephant incursions into human settlements, resulting in damage to crops and property, financial losses, and even loss of human lives (Zhang *et al.*, 2024).

Globally, elephant incursions are amplified by habitat loss and climate change, which force elephants into human-dominated landscapes. In Southern Asia, human-elephant conflicts disrupt

livelihoods and lead to psychological distress, including heightened stress and anxiety (van de Water & Matteson, 2018). Similarly, frequent elephant raids on agricultural fields in India necessitate costly deterrent measures (Naha *et al.*, 2019). These conflicts highlight the urgent need for innovative solutions such as vegetative barriers and coexistence strategies that balance human needs with wildlife conservation (Bürli *et al.*, 2024).

Africa faces similar challenges where elephant incursion often intersects with agricultural expansion, deforestation, and food insecurity (Mayengo *et al.*, 2017). For example, in Uganda, farmers endure annual crop losses, which result in significant financial instability (Nansamba *et al.*, 2022; Rutta, 2023). Botswana provides another example, where psychological distress stemming from these conflicts leads to retaliatory killings of elephants, further complicating conservation efforts (Gunawansa *et al.*, 2023). Compensation schemes and community engagement programs have been proposed to foster tolerance and promote coexistence (Kansky & Kidd, 2024). Tanzania, renowned for its rich wildlife heritage, is particularly affected by HWC. In regions bordering national parks like the Selous Game Reserve and Ruaha National Park, elephants regularly raid farms, creating economic hardships for local communities (Hoffmeier-Karimi & Schute, 2015). Efforts to mitigate the conflict—ranging from beehive fences to chili deterrents—have shown varying degrees of success. Moreover, psychological impacts, including fear and trauma, are often reported among affected individuals, indicating a need for holistic approaches that address both socioeconomic and mental health concerns (Sanare *et al.*, 2022). Translocation programs and integrating local knowledge into mitigation strategies have been explored as potential solutions, though challenges remain in implementation and sustainability (Page, 2023).

Rhotia ward in Karatu district, northern Tanzania, is remarkably vulnerable to wildlife

invasion due to its location near wildlife corridors linking national parks. This area frequently experiences elephant incursions that destroy crops, leading to economic instability and psychological stress, particularly affecting women and children (Chang'a *et al.*, 2016). Current protective measures are expensive, and the absence of sustainable, community-driven strategies worsens the situation (Rija & Kideghesho, 2020). The combination of social and economic challenges and mental health effects underscores the urgent need for research and intervention. Gaining a deeper understanding of these impacts in Rhotia could guide the development of policies and conservation strategies that safeguard both human well-being and wildlife.

The socio-economic impacts of elephants' incursions are severe. Farmers in the region face reduced agricultural productivity, increased poverty, and limited livelihood opportunities. Psychological effects, including fears, anxiety, and trauma, are also prevalent, particularly among vulnerable groups such as women and children (Li & Graham, 2017). Even with mitigation strategies like beehive fences and chili-based deterrents, their effectiveness remains inconsistent, leaving communities vulnerable to ongoing conflicts (Mayengo *et al.*, 2017). The impact of unresolved human-elephant conflict reaches beyond personal households. Conservation efforts are compromised by the rise of revengeful killings of elephants driven by frustration and economic hardship. This risks the survival of the elephant population and disrupts the ecological balance they sustain (Rumiano *et al.*, 2024). Additionally, tensions between local communities and wildlife authorities hinder collaborative efforts to address the issue effectively (Malley & Gorenflo, 2023).

This research aimed to assess elephant incursions' socio-economic and psychological impact in Rhotia Karatu Arusha, Tanzania. Examining the extent of this impact and evaluating the effectiveness of existing mitigation measures, the study sought to identify

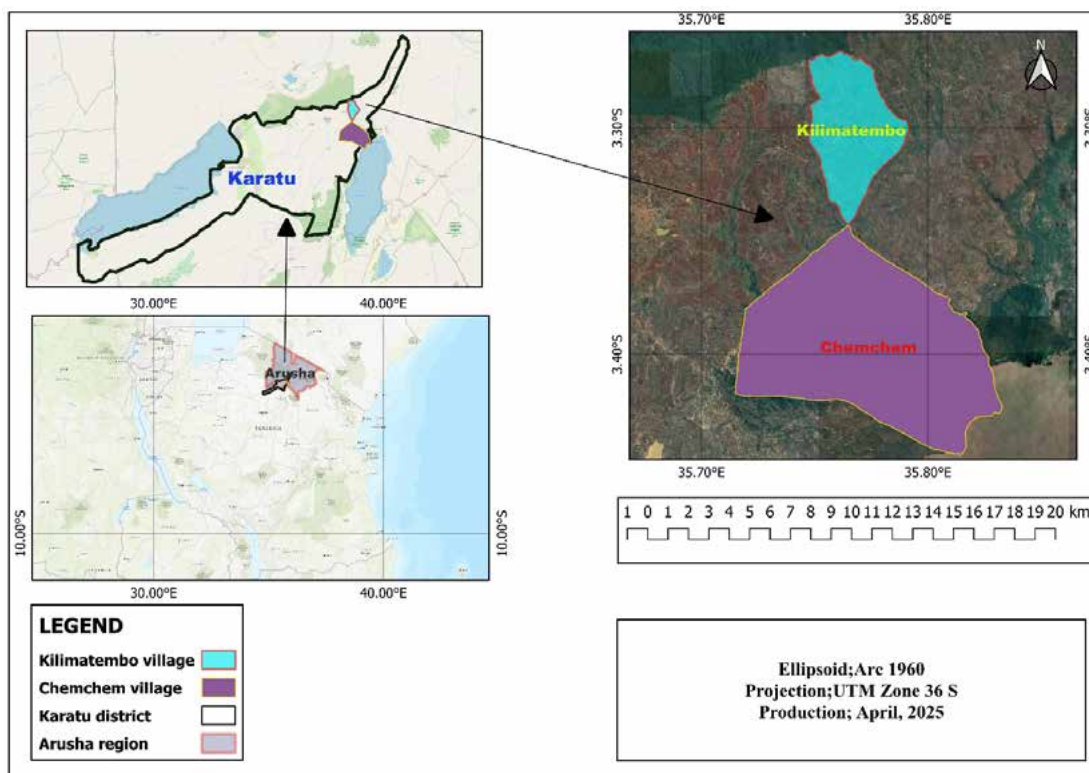
sustainable, community-based strategies for conflict management. The findings contribute to policies that promote coexistence, safeguard livelihood, and ensure the long-term survival of the region's human and elephant populations (Homewood *et al.*, 2022). The study assessed the socio-economic and psychological impacts of elephant incursions in the Rhotia ward, Karatu district, Arusha region, northern Tanzania. The study seeks to address three questions (i) What are the socio-economic impacts of elephant incursions? (ii) What are the psychological effects of recurring elephant encounters? and (iii) What are the community coping mechanisms of risk associated with elephant presence?

MATERIALS AND METHODS

Study area description

The study was conducted in the Rhotia ward, located within the Karatu district in the Arusha region of northern Tanzania (Map 1). Positioned along the escarpment of the East African Rift

Valley, Rhotia lies close to key conservation areas such as Ngorongoro Conservation Area and Lake Manyara National Park. These areas are connected by wildlife corridors that are critical for the movement of species like elephants. However, these corridors intersect with human settlements and farmlands, leading to frequent human-wildlife interactions. Rija and Kideghesho (2020) describe such regions as zones of intense human-wildlife interaction. This highlights the challenges communities living near protected areas face (Hariohay *et al.*, 2017; Hariohay *et al.*, 2020). Karatu district covers an area of about 3,207 square kilometers and boasts a variety of agroecological zones, including fertile highlands and semi-arid lowlands. Rhotia ward is in the midland zone, which enjoys moderate rainfall and fertile volcanic soils, making it ideal for agriculture. The district is mainly rural, with farming and livestock as the primary economic activities. Typical crops include maize, beans, and onions, while livestock farming involves cattle, goats, and sheep (LeFlore *et al.*, 2019).



Map 1: Map of Rhotia Ward in Karatu District, Northern Tanzania

Data collection

A cross-sectional design was applied, integrating qualitative and quantitative methods (Teherani *et al.*, 2015) to assess elephant incursions' socio-economic and psychological impacts in Rhotia, Karatu, and Arusha. Data was collected using questionnaires distributed to 141 households obtained using simple random sampling procedure in Chemchem and Kilimatembo villages to collect data on economic losses. Simple random sampling provided unbiased and representative data on economic losses, psychological effects, and coping strategies. Combining these techniques ensured a comprehensive understanding of the impacts from both experts and community-level perspectives (Sanare *et al.*, 2022). Sample size was calculated using an online sample size calculator with a 95% confidence level and 8% margin error using. The questionnaire included demographic, social, financial, psychological impacts, exposure to elephant encounters, and mitigation strategies. Additionally, questionnaires distributed to villagers collected measurable data on individual experiences, focusing on economic losses and psychological effects. This data included the frequency of elephant attacks and demographic information such as gender and age. This approach ensured a comprehensive understanding of the impacts from expert and community-level perspectives (Taherdoost, 2022). Purposive sampling was used to select key informants including local government officials, community elders, and conservation officers to gather detailed qualitative insights into the community's challenges, coping mechanisms, and broader strategies for addressing the issue. Key Informants' perspectives ensured a better understanding of existing mitigation strategies and any gaps in addressing the issue.

The study adhered to confidentiality and anonymity maintained by removing personal identifiers and anonymizing all data collected, ensuring that participants' privacy and safeguarded (Bryman, 2016). The study addressed sensitive topics including the

psychological effects of elephant incursions with care and respect to minimize stress. Participation was entirely voluntary, with individuals given the right to withdraw at any point without consequences. Additionally, cultural sensitivity was observed by engaging with local leaders and aligning the research process with the community norms and values, ensuring that cultural practices are respected (Bryman, 2016).

Data analysis

Quantitative data collected through a questionnaire was analyzed and coded using a statistical package for social science studies SPSS version 25. Qualitative data obtained from interviews with key informant leaders were analyzed thematically. This involved identifying recurring themes, patterns, and narratives within the responses to gain a deeper understanding of the socio-economic and psychological impacts faced by the community. The data were coded and categorized manually to ensure a systematic approach. Both descriptive and inferential statistics were applied to explore relationships and patterns.

Every objective had its tool for data analysis as an objective state to assess the socio-economic impacts of elephant incursions from this objective the dependent and independent variables are socio-economic impacts such as crop damage and proximity to elephant habitats respectively. The data was analyzed through a statistical package for social science studies SPSS. On objective to examine the psychological effects of elephants' incursions such as stress and fear levels were assessed as dependent variables. Independent variables like the frequency of elephant encounters, age, and gender were analyzed to determine how different groups are affected. Both descriptive and inferential statistics were applied using SPSS to explore relationships and patterns. On the last objective which states the coping mechanism of risk was explored qualitatively, factors such as education level, past experiences, and access to resources were

analyzed to understand how they influence coping strategies. Thematic analysis identified recurring themes and insights from qualitative data.

RESULTS

Demographic characteristics

Out of 141 respondents most were males. The main source of income was farming (97.2%)

while livestock keeping, and employment were in the minority (Table 1). The majority of respondents belonged to the age group 36-45 years followed by the age group 46-60 years, 26-35 years, 18-25 years, and the last age group above 60 years (Table 1). Most of the respondents were married (90.8%) while respondents who were single and divorced were in the minority (Table 1).

Table 6: Socio-demographic characteristics

Variable	Category	n	%
Gender	Male	107	75.9
	Female	34	24.1
Age	18-25 years	6	4.3
	26-35 years	18	12.8
	36-45 years	60	42.6
	46-60 years	55	39.0
	60 and above	2	1.4
Marital	Single	11	7.8
	Married	128	90.8
	Divorced	2	1.4
Income source	Farming	137	97.2
	Livestock	2	1.4
	Salary/Employment	2	1.4

Socio-Economic impacts of Elephant incursion

The study revealed a significant difference in the socio-economic impacts experienced by villages in Rhotia Ward as a result of elephant incursions (Fig. 1). Almost all households reported receiving government or NGO support, with Chemchem recording 97.6%, slightly higher than Kilimatembo at 96.6%. Property damage was reported at a relatively low level in Chemchem (20.5%) but was higher in Kilimatembo (36.2%). A reduction in household food supply was reported as high in Kilimatembo (51.7%) and markedly low in Chemchem (4.8%).

A substantial decline in family movements was observed in both villages. Crop damage was nearly universal, reported by 100.0% of households in Chemchem and 96.0% in Kilimatembo. Minor differences were noted in changes to farming practices, with 45.8% in Chemchem and 44.8% in Kilimatembo. Financial losses were more pronounced in Kilimatembo (20.7%) compared to Chemchem (16.9%). The Chi-square test ($\chi^2 = 2.091$, $df = 6$, $P = 0.031$) indicated a statistically significant difference in socio-economic impacts between the two villages, confirming variation in the severity of effects experienced.

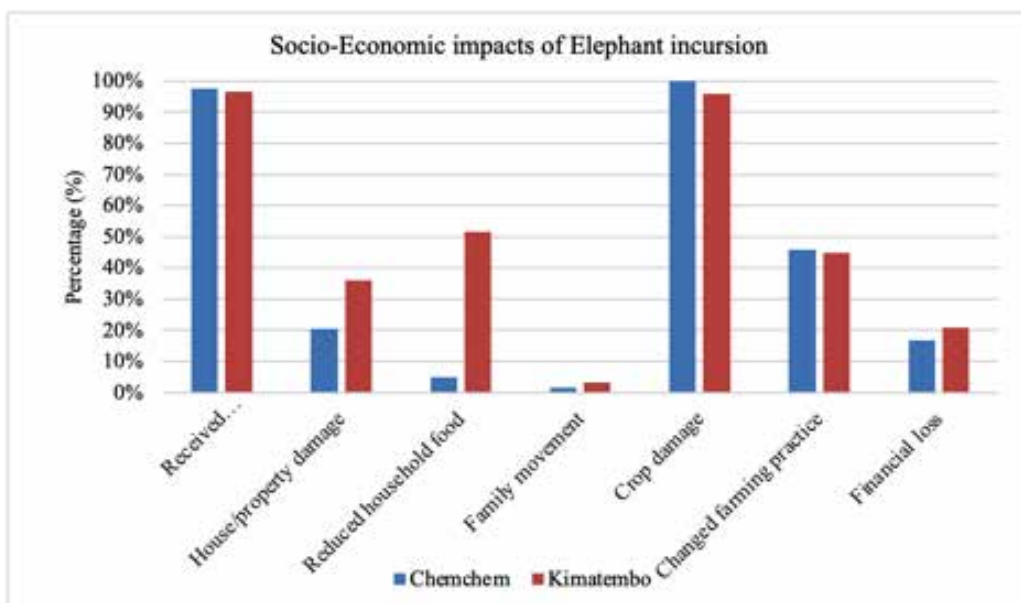


Figure 1: Social-economic impacts of elephant incursion

Psychological impacts due to elephant incursion

The study found no significant difference in psychological impacts resulting from elephant incursions, indicating similar levels of psychological stress across the villages ($\chi^2 = 8.761$, $df = 7$, $P = 0.571$). However, reported

anxiety was slightly higher in Kilimatembo (98.3%) than in Chemchem (91.6%). Concern for children's safety was high in both villages, reported by 96.4% of respondents in Chemchem and 95.1% in Kilimatembo. Sleep disturbances were substantially more prevalent in Kilimatembo (53.4%) compared to Chemchem (7.2%) (Fig. 2).

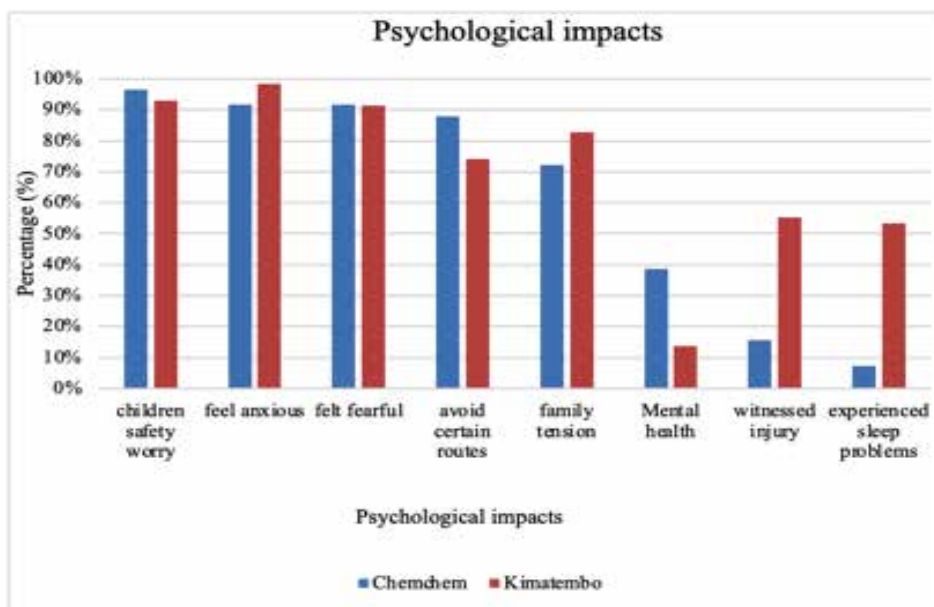


Figure 2: Psychological impacts due to elephant incursion

Community Coping Mechanisms and Responses to Elephant Incursions

Strategies to Deter Elephants

The Chi-square test indicated statistically significant differences between villages in the strategies used to deter elephants ($\chi^2 = 44.63$, $df = 5$, $P = 0.001$). Beehive fences were reported by 19.3% of households in Chemchem, compared to 6.9% in Kilimatembo. Drum use was reported by 13.3% of respondents in Chemchem

and was more common in Kilimatembo (29.3%). Chili fences were widely used in Chemchem (37.2%) but were rarely reported in Kilimatembo (1.7%). Scarecrows were used by 7.2% of households in Chemchem, whereas a higher proportion was reported in Kilimatembo (22.4%). Watchtowers were not used in Chemchem and were reported by 13.8% of respondents in Kilimatembo. Other deterrent methods were cited by 22.9% of households in Chemchem and 25.9% in Kilimatembo (Fig. 3).

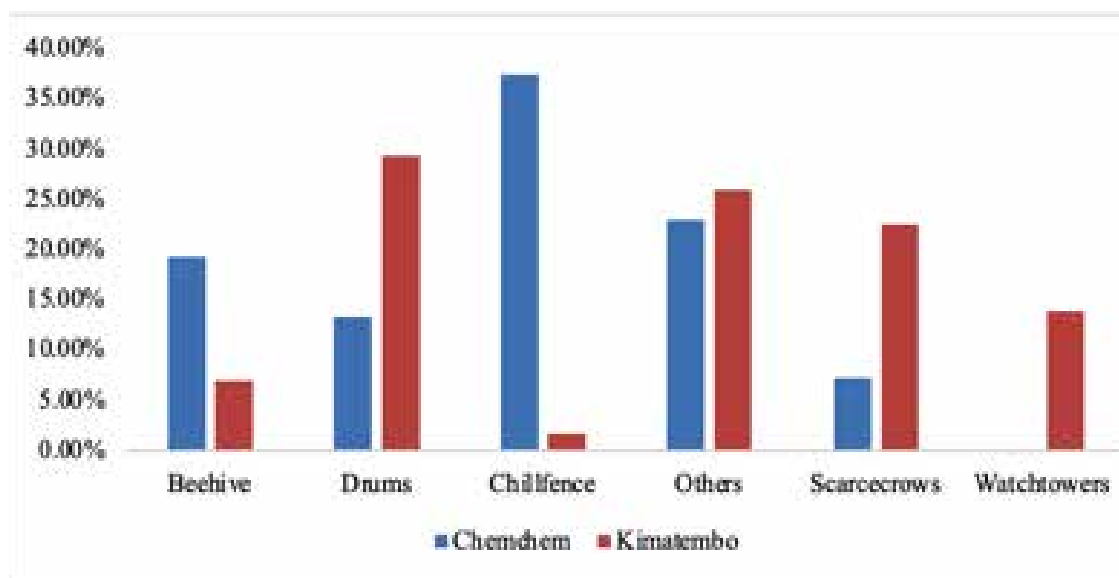


Figure 3: Strategies to deter elephants

Effectiveness of the methods

The study assessed the effectiveness of elephant deterrent methods in Chemchem and Kilimatembo villages. The findings showed that 59.0% of respondents in Chemchem considered the methods effective, compared to 39.7% in Kilimatembo. Conversely, 41.0% of households in Chemchem and 60.3% in

Kilimatembo reported the methods to be only somewhat effective (Fig. 4). The difference in perceived effectiveness between the two villages was statistically significant ($\chi^2 = 5.132$, $df = 1$, $P = 0.023$).

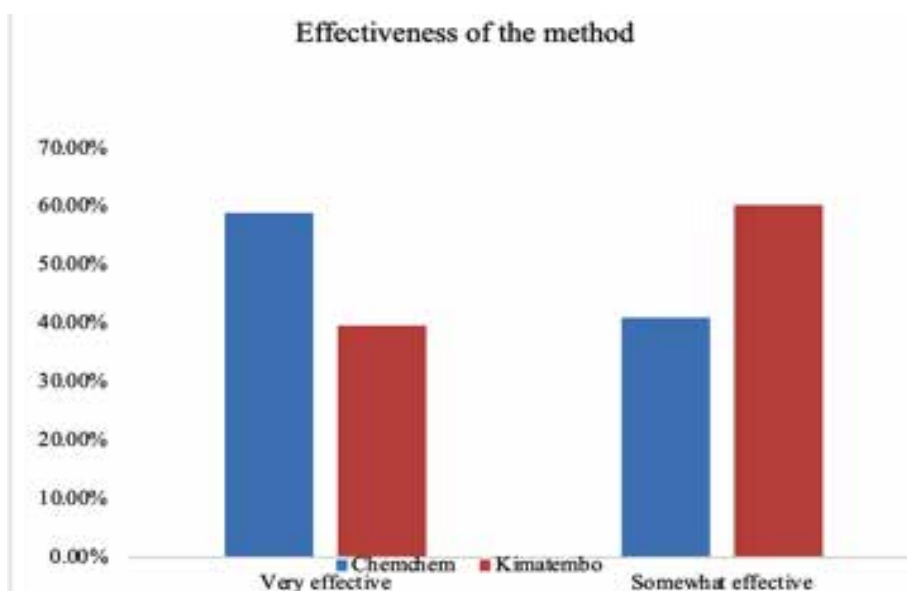


Figure 4: Effectiveness of the method

DISCUSSION

Demographic characteristics

The study findings on gender distribution revealed that the majority of respondents were male, reflecting cultural gender roles within local Tanzanian agricultural communities, where men are more actively involved in field-based livelihoods and are more directly exposed to wildlife threats. This finding is consistent with previous studies (Kansky & Kidd, 2024; Rutta, 2023), which emphasize that gender is a key factor shaping individual experiences and deterrence strategies in human–wildlife conflict areas.

The results further indicated that most respondents were middle-aged, a life stage commonly associated with long-term engagement in land-based livelihoods and a strong reliance on agricultural productivity. Prolonged exposure to wildlife conflict at this stage often leads to cumulative psychological impacts in addition to financial losses. At the occupational level, farmers experienced chronic stress, fear, and a sense of powerlessness due to repeated exposure to traumatic events such as elephant crop raiding, as reported by Ridwan *et al.* (2023). Farming emerged as the primary

source of income, with minimal diversification into other livelihoods. This heavy dependence on agriculture increases vulnerability to repeated losses when crops are damaged. Similar patterns across Tanzania have been documented by Urio (2020) who noted that agriculture-dependent communities face persistent challenges from elephant incursions, including reduced household income and limited access to health and education services. Moreover, Western and Mose (2021) highlighted that psychological impacts particularly anxiety and fear are intensified in areas with weak or absent institutional support and compensation mechanisms.

The socioeconomic impacts of Elephant incursions

Our results on the socio-economic effects of elephant incursions indicated an unequal burden of human–elephant conflict across communities, as evidenced by significant differences between Chemchem and Kilimateambo villages. Although both villages reported high levels of support from government and non-governmental organizations, the variation in impact indicators suggests that such assistance is insufficient to fully mitigate localized effects. Households in Kilimateambo

experienced higher levels of food insecurity and property damage, with a greater proportion reporting reduced household food supplies and financial losses. This pattern is consistent with Malley and Gorenflo (2023), who found that socio-economic disruptions are more severe in communities located closer to elephant corridors and with limited adaptive capacity. The high food insecurity reported in Kilimatembo also aligns with studies conducted near Ruaha National Park (Dickman *et al.*, 2014; Hariohay *et al.*, 2019), which documented significant declines in household nutrition and income stability following elephant raids.

In contrast, Chemchem reported lower levels of food supply reduction and property damage, possibly reflecting less frequent elephant incursions or more effective deterrent strategies. Nevertheless, both villages experienced near-total crop damage, underscoring agriculture as the most affected livelihood activity. This finding supports (Mayengo *et al.*, 2017) who identified crop damage as the most common and emotionally distressing form of human–elephant conflict in farming communities. The marked reduction in family movement and similar changes in farming practices observed in both villages indicate comparable behavioral responses to perceived risks. Such adaptations reflect broader trends described by Malley and Gorenflo (2023), who noted that communities often modify agricultural practices and mobility patterns in response to recurrent wildlife conflict, even where impact intensity differs.

Socio-economic impacts were further corroborated through key informant interviews. As one village leader in Kilimatembo noted: “When elephants raid village farms, they destroy crops and consume everything, including rice and maize. I usually report to game officers; sometimes they respond, but most of the time they do not” (Village leader, Kilimatembo, May 2025).

The statistically significant Chi-square results further emphasize the need for localized, context-specific interventions, as socio-

economic impact levels varied significantly between villages. As argued by Wakoli *et al.* (2023), effective mitigation requires tailored measures that account for village-specific vulnerabilities and coping mechanisms. While elephant incursions affected both Chemchem and Kilimatembo, the magnitude and nature of socio-economic impacts varied according to community resilience, institutional support, and ecological exposure. These findings highlight the importance of differentiated policy responses and sustained investment in community-based deterrent strategies to reduce the long-term burden of human–elephant conflict.

Psychological impacts due to elephant incursion

The psychological effects of elephant incursions in Chemchem and Kilimatembo villages indicate that, despite variation in specific symptoms, both communities experience a comparable emotional burden. The findings show that repeated elephant incursions generate similar levels of psychological distress in the two villages, suggesting broadly equivalent emotional impacts. Anxiety and fear were reported in both villages, with slightly higher levels observed in Kilimatembo. This finding is consistent with Banerjee *et al.* (2025), who noted that rural communities exposed to human–elephant conflict often experience chronic stress driven by the constant anticipation of elephant raids, sometimes exceeding stress caused by the actual events. Concern about children’s safety was nearly universal in both villages, reflecting persistent uncertainty regarding family wellbeing in the face of unpredictable wildlife threats. This aligns with Chambers (1995), Torres *et al.* (2018) and Walker (2019) who reported that emotional responses in human–wildlife conflict settings are frequently intensified by perceived risks to children and other dependents. A notable difference between the villages was the prevalence of sleep disturbances, which were substantially higher in Kilimatembo. This variation may be attributed to closer proximity to elephant corridors or

more frequent nocturnal elephant activity. Similar patterns were reported by Zhang *et al.* (2024), who found that sleep disruption caused by nighttime vigilance and raid-related anxiety is common among communities living near protected areas. Sleep deprivation may further exacerbate existing socio-economic challenges by increasing anxiety, reducing productivity, and impairing decision-making.

These findings were supported by a village leader in Kilimatembo, who explained that frequent elephant activity at night prevents residents from sleeping and compels them to remain alert and actively deter elephants using light and noise (Village leader, Kilimatembo, May 2025). Despite some differences, the overall similarity in psychological impacts underscores the pervasive emotional strain associated with persistent fear of elephant incursions. As noted by Kinsky and Kidd (2024), psychological distress linked to human–elephant conflict is often underreported but plays a critical role in shaping community perceptions of wildlife management and conservation. The findings therefore emphasize the need for human–wildlife conflict mitigation strategies that incorporate trauma-informed approaches and mental health support, particularly in areas where conflict is frequent and unresolved.

Copying mechanisms of the elephant incursion deterrent

Various coping mechanisms were used across the villages, indicating that the two communities employ different strategies to reduce human–elephant conflict. These differences are likely shaped by variations in ecological conditions, traditional practices, and access to resources. The use of beehives and chili fences was notably more common in Chemchem, reflecting a preference for low-cost, nature-based deterrents that are widely promoted in community-based conservation programs due to their affordability and local availability. This finding is consistent with Mayengo *et al.* (2017), who reported that communities surrounding Mikumi National Park adopted inexpensive

deterrent methods such as string, oil, and paper to protect crops from elephants. Although these approaches were initially effective, farmers later reported a decline in effectiveness as elephants became habituated, resulting in continued crop damage. Similarly, Western and Mose (2021) demonstrated that deterrents based on chilies and beehives can be effective while also supporting sustainability and community ownership. In contrast, Kimatembo relied more heavily on drums, scarecrows, and watchtowers, indicating a more labor-intensive and reactive approach to elephant deterrence. Page (2023) noted that noise-based deterrents and nighttime guarding, although accessible, often require continuous human presence and can negatively affect community wellbeing by increasing stress and fatigue. Differences in the application of deterrent methods also suggest adaptive responses to local elephant behavior. For example, the presence of watchtowers in Kimatembo indicates a need for increased vigilance, whereas their absence in Chemchem may reflect fewer nocturnal incursions or differences in terrain. The use of “other methods” in both villages further illustrates the creativity and resilience of local communities in responding to persistent threats.

The study also identified significant differences in the perceived effectiveness of coping mechanisms, with Chemchem reporting a higher proportion of methods as highly effective. This highlights the importance of selecting context-appropriate strategies and adapting them to local conditions. The Human–Elephant Coexistence Toolbox emphasizes that locally tailored approaches, supported by community training, tend to achieve greater effectiveness and long-term adoption. The findings underscore the value of community-led, context-specific deterrent strategies and confirm that there is no single solution applicable to all settings. As noted by van de Water and Matteson (2018), empowering communities to select and refine mitigation measures in line with cultural preferences and ecological realities is essential for sustainable human–elephant coexistence.

CONCLUSION AND RECOMMENDATIONS

Conclusion

The study revealed a complex relationship between socio-economic factors, psychological distress, and community-driven deterrent measures associated with elephant incursions in Rhotia Ward. In agriculture-dependent communities such as Kimatembo and Chemchem villages, the socio-economic impacts particularly crop damage, food insecurity, and financial losses placed substantial strain on household livelihoods. These economic losses not only threatened immediate household stability but also undermined long-term livelihood sustainability and conservation efforts. The psychological consequences of living under persistent risk of elephant incursions were evident in elevated levels of anxiety, fear for family safety, and sleep disturbances reported in both villages. These findings highlight the cumulative emotional burden experienced by communities exposed to recurrent wildlife threats. In terms of coping mechanisms, the study demonstrated that residents have shown considerable adaptability in addressing challenges posed by elephant incursions through the use of various deterrent methods, including scarecrows, beehives, chili fences, and drums. However, the effectiveness of these deterrents varied substantially between villages due to differences in local environmental conditions, elephant behavior, and access to resources. This variation underscores the need for structured and coordinated programs to enhance the effectiveness of deterrent measures and to inform the study's recommendations.

Recommendations

The study recommends prioritizing practical, community-based approaches to addressing human–wildlife conflict. These include the establishment of early warning systems, the construction of effective physical barriers such as electric or beehive fences, and the continued

use of non-lethal traditional deterrents. In addition, community participation and school-based awareness programs should be strengthened to build knowledge, confidence, and resilience among local residents regarding elephant behavior and coexistence strategies. Effective collaboration among government institutions, wildlife authorities, non-governmental organizations, and local communities is essential to ensure inclusive and sustainable conflict management. Addressing the socio-economic and psychological impacts of elephant incursions is equally important. Fair and timely compensation schemes for crop and property losses should be provided, alongside support for livelihood diversification through small-scale enterprises, ecotourism, or artisanal activities. Further research is needed to examine psychological impacts across different demographic groups within affected communities. Moreover, community counseling services and mental health outreach programs should be introduced to address anxiety, stress, and fear associated with recurrent elephant incursions. Long-term coexistence between people and wildlife in the Rhotia–Karatu area will be strengthened through continued research and mapping of elephant movement patterns, improved local institutional capacity to respond to wildlife threats, and further evaluation of the effectiveness of specific deterrent mechanisms in supporting conservation outcomes.

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PERCEIVED PATTERNS OF HYENA-INDUCED LIVESTOCK LOSSES IN COMMUNITIES ADJACENT TO MKOMAZI NATIONAL PARK

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ABSTRACT

Human–wildlife conflict remains a critical challenge in landscapes where protected areas border rural communities, particularly where carnivores interact with livestock-based livelihoods. This study examined the spatial and socio-environmental dimensions of human–hyena conflict in villages surrounding Mkomazi National Park, with a focus on perceived livestock depredation patterns, recent trends, and mitigation responses. Data were collected through structured household questionnaires administered to 180 respondents across six villages located at varying distances from the park boundary. Descriptive analyses were used to characterise livestock depredation by species, location, time, and season, while binary logistic regression was applied to identify factors associated with reported depredation incidents. In addition, non-parametric tests were employed to assess differences in perceived hyena sightings among villages. The results indicated that sheep were the most frequently affected livestock, with most attacks occurring at homesteads, during the night, and in the dry season. Although the majority of respondents perceived a decline in livestock attacks over the previous two years, village proximity to the park emerged as the strongest predictor of depredation risk, with closer villages experiencing higher exposure. Socio-demographic factors such as age, gender, education level, and resident status were not significantly associated with reported depredation incidents. Commonly adopted mitigation measures included guarding livestock, using noise and scaring techniques, and employing domestic dogs. These findings highlight the importance of spatial context in shaping human–hyena interactions and underscore the need for location-specific mitigation strategies to reduce livestock losses and promote coexistence between rural communities and large carnivores.

Keywords: *Depredation control, human–wildlife conflict, livestock loss, spotted hyena*

INTRODUCTION

Protected areas play a central role in safeguarding biodiversity by conserving habitats, species, and ecological processes (Pulido-Chadid *et al.*, 2023). However, the effectiveness of these areas is increasingly challenged by rapid human population growth and expanding land-use activities along their boundaries (Veldhuis *et al.*, 2019; Achiso, 2020). As communities seek land for agriculture, livestock keeping, and settlement, interactions between people and wildlife have intensified, often resulting in conflict (Htay *et al.*, 2022). These interactions are particularly pronounced in rural landscapes where livelihoods depend heavily on natural

resources and where wildlife frequently moves beyond protected area boundaries (Mbise *et al.*, 2021; Kegamba *et al.*, 2024).

Living in proximity to protected areas can expose communities to a range of negative impacts, including livestock predation, crop damage, destruction of property, and threats to human safety (Chang’a *et al.*, 2016; Mbise, 2022). Such costs can undermine local support for conservation initiatives and generate antagonistic attitudes towards wildlife and conservation authorities (Htay *et al.*, 2022). Consequently, human–wildlife conflict has become one of the most pressing challenges to biodiversity conservation, requiring

approaches that balance ecological objectives with the livelihood needs of local communities (Raycraft, 2024).

Among wildlife species involved in conflict, large carnivores are often perceived as particularly problematic due to their tendency to prey on livestock. In Africa, spotted hyenas (*Crocuta crocuta*) are among the most adaptable and widespread carnivores, occupying both protected and human-dominated landscapes (Wilkinson *et al.*, 2024). Their behavioural flexibility and opportunistic feeding strategies allow them to exploit a wide range of food sources, including domestic animals (Kissui, 2008; Raycraft, 2024). As a result, hyenas are frequently implicated in livestock depredation incidents in villages located near protected areas, making them a focal species in studies of human–carnivore conflict. Livestock depredation by spotted hyenas represents a significant threat to rural livelihoods, especially in communities where livestock play a central economic, cultural, and social role (Wilkinson *et al.*, 2024). Losses of animals can have immediate and long-term consequences for household income, food security, and social status (Kozierski *et al.*, 2016). These impacts often shape negative perceptions of carnivores and may lead to retaliatory killings, illegal hunting, or reduced tolerance of wildlife presence (Mbise, 2021). Therefore, understanding the patterns and drivers of livestock depredation is essential for designing effective mitigation strategies and promoting their coexistence.

Human–carnivore interactions are influenced by a combination of ecological, spatial, and socio-economic factors (Abade *et al.*, 2014). Variables such as village proximity to protected areas, livestock husbandry practices, prey availability, seasonal conditions, and household characteristics can all affect the likelihood of depredation events (Mbise *et al.*, 2018). Spatial context is particularly important, as villages located closer to protected area boundaries are often more exposed to wildlife incursions (Tarimo *et al.*, 2021). At the same time, perceptions of conflict may vary among

communities depending on prior experiences, cultural values, and access to mitigation measures.

In Tanzania, human–wildlife conflict is widely reported in villages bordering national parks and game reserves (Mbise, 2018). The expansion of agriculture and settlements into former wildlife habitats has increased the frequency of encounters between people and large carnivores (Veldhuis *et al.*, 2019; Chaka *et al.*, 2021). In the case of Mkomazi National Park, communities living along the park boundary regularly report interactions with hyenas, highlighting the need for site-specific assessments of conflict dynamics. Despite the prevalence of these interactions, empirical evidence on the spatial patterns of livestock depredation and local responses remains limited in many areas. Therefore, this study seeks to address this gap by examining the spatial dynamics of human–hyena conflict in villages surrounding Mkomazi National Park. Specifically, it aimed to (i) assess patterns of livestock depredation in relation to livestock type, location, time, and season; (ii) evaluate community perceptions of recent trends in hyena-related livestock losses; and (iii) identify factors influencing reported depredation incidents and mitigation responses. By integrating spatial and social perspectives, the study contributes to a more comprehensive understanding of human–hyena interactions and provides evidence to inform targeted conflict mitigation and conservation planning in Mkomazi and similar landscapes.

MATERIALS AND METHODS

Study Area

The study was conducted in villages located around Mkomazi National Park (MNP) in northern Tanzania (Fig. 1). MNP lies within a semi-arid to sub-humid ecological zone characterised by seasonal rainfall patterns, with a distinct dry season and a shorter wet season (Brockington, 2001). The park supports a variety of wildlife species, including large carnivores such as spotted hyenas, which frequently range beyond protected boundaries

in search of food resources (Mseja *et al.*, 2020). The surrounding villages are predominantly rural and rely largely on agro-pastoral livelihoods. Livestock keeping, particularly of sheep, goats, cattle, and donkeys, forms a major component of household economies, complemented by subsistence crop farming. Settlements are unevenly distributed around the park, with some villages located close to the park boundary and others situated at greater distances. This spatial variation provided an opportunity to examine how proximity to

the protected area influences human–hyena interactions and livestock depredation risk. Villages included in the study were selected to represent a gradient of distances from the park boundary and varying exposure to wildlife incursions (Fig. 1). The selection ensured coverage of areas with frequent reports of hyena activity as well as villages where such interactions were perceived to be less common. This approach allowed for comparative analysis of conflict patterns across different spatial contexts.

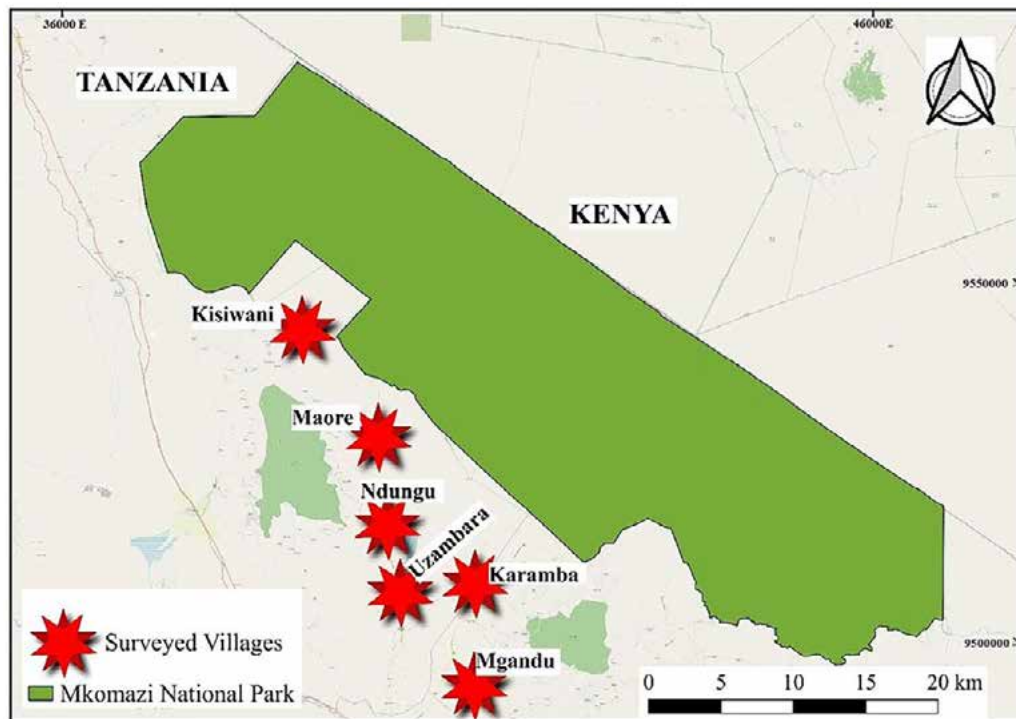


Figure 1: Map of six surveyed villages adjacent to Mkomazi National Park in Tanzania

Data Collection

Primary data were collected using structured household questionnaires administered through face-to-face interviews. A total of 180 households were surveyed across six villages surrounding Mkomazi National Park. Households were selected using systematic random sampling within each village to ensure a broad representation of residents. Only adult household members aged 18 years and above were interviewed, and participation was voluntary. The questionnaire was designed to

capture information on socio-demographic characteristics, including age, gender, education level, resident status, and primary livelihood activity. Respondents were also asked to report livestock ownership, experiences with hyena-related livestock depredation, and perceptions of changes in depredation trends over recent years. Additional questions focused on the timing and location of livestock attacks, including whether incidents occurred at homesteads or grazing areas, during the day or night, and across different seasons. To assess spatial variation in conflict exposure, village

distance from the park boundary was recorded and categorized for analysis. Respondents were further asked about hyena sightings in and around their villages and the mitigation measures employed to reduce livestock losses. These included traditional and locally adopted strategies such as livestock guarding, the use of noise or light, fencing, and the presence of domestic dogs.

Data Analysis

Data were coded and entered into a statistical software package for analysis. Descriptive statistics were used to summarize socio-demographic characteristics, livestock ownership patterns, and reported depredation incidents. Frequencies and percentages were calculated to describe livestock species affected, locations and timing of attacks, perceived seasonal variation, and commonly used mitigation measures. Binary logistic regression analysis was applied to identify factors associated with reported livestock depredation by hyenas. The dependent variable was coded as whether or not a household had experienced livestock loss due to hyenas. Independent variables included village distance from the park, resident status, education level, age category, gender, and primary economic activity. Model outputs were evaluated using regression coefficients, standard errors, Wald

statistics, and significance levels. To examine differences in perceived hyena sightings among villages, non-parametric tests were employed due to the ordinal nature of the response variables. Statistical significance was assessed at conventional probability thresholds. Results were interpreted in relation to spatial proximity to the protected area and local livelihood characteristics, providing insights into the drivers of human–hyena conflict in the study area.

RESULTS

Socio-demographic characteristics of respondents

A total of 180 respondents from six villages surrounding Mkomazi National Park participated in the study. The majority of respondents were male, and most households relied on a combination of livestock keeping and small-scale crop farming as their primary livelihood activities. Educational attainment was generally low, with most respondents having completed primary education, while a smaller proportion had attained secondary education or higher. Respondents represented a wide range of age groups and lengths of residence within the study villages, indicating a mix of long-term residents and more recent settlers (Table 1).

Table 1: Socio-demographic characteristics of respondents in villages surrounding Mkomazi National Park, Tanzania (N = 180)

Variable	Category	Frequency (n)	Percentage (%)
Gender	Male	115	63.9
	Female	65	36.1
Age group (years)	18–30	45	25.0
	31–40	60	33.3
	41–50	35	19.4
	≥51	40	22.2
Education level	No formal education	3	1.7
	Primary education	67	37.2
	Secondary education	93	51.7
	College/University	17	9.4

Variable	Category	Frequency (n)	Percentage (%)
Resident status	Resident	147	81.7
	Non-resident	33	18.3
Tribe	Pare	139	77.2
	Sambaa	41	22.8

Notes: Percentages are calculated based on total respondents (N = 180). Each respondent was assigned to one category per variable.

Perceived Frequency of Livestock Depredation

Livestock depredation incidents were reported by a significant proportion of households, although the frequency and intensity of losses varied across villages. Households owned a variety of livestock species, including sheep, goats, and cattle. Among these, sheep were most frequently reported as being preyed upon by hyenas 57.1% (n = 28) of the reported cases, followed by goats (36.7%, n = 18). Losses of

cattle was comparatively rare (6.1%, n = 3). Most depredation events were reported to occur at homesteads 59.2% (n = 29) of the reported cases, rather than in grazing fields 40.8% (n = 20). Night-time attacks were more commonly reported 65.3% (n = 32), than those occurring during daylight hours 34.7% (n = 17). Seasonal patterns were also evident, with respondents indicating that livestock losses were more frequent during the dry season (75.5%, n = 37) compared to the wet season (24.5%, n = 12) (Fig. 2).

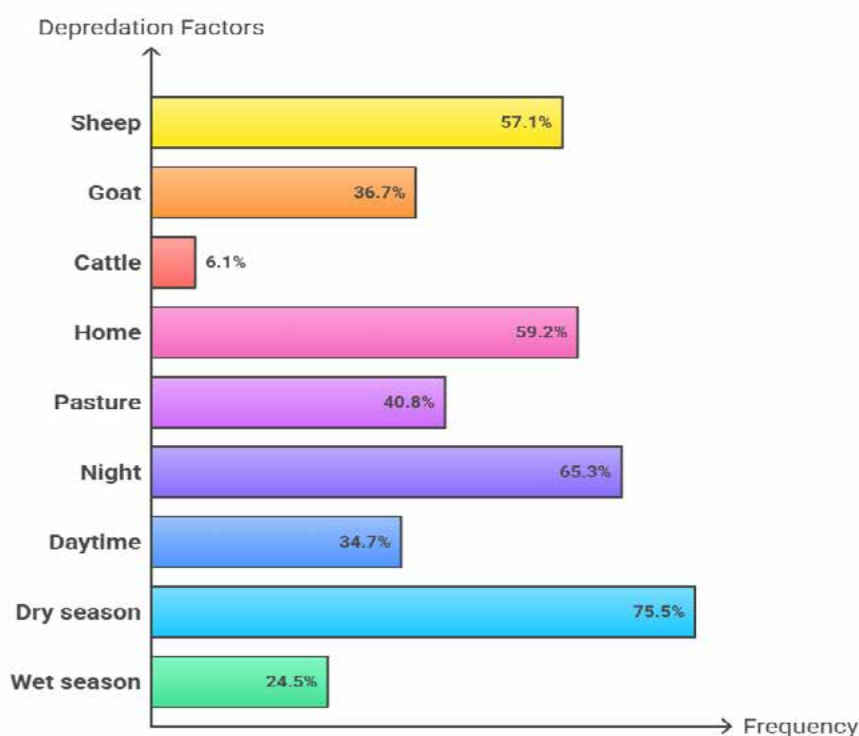


Figure 2: Perceived Frequency of Livestock Depredation

Table 2: Model results showing village distance from the park as a determinant predictor of livestock depredation

Variable	Category	B	S.E	Wald	P value	95% CI
Village distance	Reference: Kisiwani					
	Maore	1.075	0.602	3.186	0.074	2.930
	Karamba	1.599	0.631	6.426	0.011*	4.949
	Ndungu	1.552	0.627	6.128	0.013*	4.719
	Uzambara	2.956	0.876	11.395	0.001**	19.214
	Mgandu	3.364	0.886	14.432	< 0.001***	28.904
Tribe	Reference: Sambaa	−0.966	0.596	2.624	0.105	0.381
Gender	Reference: Female	−0.813	0.430	3.570	0.059	0.444
Age	Reference: 51 and above					
	18–30	1.271	0.916	1.925	0.165	3.563
	31–40	0.060	0.748	0.006	0.936	1.061
	41–50	0.889	0.811	1.204	0.273	2.434
Education level	Reference: Primary	−0.362	0.454	0.638	0.425	0.696
Resident status	Reference: Non resident	0.606	0.572	1.122	0.289	1.833

Factors Influencing Responses on Reported Livestock Depredation Incidents

When asked about changes in livestock depredation over recent years, a majority of respondents perceived a decline in hyena-related attacks (72.8%, $n = 131$). A smaller proportion reported that losses had remained stable, while only a few respondents perceived an increase in depredation incidents (27.2%, $n = 49$). Perceptions of trends varied among villages, with those located closer to the park boundary more likely to report ongoing encounters with hyenas (Table 2).

Spotted Hyena Sightings in the Surveyed Villages

Respondents reported varying frequencies of spotted hyena sightings in and around their villages. Villages situated closer to Mkomazi National Park generally reported higher levels of hyena presence than those located further away. Statistical analysis indicated significant differences in perceived hyena sightings among villages ($p = 0.039$) (Table 3), reflecting spatial variation in exposure to hyena activity across the study area.

Table 3: Kruskal–Wallis test results for respondents’ perceptions of hyena sightings near villages

Variable	Kruskal–Wallis H	<i>p</i> -value
Village distance from park	6.470	0.039*
Tribe	1.071	0.585
Gender	0.932	0.628
Age group	2.845	0.241
Education level	2.055	0.358
Resident status	4.151	0.125

Current Mitigation Measures Against Livestock Depredation

Households reported using a range of strategies to reduce livestock losses to hyenas. The most commonly mentioned measures included guarding livestock at night (28.3%), producing noise to scare away predators (42.2%), and

keeping domestic dogs around homesteads (29.4%). Some respondents also reported reinforcing livestock enclosures, although the effectiveness of these measures varied. Despite the adoption of mitigation strategies, livestock depredation continued to be reported, particularly in villages closest to the park (Fig. 3).

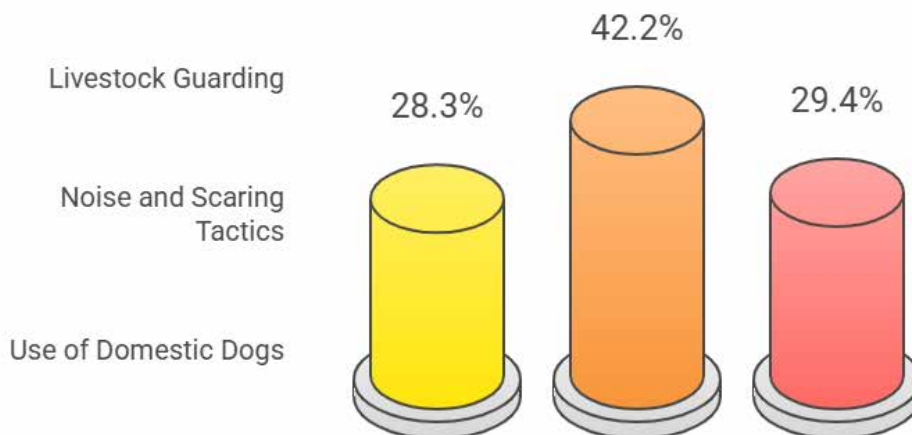


Figure 3: Current mitigation measures against livestock depredation

DISCUSSION

The findings of this study provide clear evidence that human–spotted hyena conflict in villages surrounding Mkomazi National Park is shaped primarily by spatial factors rather than by household socio-demographic characteristics. Livestock depredation was reported more frequently in villages located closer to the park boundary, highlighting the influence of protected areas as sources of large carnivores that move into neighbouring human-dominated landscapes. This pattern is consistent with broader research indicating that proximity to wildlife habitats increases the likelihood of encounters between people and carnivores (Smith & Kasiki, 2000; Carter *et al.*, 2012; Raycraft, 2024). The predominance of sheep and goats among livestock losses reflects the vulnerability of small-bodied livestock to hyena predation (Mbise *et al.*, 2018). These animals are easier for hyenas to capture and transport compared to larger livestock such as cattle. The concentration of attacks at homesteads,

particularly during the night, suggests that livestock enclosures may not provide sufficient protection against nocturnal predators (Chaka *et al.*, 2021). Night-time predation is a common characteristic of hyena behaviour and is often exacerbated by poorly maintained enclosures, especially in areas with limited access to predator-proof housing materials.

Seasonal variation in depredation patterns, with higher losses reported during the dry season, may be linked to changes in prey availability and livestock management practices. During dry periods, natural prey within protected areas may become scarce, prompting hyenas to expand their foraging range into surrounding villages. At the same time, livestock are often kept closer to homesteads or grazed over wider areas in search of pasture, increasing their exposure to predation. These seasonal dynamics underline the need for flexible mitigation strategies that account for temporal changes in risk (Mbise, 2018). Although many respondents perceived a decline in livestock

losses over recent years, depredation remains a persistent concern, particularly in villages closest to the park. Perceived reductions in attacks may reflect improved livestock management practices, increased awareness of conflict mitigation techniques, or changes in hyena population dynamics. However, perception-based assessments should be interpreted with caution, as reported trends may also be influenced by recall bias or shifting tolerance levels among community members.

The absence of significant relationships between livestock depredation and socio-demographic variables such as age, gender, education, and length of residence suggests that exposure to hyena-related conflict is experienced broadly across households (Kozierski *et al.*, 2016; Chaka *et al.*, 2021; Raycraft, 2024). This finding implies that conflict risk is less dependent on individual characteristics and more strongly linked to environmental and spatial conditions. As a result, interventions aimed at reducing conflict may be more effective if they prioritise high-risk locations rather than targeting specific social groups. The widespread use of traditional mitigation measures, including guarding livestock, using noise to deter predators, and keeping domestic dogs, demonstrates local efforts to cope with livestock losses (Mbise, 2018). While these methods may reduce risk to some extent, their effectiveness is often limited, particularly in areas with high hyena activity (Kissui, 2008; Raycraft, 2024). Strengthening livestock enclosures and promoting the adoption of predator-resistant enclosures could provide more sustainable protection (Lichtenfeld *et al.*, 2015; Manoa *et al.*, 2023). Additionally, community-based approaches that encourage collective action and knowledge sharing may enhance the effectiveness of mitigation strategies.

CONCLUSION AND RECOMMENDATIONS

In conclusion, this study demonstrates that human–spotted hyena conflict in villages surrounding MNP is largely driven by spatial

proximity to the protected area rather than by household-level socio-demographic characteristics. Communities located closer to the park boundary experience higher exposure to spotted hyena activity and are therefore more vulnerable to livestock depredation. These findings underscore the importance of considering landscape context when assessing human–carnivore interactions and designing conflict mitigation strategies. Livestock losses were most commonly reported for small stock, particularly sheep and goats, with attacks occurring predominantly at night and near homesteads. Seasonal variation further influenced depredation patterns, with higher losses reported during the dry season. Although many respondents perceived a reduction in livestock attacks in recent years, depredation remains a significant concern for households living closest to the park, indicating that existing mitigation measures may be insufficient in high-risk areas. The lack of significant associations between depredation and socio-demographic variables suggests that conflict impacts are broadly shared across communities, reinforcing the need for interventions that prioritise location-based risk rather than targeting specific social groups. Strengthening livestock protection, especially through improved enclosure design and night-time security, is likely to yield greater reductions in losses than reliance on traditional deterrent methods alone.

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AVIAN-AIRCRAFT COLLISION RISK ASSESSMENT AT THE JULIUS NYERERE INTERNATIONAL AIRPORT IN DAR ES SALAAM, TANZANIA

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ABSTRACT

Airports provide vital habitats for many bird species, but this co-occurrence of aviation activity and wildlife poses a persistent risk of bird-aircraft collisions or bird strikes. Birds' habitat quality in large African cities is increasingly being threatened by urbanisation and climate change, driving wildlife populations to patchy green spaces such as airports resulting into conflicts. This study assessed avian species diversity and their potential collision risks at Julius Nyerere International Airport (JNIA). Data were generated from field observations and bird strike data (2017-2025). A total of 1,966 individual birds representing 49 species from 27 families were recorded across four habitats: runway, grassland, sewage ponds, and landside areas. Species abundance and diversity varied significantly among habitats with highest abundance at sewage ponds. The runway had the highest species diversity but with the lowest abundance, while landside areas were dominated by the invasive alien House Crow. Avian-aircraft collision risk model generated using ecological (e.g. body size, abundance, flock size) and behavioural traits (e.g. bird flight height and direction, distance from the runway) identified the Black-headed Heron, House Crow and African Openbill as the high collision-risk species. The Black-headed Heron was also among the top species involved in bird strike between 2017 and 2025 accounting for 33% of all the collisions, which was in agreement with the model. The findings of this study demonstrate that JNIA's environments support a diverse avifauna some of which pose aviation safety issues. This study underscores the importance of species-specific monitoring and strategic habitat management to reduce bird strike incidents thereby enhancing aviation safety while promoting the conservation of avian diversity.

Keywords: aircraft, aviation hazard, bird strikes, Dar es Salaam airport, invasive alien species

INTRODUCTION

Airports provide important habitats for many bird species, especially in urban areas where natural ecosystems are rapidly disappearing. Features such as grasslands, wetlands, and sewage ponds within airport environments attract a variety of birds seeking food, nesting, and roosting areas. However, these same habitats increase the risk of collisions between birds and aircraft, which pose serious threats to aviation safety, cause costly damage to aircraft, and, in severe cases, lead to human fatalities. The history of the conflict between birds and aircraft began soon after powered flight. In December 1903, Orville and Wilbur Wright

achieved the first powered flight at Kitty Hawk, North Carolina, ushering in a new era of aerial navigation (Dolbeer 2013). Just two years later, on September 7, 1905, Orville Wright recorded the first known bird strike near Dayton, Ohio (Dolbeer 2013). Since then, bird-aircraft collisions have become a global concern as air traffic and bird populations share airspace.

Bird strikes can occur when birds are ingested into jet engines or collide with critical parts of the aircraft, such as the windshield, nose, wings, or tail. Engine ingestion can lead to severe damage or even engine failure, while structural impacts may impair visibility or flight stability (Goraj & Kustron 2018). The International

Bird Strike Committee further emphasizes that engine ingestion and windshield strikes are among the most dangerous types of collisions, underscoring the need for effective mitigation strategies (Thorpe, 2003). A recent tragic example is the Jeju Air Boeing 737-800 accident in South Korea on December 29, 2024, where bird ingestion into both engines caused the loss of 179 lives (ARAIB, 2025). Although fatal events are relatively rare, the economic and operational impacts of bird strikes are significant, leading to expensive repairs, delays, and cancellations. From 1912 to 2002, bird strikes destroyed 80 civil aircraft and caused 231 fatalities worldwide (Thorpe, 2003). Between 1988 and 2022, more than 305 aircraft were destroyed and 464 lives lost due to wildlife strikes (FAA, 2023). In the United States alone, bird strikes are estimated to cause over \$900 million in damages annually (Elias, 2024).

Airports located near ecologically diverse habitats face higher risks of bird strikes due to increased avian activity (Sodhi, 2002). Mekelle and Bahir Dar International Airports in Ethiopia exemplify this, as they are situated in areas surrounded by vegetation, wetlands, and grasslands that attract numerous bird species (Alelign *et al.* 2017; Tefera *et al.* 2022). Entebbe International Airport near Lake Victoria, attract waterfowl and shorebirds (Nature Uganda, 2023), Jeju International Airport (South Korea), situated on an island surrounded by the ocean, is vulnerable to collisions with migratory seabirds (ARAIB 2025) and Istanbul Airport (Turkey), located along a major migratory route between Africa and Europe, experiences high bird activity throughout the year, with the greatest risks occurring during the spring and autumn migration periods (Tutkun and Ateş 2024). Studies show that both natural features and human-made structures, such as roofs, fences, and light poles, provide suitable feeding and nesting sites for birds (Alelign *et al.*, 2017). Additionally, airfield vegetation and animal carcasses further attract birds and increase their presence within airport environments (Tefera *et al.*, 2022). These ecological and

anthropogenic factors collectively contribute to frequent bird strikes in such airports. Gulls, raptors, and waterfowl are among the species most frequently involved, with gulls alone accounting for nearly half of all transport aircraft strikes worldwide (Sodhi, 2002).

Understanding bird strike risks requires assessing avian diversity, abundance, behaviour and distribution and habitats around and within airports. Common habitats found near many airports, such as grasslands, wetlands, water reservoirs, drainage ditches, waste disposal sites, and vegetated buffer zones, provide abundant food, nesting, and resting opportunities that attract birds and consequently elevate collision risks (Wright & Dolbeer 2005; Thorpe 2003). This information is important in predicting the risk of collision potentials. Larger birds and species that form flocks tend to cause more severe damage (Allan *et al.*, 1999). Proximity of bird habitats to active runways, especially wetlands and agricultural fields, also increases the likelihood of strikes (Lee, 2017). In addition to short period studies, historical bird strike data, when analysed effectively, provide valuable insights into high-risk species, times, and locations, guiding more targeted management approaches (Wheeler, 2024).

Globally, airports implement a variety of wildlife hazard management strategies to minimise bird strike risks and enhance aviation safety. Common approaches include vegetation management, habitat modification, regular patrols, and the use of visual and acoustic deterrents, while some airports also employ falconry or lethal control for persistent species (Barras & Seamans 2002; Wright & Dolbeer, 2005). For example, major international airports such as Heathrow (UK), John F. Kennedy (USA), and OR Tambo (South Africa) employ integrated wildlife management programmes that combine systematic monitoring, habitat management, and real-time bird detection systems (Wright & Dolbeer, 2005). Despite these efforts, many airports worldwide continue to experience bird strikes, demonstrating that

even well-managed airports are not entirely risk-free. Julius Nyerere International Airport (JNIA) in Tanzania is among the airports still facing challenges with frequent bird strikes. Although studies have documented avifauna and monitored bird activity at airports in various regions (Allan *et al.*, 1999; Metz *et al.*, 2020), information on bird diversity, abundance, and spatial distribution at JNIA remains limited, constraining the development of targeted, evidence-based bird management strategies suited to its local conditions.

Therefore, this study aims to assess avian species diversity, relative abundance, and potential collision risks at JNIA. The findings will contribute to enhancing aviation safety and promoting sustainable coexistence between airport operations and biodiversity conservation.

MATERIALS AND METHODS

Study Area

This study was conducted at JNIA, the largest and busiest airport in Tanzania, handling more air traffic and receiving more passengers than any other airport in the country. It is located about 12 km southwest of Dar es Salaam's city centre in Kipawa Ward, Ilala District, roughly 11 km from the Indian Ocean. Positioned at an elevation of 55 m above sea level (06°52'41"S, 39°12'10"E), the airport spans 1,700 hectares and features two intersecting runways: the primary runway (05/23) measuring 3,000 m by 46 m and the secondary runway (14/32) measuring 1,000 m by 30 m. The surrounding landscape comprises urban, commercial, and open areas that support diverse avian species such as cattle egrets (*Bubulcus ibis*), sacred ibises (*Threskiornis aethiopicus*) and pied crows (*Corvus albus*) (Howell & Msuya 2004), as well as other animals like jackals, mongooses, and stray cats. This ecological diversity makes JNIA a suitable site for studying bird-aircraft collisions and avian activity patterns affecting aviation safety.

Survey sites were located in all major habitats at the airport include grasslands, artificial wetlands (sewage ponds), along the runways, and landside; areas of the airport where access is not restricted but regulated, including parking areas, terminal entrances and lobby areas, baggage claim areas, retail and food outlet areas, access roads and transportation hubs.

Data Collection

Data was collected using transect and point count methods to assess bird distribution and potential collision risks. Each transect was surveyed twice, during peak bird activity hours in the early morning (07:00–10:00) and late afternoon (15:00–18:00). Four transects were established in the study sites. Each transect had three fixed observation points, at the start, middle, and end.

At each observation point, the point count method was used within a 25m radius circular plot, where birds seen or heard were recorded over 10 minutes. Point count observations were supplemented by vantage points and transect walks. Because of security issues, for runways, taxiways and aprons, in addition to vantage points, transects were established within a few meters in adjacent areas to record all the birds in these habitats. Key data recorded include bird species, size, flight direction, flight height, flock size, flocking behaviour, and nesting or roosting activity.

Secondary data were obtained from the Tanzania Airports Authority (TAA) covering the period from March 2017 to March 2025, documenting wildlife collisions, including bird strikes, and detailing the species involved. In addition, secondary data from Howell and Msuya (2004) were used to provide historical context and comparison on avian species previously recorded and their involvement in bird–aircraft collisions at JNIA.

Data Analyses

Data were organized in Microsoft Excel and analysed using PAST software. Descriptive statistics were calculated to summarize species richness and relative abundance across four habitat types: runway, grassland, sewage pond, and landside. Species diversity and evenness were measured using the Shannon-Wiener (H') and Pielou's (J') indices. The Shapiro-Wilk test indicated that bird abundance data were not normally distributed ($p < 0.05$), leading to the use of non-parametric tests. Consequently, a Kruskal-Wallis's test assessed differences in

bird abundance among habitats, and Dunn's post hoc test identified which habitat pairs differed significantly.

For the avian collision risk assessment, each bird species was evaluated using six ecological and behavioural criteria: body size, flight altitude, flight direction, relative abundance, flock size, and proximity to the runway. Each of these criteria was scored on a Likert scale from 0 to 3, where 0 indicated very low or no risk, 1 indicated low risk, 2 indicated moderate risk, and 3 indicated high risk (Table 1)

Table 7: The scoring table for predicting avian collision risks at Julius Nyerere International Airport

Criteria	Score 0–3	Description / Range
Body size	1–3	Small (<250 g) = 1, Medium (250–1000 g) = 2, Large (>1000 g) = 3
Flight height	0–3	High (>101 m) = 1, Medium (51–100 m) = 2, Low (1–50 m) and Ground-dwelling = 3
Flight direction	1–3	Away from runway = 1, Parallel = 2, Crossing runway = 3
Relative abundance	1–3	Low (<0.1) = 1, Moderate (0.2–0.99) = 2, High (>1) = 3
Flock size	1–3	Single = 1, Pair = 2, Group (>2) = 3
Proximity to the runway	0–3	Landside = 0, Sewage ponds = 1, Grassland = 2, Runway = 3

The scores for all criteria were summed and then averaged to calculate an overall Risk Index for each species. Based on this average, species were categorized into low risk (mean score 0–1.0), moderate risk (1.1–2.0), or high risk (2.1–3.0) groups. Bird strike records from March 2017 to March 2025 were analysed by identifying the species involved in each incident and comparing their observed strike frequencies with the model's predicted risk categories. Species classified as high risk were expected to show more strike occurrences than those in moderate or low categories.

RESULTS

Avian Species Diversity in Different Habitats

A total of 1,966 individual birds, comprising 49 species from 27 families, were recorded across four habitats at JNIA: runway, sewage ponds, grassland, and landside. Overall, five dominant species accounted for 72.6% of all individuals observed: White-faced Whistling Duck (*Dendrocygna viduata*, 32.3%), Little Grebe (*Tachybaptus ruficollis*, 15.8%), House Crow (*Corvus splendens*, 13.3%), Black-headed Heron (*Ardea melanocephala*, 5.8%), and

Zanzibar Red Bishop (*Euplectes nigroventris*, 5.4%) (Appendix 1). Species abundance and diversity metrics varied notably among habitats (Table 2). A Kruskal-Wallis's test revealed a significant difference in bird abundance among habitats ($H = 88.41$, $p < 0.05$). Dunn's post hoc test showed that the sewage ponds differed significantly from all other habitats: landside ($p < 0.05$), grassland ($p = 0.03$), and runway ($p = 0.002$). The rest of the habitats did not show a significant difference in avian abundance when paired ($p > 0.05$).

The sewage ponds recorded the highest abundance ($n = 1,323$), primarily due to large flocks of White-faced Whistling Duck ($n = 635$) and Little Grebe ($n = 310$). However,

despite the high number of individuals, species diversity ($H' = 1.66$) and evenness ($J' = 0.33$) were moderate to low, indicating dominance by a few species. In contrast, the runway exhibited the lowest abundance ($n = 123$) but had the highest species diversity ($H' = 2.40$) and evenness ($J' = 0.55$), with 19 species recorded (Table 2). Similarly, the grassland habitat supported a well-balanced bird community with moderate abundance ($n = 197$), high diversity ($H' = 2.32$), and relatively high evenness ($J' = 0.51$), across 20 species. The landside recorded the highest species richness (27 species) but had very low diversity ($H' = 0.16$) and evenness ($J' = 0.16$) (Table 2), largely due to the numerical dominance of House Crow ($n = 214$, 66.3%).

Table 8: Summary of avian community metrics across four habitat types at Julius Nyerere International Airport, including diversity (Shannon Index, H'), evenness (J'), minimum and maximum number of individuals per species, mean abundance, and standard error.

Habitat	Species Richness	Abundance	Diversity (H')	Evenness (J')	Min	Max	Mean	Std. error
Sewage Ponds	16	1,323	1.66	0.33	2	635	82.69	41.37
Landside	27	323	2.32	0.16	1	214	11.96	7.88
Grassland	20	197	2.32	0.51	1	56	9.85	3.15
Runway	19	123	2.4	0.55	1	28	6.47	1.84

Avian Potential Collision Risks with Aircraft

Collision risk assessment for bird species at JNIA was based on a combination of six behavioural and ecological factors: bird size, flocking behaviour, flight direction, relative abundance, flock size, and runway proximity. Species scoring the highest collision risk include African Openbill (*Anastomus lamelligerus*), House Crow (*Corvus splendens*) and Black-headed Heron (*Ardea melanocephala*). Species falling in the moderate risk category include Little Swift (*Apus affinis*), Hamerkop (*Scopus umbretta*), Cattle Egret (*Bubulcus ibis*), Sacred Ibis (*Threskiornis aethiopicus*), Zanzibar Red Bishop (*Euplectes nigroventris*), Pied Crow (*Corvus albus*), Lilac-breasted Roller (*Coracias caudatus*) and Little Grebe. Nonetheless, most species fall into the low-risk

category as indicated in Appendix 1.

Of the three species classified as high risk, the Black-headed Heron recorded the highest number of bird strikes at JNIA, with 15 incidents (33%) occurring between 31 March 2017 and 15 March 2025. This makes it the top species involved in aircraft collisions. Additionally, in the same period, several species categorised as moderate risk were also involved in collisions. These include the House Crow (1 strike), Little Swift (*Apus affinis*) (1 strike), Hamerkop (1 strike), and Sacred Ibis (5 strikes, accounting for 11% of total strikes) (TAA data; 2017-2025).

These results validate the potential collision risk model, confirming that many species assessed as high or moderate risk based on

behaviour and habitat use are potential good candidates for aircraft collisions. However, of the 17 species involved in strikes at JNIA in the last 8 years, six species, namely Hadada Ibis (*Bostrychia hagedash*), Barn Owl (*Tyto alba*), Cranes, Eagle, Marabou Stork (*Leptoptilos crumenifer*), and Palm-nut Vulture (*Gypohierax angolensis*), were not recorded during the current study. Additionally, two species, the Black Kite (*Milvus migrans*) and the Bateleur (*Terathopius ecaudatus*) reported by Howell and Msuya (2004) to have been involved in bird collision at JNIA were not sighted. Bateleur was involved in a strike after it came to the ground to feed on Black-kite, which had been killed during a strike, a case in which one strike led directly to another strike. Notably, nightjars as a group have been involved in multiple bird strikes at JNIA, accounting for a total of seven collisions (16%) over the period from 31 March 2017 to 15 March 2025. Among them, the Square-tailed Nightjar has been recorded in at least one confirmed strike, despite being ranked as low risk in the potential collision assessment.

DISCUSSION

Avian Species Diversity in Different Habitats

Our study represents the second systematic survey of bird species at Julius Nyerere International Airport (JNIA), following the work of Howell & Msuya (2004). Earlier, Harvey & Howell (1987) provided an account of several bird species of Dar es Salaam documented which listed ed species that were once common at JNIA, including the Black Kite (*Milvus migrans*), Bateleur (*Terathopius ecaudatus*), and Marabou Stork (*Leptoptilos crumenifer*), which we did not record in the current survey. The apparent decline of these species in Dar es Salaam may be linked to increased human activity, urbanization, higher flight traffic, and habitat management around the airport. Conversely, human-commensal species such as the House Sparrow (*Passer domesticus*) and House Crow have become more abundant, now dominating the landside

areas, reflecting shifts in the airport's avian community over time.

The study also revealed significant variation in avian abundance, diversity, and evenness across runways, sewage ponds, grasslands, and landside areas, consistent with previous research on the influence of habitat characteristics on bird presence at airports (Oduntan *et al.* 2012; Robinson *et al.* 2021). Sewage ponds had the highest bird abundance, with 1,323 individuals, mainly White-faced Whistling Ducks and Little Grebes, supporting literature that aquatic habitats are hotspots for waterbirds (Alelign *et al.*, 2017; Blackwell *et al.*, 2019). These ponds attract birds due to constant water availability and abundant food. Despite high abundance, species diversity and evenness were moderate to low, reflecting dominance by a few species. This skewed composition, typical of wetlands, increases collision risk because of flocking behaviour and the relatively large body size of these birds (Hu *et al.* 2020).

Runway habitats showed the lowest bird abundance (123 individuals) but surprisingly recorded the highest species diversity and evenness. This suggests a more balanced and distributed avian community, likely due to limited resource availability and constant human activity. Bird presence on the runway is shaped by continuous surveillance and frequent hazing conducted by the Airport Wildlife Unit Crew, which effectively deters most birds. As a result, although fewer birds are observed on runways, the variety of species present is broader and more evenly distributed, reducing the dominance of high-risk species.

Grasslands similarly supported high diversity and evenness, with moderately high abundance. These open habitats hosted more evenly distributed bird communities, likely due to the uniform vegetation cover that supports a variety of foraging opportunities without favouring a single dominant species. This aligns with a study from Ethiopian airports which found scattered bird populations with moderate strike risks in grasslands (Tefera *et al.* 2022).

Landside areas exhibited the highest species richness (27 species), yet had notably low diversity and evenness due to the dominance of the invasive alien House Crow. These urban-edge habitats are highly influenced by human activity and waste, which attract synanthropic species. The dominance of House Crow reflects findings from Lagos Airport (Oduntan *et al.* 2012), where crows were drawn by accessible anthropogenic food sources, leading to high concentrations and increased collision risks.

Avian Potential Collision Risks with Aircraft

The collision risk assessment, based on behavioural and ecological traits, identified species such as the Black-headed Heron, House Crow and African Openbill as high-risk. The actual strike data validate this, with the Black-headed Heron involved in 15 confirmed collisions (33%), affirming the utility of the predictive model. This finding is consistent with global research indicating that large-bodied, flocking, and low-flying birds contribute disproportionately to bird strikes (Dolbeer *et al.*, 2015; Goraj & Kustron, 2018).

Several moderate-risk species, such as Sacred Ibis and Little Swift, were also involved in strikes, confirming the practical relevance of species' risk scores. Interestingly, some low-risk species, such as nightjars, were also involved in multiple strikes (n=7), highlighting the role of activity period (e.g., crepuscular or nocturnal behaviour) in underestimating risk (Blackwell *et al.* 2019). The Square-tailed Nightjar's (*Caprimulgus fossii*) involvement, despite a low-risk score, suggests that factors like cryptic behaviour, low visibility, and silent flight may increase undetected risks, particularly during night operations. According to regional avifaunal records (Harvey & Howell, 1987), three nightjar species are expected to occur in the JNIA area. These include the Square-tailed Nightjar (*Caprimulgus fossii*), a year-round resident, the Freckled Nightjar (*C. tristigma*), an uncommon resident typically found in rocky or built environments, and the European Nightjar (*C. europaeus*), which is a Palearctic

migrant present between October and April. Given this seasonal pattern, bird strikes involving nightjars that occur between May and September, when the European Nightjar is absent, can be confidently attributed to resident species, the Square-tailed Nightjar.

Birds such as the African Openbill, despite their large size and high collision risk, may avoid aircraft due to behavioural adaptability. Frequent exposure to human activity and noise at airports can lead to habituation, allowing them to adjust flight patterns and avoid active runways. Their ability to respond to disturbances, including aircraft movements and wildlife control measures, enables them to forage or roost near hazardous areas without entering flight paths during critical times. This adaptive avoidance lowers collision risk, though it persists if disturbances are inconsistent or large flocks are suddenly startled.

Over six strike-involved species, including Hadada Ibis (*Bostrychia hagedash*), Barn Owl, and Marabou Stork, were not observed during the current survey, possibly due to seasonal movements, migration timing, or changes in habitat use. This reflects seasonal variability in bird activity and strike patterns, as reported by Metz *et al.*, (2020), and underscores the importance of long-term and continuous monitoring to capture full avifaunal dynamics.

The unusual case involving a Bateleur feeding on a struck Black Kite, leading to a secondary strike (Howell & Msuya 2004), also reflects how scavenger attraction to strike remnants can generate cascading collision events. This is rarely documented in literature but has clear implications for strike response protocols and carcass removal.

Most collisions at JNIA occur during take-off and landing phases, when aircraft are flying at low altitudes and are more likely to encounter birds (TAA wildlife strike data from 2017 to 2025). However, these bird collisions generally did not result in severe damage. The main consequences observed include delays in

transportation and aborted flights, rather than significant aircraft damage or threats to human safety. Despite the relatively low severity, these incidents still pose operational challenges and highlight the importance of maintaining consistent bird hazard management and timely response measures.

CONCLUSION AND RECOMMENDATIONS

Our study revealed that bird activity and distribution at JNIA vary across habitats, with wetlands and open areas showing the highest species richness and abundance. Waterbirds and scavengers pose the greatest collision risk due to their flocking behaviour and large body size. The analysis of the secondary data (TAA data; 2017-2025) highlighted limited species identification capacity and insufficient long-term monitoring, particularly of the nocturnal species. We also noted that the presence of food attractants (e.g. live prey, carcasses, litter) around the runway is a key challenge that may lead bird strike. Therefore, in order to reduce high-risk bird presence and minimise bird-aircraft collisions at JNIA, we recommend implementing habitat modifications such as maintaining grass height at or below 20 cm to reduce suitable habitats for birds and their prey; removing food attractants to prevent scavenger presence; conducting long-term studies to understand activity patterns; and strengthening the capacity of TAA Wildlife Units for accurate species identification to enable proper records for bird strike. Moreover, the avian collision risk model from our study can be incorporated into the TAA Wildlife Hazard Management Programme to support daily and seasonal operations. Specifically, the model should be applied during routine airfield patrols and periodic monitoring for the identified high collision risk species and hotspots around the airfield.

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Appendix 1: Avian species' potential collision risk score and the actual collision incidences, where; 3 = high potential collision risks with aircraft, 2 = moderate potential collision risks with aircraft, 1= low potential collision risks with aircraft, 0 =very low/no potential collision risks with aircraft. B.S = Bird size, D.R = Distance from the runway, D.F = Direction of flight, F.H = Flight height, F.S = Flock size, R.A = Relative abundance, T.S = Total scores, T.M.S= Total mean scores, P.C.R= Potential Collision Risks, E= Encounters and C.I.F= Collision Incidence Frequency (used data from TAA; 2017-2025). Species are arranged based on the percentage of recorded bird-aircraft collisions, from the highest to the lowest incidence, thereafter, alphabetically.

Bird species	Scientific name	B.S	D.R	D. F	F.H	F.S	R.A	T.S	T.M.S	P.C.R	E	C.I.F (%)
Black-headed Heron*	<i>Ardea melanocephala</i>	3	2	2	2	3	3	15	2.5	3	15	33%
Square-tailed Nightjar*	<i>Caprimulgus fossii</i>	2	0	0	2	1	2	7	1.17	1	8	18%
Sacred Ibis*	<i>Threskiornis aethiopicus</i>	3	0	0	0	3	3	9	1.5	2	5	11%
Hadada Ibis*	<i>Bostrychia hagedash</i>	-	-	-	-	-	-	-	-	-	4	9%
White-faced Whistling Duck*	<i>Dendrocygna viduata</i>	2	0	0	0	3	3	8	1.33	1	2	4%
Barn Owl*	<i>Tyto alba</i>	-	-	-	-	-	-	-	-	-	1	2%
Cranes*	<i>Grudae</i>	-	-	-	-	-	-	-	-	-	1	2%
Eagle*	<i>Accipitridae</i>	-	-	-	-	-	-	-	-	-	1	2%
Hamerkop*	<i>Scopus umbretta</i>	2	2	3	2	2	3	14	2.33	2	1	2%
House Crow*	<i>Corvus splendens</i>	2	2	3	3	3	3	16	2.67	3	1	2%
Lapwing*	<i>Vanellia</i>	-	-	-	-	-	-	-	-	-	1	2%
Little Swift*	<i>Apus affinis</i>	1	1	2	3	2	2	11	1.83	2	1	2%
Marabou Stork*	<i>Leptoptilos crumenifer</i>	-	-	-	-	-	-	-	-	-	1	2%
Owl*	<i>Strigiformes</i>	-	-	-	-	-	-	-	-	-	1	2%
Palm-nut Vulture*	<i>Gypohierax angolensis</i>	-	-	-	-	-	-	-	-	-	1	2%
Spectacled Weaver*	<i>Ploceus ocularis</i>	-	-	-	-	-	-	-	-	-	1	2%
African Openbill	<i>Anastomus lamelligerus</i>	3	3	3	2	3	3	17	2.83	3	0	0%
African Pipit	<i>Anthus cinnamomeus</i>	1	1	3	3	2	3	13	2.17	2	0	0%
Barn Swallow	<i>Hirundo rustica</i>	1	1	3	0	1	2	8	1.33	1	0	0%
Black-winged Red Bishop	<i>Euplectes hordeaceus</i>	1	1	0	0	1	2	5	0.83	1	0	0%

Bird species	Scientific name	B.S	D.R	D. F	F.H	F.S	R.A	T.S	T.M.S	P.C.R	E	C.I.F (%)
Black-winged Stilt	<i>Himantopus himantopus</i>	1	0	0	0	3	3	7	1.17	1	0	0%
Blue-capped Cordon-Bleu	<i>Uraeginthus cyanocephalus</i>	1	1	0	0	1	1	4	0.67	1	0	0%
Bronze Mannikin	<i>Spermestes cucullata</i>	1	1	0	0	1	1	4	0.67	1	0	0%
Brown-hooded Kingfisher	<i>Halcyon albiventris</i>	1	1	0	0	1	2	5	0.83	1	0	0%
Cattle Egret	<i>Bubulcus ibis</i>	2	1	0	2	3	3	11	1.83	2	0	0%
Coastal Cisticola	<i>Cisticola haematocephalus</i>	1	2	0	0	2	3	8	1.33	1	0	0%
Common Bulbul	<i>Pycnonotus barbatus</i>	1	1	0	0	2	2	6	1	1	0	0%
Common Sandpiper	<i>Actitis hypoleucos</i>	1	0	0	0	3	3	7	1.17	1	0	0%
European Bee-eater	<i>Merops apiaster</i>	1	0	0	0	2	2	5	0.83	1	0	0%
European Nightjar	<i>Caprimulgus europaeus</i>	2	0	0	0	1	2	5	0.83	1	0	0%
European Roller	<i>Coracias garrulus</i>	2	1	0	0	2	2	7	1.17	1	0	0%
Fan-tailed Widowbird	<i>Euplectes axillaris</i>	1	1	0	3	2	2	9	1.5	2	0	0%
Great Egret	<i>Ardea alba</i>	2	1	0	0	2	2	7	1.17	1	0	0%
Grey Heron	<i>Ardea cinerea</i>	3	0	0	0	2	3	8	1.33	1	0	0%
Grey Tit-Flycatcher	<i>Fraseria plumbea</i>	1	0	0	0	1	1	3	0.5	1	0	0%
Grey-headed Kingfisher	<i>Halcyon leucocephala</i>	1	1	0	0	1	1	4	0.67	1	0	0%
House Sparrow	<i>Passer domesticus</i>	1	1	0	3	3	3	11	1.83	2	0	0%
Intermediate Egret	<i>Ardea intermedia</i>	2	0	0	0	1	1	4	0.67	1	0	0%
Lilac-breasted Roller	<i>Coracias caudatus</i>	2	1	3	0	3	3	12	2	2	0	0%
Little Bee-eater	<i>Merops pusillus</i>	1	1	0	0	2	2	6	1	1	0	0%
Little Egret	<i>Egretta garzetta</i>	2	0	0	2	1	2	7	1.17	1	0	0%
Little Grebe	<i>Tachybaptus ruficollis</i>	2	0	0	0	3	3	8	1.33	1	0	0%
Olive Bee-eater	<i>Merops superciliosus</i>	1	1	0	0	2	2	6	1	1	0	0%
Pied Crow	<i>Corvus albus</i>	2	1	3	3	2	3	14	2.33	2	0	0%
Rattling Cisticola	<i>Cisticola chiniana</i>	1	0	0	0	1	1	3	0.5	1	0	0%
Red-backed Shrike	<i>Lanius collurio</i>	1	0	0	0	1	2	4	0.67	1	0	0%

Bird species	Scientific name	B.S	D.R	D. F	F.H	F.S	R.A	T.S	T.M.S	P.C.R	E	C.I.F (%)
Scarlet-chested Sunbird	<i>Chalcomitra senegalensis</i>	1	1	0	0	1	2	5	0.83	1	0	0%
Speckled Mousebird	<i>Colius striatus</i>	2	1	0	0	2	2	7	1.17	1	0	0%
Spotted Palm Thrush	<i>Cichladusa guttata</i>	1	0	0	0	1	2	4	0.67	1	0	0%
Stripped Kingfisher	<i>Halcyon chelicuti</i>	1	0	0	0	1	1	3	0.5	1	0	0%
Tawny-flanked Prinia	<i>Prinia subflava</i>	1	0	0	3	3	2	9	1.5	2	0	0%
Village Indigobird	<i>Vidua chalybeata</i>	1	0	0	0	1	1	3	0.5	1	0	0%
Village Weaver	<i>Ploceus cucullatus</i>	1	1	0	0	3	3	8	1.33	1	0	0%
White-winged Widowbird	<i>Euplectes albonotatus</i>	1	1	0	0	1	1	4	0.67	1	0	0%
Yellow-billed Stork	<i>Mycteria ibis</i>	3	0	0	0	1	2	6	1	1	0	0%
Yellow-breasted Apalis	<i>Apalis flavida</i>	1	0	0	0	1	1	3	0.5	1	0	0%
Yellow-throated Longclaw	<i>Macronyx croceus</i>	1	1	0	0	2	2	6	1	1	0	0%
Zanzibar Red Bishop	<i>Euplectes nigroventris</i>	1	2	0	0	3	3	9	1.5	2	0	0%
Total		-	-	-	-	-	-	-	-	-	45	100%

* Species involved in bird strike at JNIA between 31/03/2017 to 15/03/2025.

HUMAN-ELEPHANT INTERACTIONS VIEWED AS ASSETS OR LIABILITIES IN RURAL TRANSFORMATIONS: A CASE STUDY FROM TANZANIA WITH EXAMPLES FROM OTHER ELEPHANT RANGE STATES

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ABSTRACT

The presence of humans in perceived elephant range areas across urban, peri-urban, and rural landscapes sparks ongoing debates among development practitioners, conservationists, local communities, and governments. Rising demands for land and rural transformation have intensified human-elephant contacts. Human population growth has escalated the need for infrastructure, agriculture, and pastures, thereby fragmenting elephant habitats. The current study aims to determine whether the elephant is viewed as an asset or a liability in the rural transformation. The desk review was deployed to search for the literature in Scopus and Google Scholar, and EBSCOhost. The results show that regional and local conservation efforts have yielded positive outcomes, such as increasing elephant populations, with animals reappearing in areas where they had been absent for decades. However, growing human and elephant populations heighten competition for limited resources, escalating conflicts. Common incidents include crop raiding, human and animal injuries or deaths, and infrastructure damage. Retaliatory actions against elephants often involve killing, displacement, or barriers. In conclusion, these interactions are increasingly seen as liabilities by rural communities. Recommendations urge governments, conservationists, and communities to collaborate on solutions that shift perceptions towards viewing elephants as assets. This includes extensive dialogues to develop programs benefiting individuals directly, exploring consumptive and non-consumptive utilizations, and implementing strategies amid rising elephant numbers.

Keywords: Human-wildlife conflict, transboundary conservation, rural communities and range states

INTRODUCTION

Humans and animals have coexisted since ancient times, though interpretations of this relationship vary across disciplines and cultures (Buijs & Jacobs, 2021). Human-wildlife interactions encompass diverse contacts, from harmonious coexistence to conflict. Human population growth and development activities increasingly pressure natural ecosystems, especially near protected areas, leading to habitat degradation, fragmentation, and reduced animal ranges, which alter wildlife behavior and intensify negative interactions (Heinrichs *et al.*, 2016). Elephants exemplify

these dynamics, evoking strong emotional and cultural responses. As charismatic megafauna, they are perceived as rare, beautiful, impressive, endangered, and dangerous (Albert *et al.*, 2018). They function as flagship species for conservation funding, keystone species for ecological roles, and umbrella species benefiting broader biodiversity (Caro, 2003) which if given sufficient protected habitat area, will bring many other species under protection'. Historically, umbrella species were employed to delineate specific reserve boundaries but are now used in two senses: (1. Elephants' ranging and feeding behaviors, such as breaking branches or uprooting trees shape ecosystems.

Though sometimes viewed as destructive, these actions promote biodiversity by opening canopies for grass growth, aiding grazers and smaller herbivores, and creating microhabitats for insects, reptiles, and mammals (Ishida *et al.*, 2018). However, elephants and humans compete for habitat, which supports nutritional needs, space, health, behavior, and well-being for both (Mellor, 2017). This overlap generates tension as both pursue biological and economic needs in limited spaces, resulting in human-elephant conflicts. Rapid population growth drives demands for agricultural land, settlements, and infrastructure, encroaching on wildlife habitats and fragmenting migratory corridors (Kideghesho, 2001; Sarma *et al.*, 2021). Elephants, as wide-ranging species reliant on routes, are vulnerable, leading to increased conflict frequency and intensity across range states.

MATERIALS AND METHODS

This review followed a desk-review method to synthesize existing literature on human-elephant interactions and their effects on rural transformations. Relevant publications were gathered from academic databases such as Google Scholar, EBSCOhost, and Scopus, along with grey literature from reports by organizations like IUCN and national wildlife authorities. Keywords included “human-elephant conflicts,” “elephant conservation,” “transboundary conservation,” “rural communities,” “elephant range states,” “human-wildlife interactions,” “community-based conservation,” and other related phrases, combined using Boolean operators such as AND, OR, and NOT. Inclusion criteria targeted relevant peer-reviewed articles, reports, and case studies from 1990 to 2025, with a focus on African elephant range states. Data were analyzed thematically to identify patterns in elephant populations, conflicts, mitigation strategies, and community initiatives with a primary focus on Tanzania but also including

examples from West, Southern, and Eastern Africa. Literature gaps, such as inconsistent monitoring data, were identified to guide future recommendations.

RESULTS AND DISCUSSION

Status of elephant populations, Human-Elephant Conflict, and mitigation measures in African Elephant range states. A total of 75 publications were included in the current review.

West Africa

West Africa hosts 13 elephant range states with forest (*Loxodonta cyclotis*) and savanna (*Loxodonta africana*) elephants. Habitats are small, fragmented, and isolated, with populations estimated at 9,106 -13,872 across 142,500 km² (2.8% of total land area) (IUCN, 2016). Anthropogenic pressures from mining, logging, and agriculture exacerbate habitat loss (IUCN, 2016). Key populations reside in the WAP transboundary complex (Benin, Burkina Faso, Niger), accounting for over half of regional forest elephants and two-thirds of savanna elephants (Blanc, 2003). Transboundary areas often serve as corridors, dispersal zones, or migration routes, overlapping with human activities (Mpanduji *et al.*, 2002; Osborn & Parker, 2003; Kideghesho, 2015; Munthali *et al.*, 2018).

The IUCN’s 2003 action plan identified 18 transboundary ranges, with five critical for conservation (Sebogo & Barnes, 2003). Table 1 summarizes these corridors, threats, and efforts. Elephants are subjected to intense anthropogenic pressures, illegal offtakes, and Human Elephant Conflict (HEC), including crop raiding, injuries, deaths, encroachments (Boafo *et al.*, 2004; Compaore *et al.*, 2020). Strategies emphasize species conservation but lack clear individual benefits or sustainable monitoring.

Table 1: Descriptions of Important Trans frontier elephant corridors, threats and Conservation Efforts made in West Africa range states

S/N	Area Description	Threats	Conservation efforts
1	Complex of protected areas in Burkina Faso, Niger, Benin and Part of Northern Togo	• Human pressure (Poaching and Cattle herding) • Land loss and degradation of habitats	• Development of elephant management strategy • Develop and sign sub regional protocol for elephant management • Develop management plan for elephant habitats • Comprehension of HEC and setting conflict prevention • Develop effective tools to involve locals in elephant management • Develop compensation schemes for damages caused by elephants • Develop joint law enforcement capacity and anti-poaching activities • Development of sustainable elephant monitoring program
2	The Sahelian elephants of the Gourma which move between Mali and northern Burkina Faso	• Land cover degradation • Anthropogenic pressure (human habitation on banks of permanent water points, expansion of agricultural lands, pressure from domestic livestock) • Human elephant conflicts (damage to crops and harvests, fatal accidents)	• Development of elephant management strategy including population monitoring • Strengthens trans frontier cooperation • Strategies to reduce HEC • Identify elephant corridors and migration routes • Develop and implement elephant management plan
3	Network of protected areas in Southern Burkina Faso, Northern Ghana and Togo	• Lack of concerted efforts to mitigate HEC • Poaching	• Development of elephant management strategy including population monitoring • Strategies to reduce HEC • Strategies to reduce poaching • Strategies to secure corridor • Education programs to empower local communities on elephant management

S/N	Area Description	Threats	Conservation efforts
4	Forest complex in the Bia Gaoso, western Ghana that connects with Djimbarakrou Forest Reserves in eastern Cote D Ivoire and Grebo Forest in Liberia	• Habitat loss (Agriculture expansion and human pressure) • Hunting pressure • Loss of genetic variability due to lack of interbreeding as a consequence of habitat fragmentation	• Develop a harmonized strategy for elephant conservation on the corridor • Develop HEC mitigation strategies • Develop strategies to reduce poaching • Develop sustainable elephant population monitoring • Secure stakeholder support for management of elephant corridors • Establish long term sustainable financial mechanisms • Ensure sustainable cocoa production in the corridor
5	Tai National Park and associated forests in eastern Cote d Ivoire and Grebo Forest in Liberia	• Habitat fragmentation due to human pressure • Elephant range reduction • Paucity of information on elephant population and movements	• Harmonize elephant conservation and management programmes • Reduce illegal elephant killings • Sustainable management of the corridor • Elephant population viability management

Note: Modified and customized from Sebogo & Barnes, (2003)

Southern Africa

Southern Africa includes nine savanna elephant range states (Zambia, South Africa, Lesotho, Zimbabwe, Angola, Mozambique, Eswatini, Malawi and Botswana), with ~352,271 elephants (74% of Africa's total) across ~7 million km² (Purdon *et al.*, 2018; Huang *et al.*, 2022). The Kavango-Zambezi (KAZA) TFCA spans 300,000 km², protecting ~250,000 elephants across five countries (Van Aarde & Jackson, 2007; Muboko, 2017; Cassidy & Salerno, 2020).

HEC is inevitable, with 80% of elephants ranging outside protected areas (von Gerhardt-Weber, 2011). Conflicts involve deaths, injuries, crop raiding, property damage, and resource competition (Le *et al.*, 2011). High densities in KAZA amplify issues, leading to

retaliatory killings and poaching (IUCN, 2016). Habitat fragmentation from land use changes exacerbates contacts (Warner, 2008).

Mitigation includes compensation, land planning, barriers, culling, deterrents, and translocation (Le *et al.*, 2011), but many fail. Integrated solutions are needed (Ceaşu *et al.*, 2019; Shaffer *et al.*, 2019). Examples include elephant auctions in Namibia, Transfrontier Conservation Areas (TFCAs) for conservation and development (Muboko, 2017), Community-Based Natural Resource Management (CBNRM) in Botswana, Namibia, and Zimbabwe (Nattrass, 2021), CAMPFIRE in Zimbabwe (Le *et al.*, 2011; Delsink *et al.*, 2023), immuno-contraception in South Africa (Delsink *et al.*, 2023), and unpalatable crops in Limpopo and Lubombo TFCAs (Henley *et al.*, 2023). Conflicts persist despite efforts.

Eastern Africa

Eastern Africa comprises eight range states (Eritrea, Ethiopia, Kenya, Rwanda, Somalia, South Sudan, Tanzania and Uganda), with 880,648 km² of elephant range (20.5% of total land) and ~86,373 elephants (IUCN, 2016). Development in Ethiopia, Kenya, Rwanda, Uganda, and Tanzania fragments habitats, while instability in South Sudan and Somalia hinders conservation. Tanzania hosts 80% of the regional elephants, Kenya 12%.

HEC varies but includes socio-economic (property damage) and biological (encroachment, fires, deaths) impacts (Nyumba *et al.*, 2018; Hariohay *et al.*, 2020; Munyao *et al.*, 2020; Geleta & Mengesha, 2022). Conflicts intensify outside protected areas (Tiller *et al.*, 2021). In Tanzania, conflicts occur where ranges overlap human areas, especially near boundaries, corridors, or dispersal zones (Penrod *et al.*, 2022). Crop raiding is widespread (Kiffner *et al.*, 2021), e.g., cashew destruction in Ruvuma (Montero Botey *et al.*, 2022) and plantations in West Kilimanjaro and Rombo (Mmbaga *et al.*, 2017). Hotspots include western Serengeti, Burunge Wildlife Management Area, Mkomazi National Park, and others (Kwaslema *et al.*, 2017; Penrod *et al.*, 2022). Human deaths peaked in districts like Bariadi and Kilombero (2017–2019) (Penrod *et al.*, 2022).

A 2009 survey (covering 2007–2009) found 94% of 64 districts with HEC, averaging 500 incidents annually, including 30–40 injuries and 40–50 deaths (Kaiza-Boshe, 2024). Factors include improved protection, ivory bans, population growth, habitat loss, and climate change (Van De Perre *et al.*, 2014; Riggio & Caro, 2017; Mmbaga *et al.*, 2017; Matata *et al.*, 2022; Ntuke *et al.*, 2022; Shiweda *et al.*, 2023).

Communities view elephants as a major threats to livelihoods (Kaiza-Boshe, 2024). Elephants are perceived as dangerous animals that cause significant crop destruction, damage storage

facilities, and occasionally result in human injuries and fatalities, leading to fear and anxiety among local communities. Mitigation relies on communities, with support using repellent pests (Tiller *et al.*, 2021), noises, lights, beehives (King & Kaelo, 2011), vehicles, drones, and barriers (Matsika *et al.*, 2020). Elephants adapt, reducing efficacy. A 31-year review found chili-based methods and guarding effective but limited in scalability (Montgomery *et al.*, 2022).

Community efforts to co-exist with elephants in Africa: Has it paid?

(i) Communal Areas Management Programme (CAMPFIRE) in Zimbabwe

CAMPFIRE, initiated in 1989 under the 1975 Parks and Wildlife Act, empowers communities to manage wildlife for income (Karl, 2002). Rural District Councils distribute hunting/tourism revenues (Frost & Bond, 2008). It aimed to reduce poaching via sustainable use. USAID funded it until 2000 (Frost & Bond, 2008).

Pros: Increased tolerance from economic value; growing elephant populations.

Cons: Habitat encroachment from population growth; HEC (crop/house damage, injuries, deaths); no compensation, relying on CAMPFIRE/ecotourism. Affected individuals view elephants as liabilities.

(ii) Communal Wildlife Conservancies in Kenya

Established in 2013 under the Wildlife Conservation and Management Act, conservancies manage wildlife on private/communal land (King & Kaelo, 2011). By 2015, over 140 covered 30,300 km². They protect habitats, provide payments/jobs (Gamewatchers-Safaris, 2023).

Porini Conservancy exemplifies success: leasing Maasai land for tourism, ensuring income (1 tent/700 acres) and expanding ranges (Gamewatchers-Safaris, 2023).

Challenges: Donor dependency; small community gains; intensified HEC; non-local operators; resource exclusion; compensation sustainability. Fencing is considered despite legal conflicts.

(iii) Community-Based Conservation in Tanzania

Community Wildlife Management (CWM), or Community-Based Conservation (CBC), began in 1987/88 to reconcile

conservation and development (Songorwa, 1999). Wildlife Management Areas (WMAs) cover 27,683.6 km². Table 2 lists established WMAs, status, and challenges.

Over 20 years since formal implementation (2003), WMAs achieved national policy advancements (e.g., 1998 Wildlife Policy, 2009 Act, 2012 Regulations) and local benefits (habitat protection, revenues, user rights) (USAID, 2013). Challenges include governance (power imbalances, transparency), economics (high costs, limited revenues), conservation (attitudes, monitoring), establishment processes, impact assessments, and gender equity (USAID, 2013).

Table 2: Established WMA in Tanzania their current status and challenges

SN	Name	District	Size (km ²)	Status	Challenges/ Problems
1	Ikona	Serengeti	242.3	Operational	The dissatisfaction of the community due to loss of access to resources that were once accessible before the establishment of WMA, unemployment, HWC (crop raiding by wildlife), lack of compensation following crop raiding and or livestock depredation, low revenue generation (Kicheleri <i>et al.</i> , 2018; Mgonja & Uswege, 2022)
2	Burunge	Babati	617.0	Operational	Increased environmental destructions outside the WMA due to hard laws to access WMA resources, lack of economic benefits, unfair benefit distribution between villages, HWC due to increased population of humans, wildlife, and livestock, the economic benefit is worse than before WMA, food insecurity, poor governance, few benefits and low revenue (Hernold, 2017; Kicheleri <i>et al.</i> , 2018; Salerno <i>et al.</i> , 2016)development and sustainable land-use in the rural areas. The aim of the organization is to provide local communities with economic benefits and including them in the projects of wildlife and habitat conservation (Kicheleri <i>et al.</i> , 2018)
3	Makao	Meatu	768.9	Operational	Loss of access to resources after the establishment of WMA, unemployment, HWC (crop raiding by wildlife), lack of compensation following crop raiding and or livestock depredation (Mgonja & Uswege, 2022), food insecurity (Salerno <i>et al.</i> , 2016)efforts focus on balancing trade-offs in local communities near the borders of protected areas. Devolving rights and control over certain resources to communities is increasingly considered necessary, but decades of attempts have yielded limited success and few lessons on how such interventions could be successful in improving livelihoods. We investigated a key feature of household well-being, the experience of food insecurity, in villages across Tanzania's northern wildlife tourist circuit. Using a sample of 2,499 primarily livestock-keeping households we compared food insecurity in villages participating in the country's principal community-based conservation strategy with nearby control areas. We tested whether community-based projects could offset the central costs experienced by households near strictly protected areas (i.e. frequent human-wildlife conflict and restricted access to resources, poor institutional framework (Lwankomezi <i>et al.</i> , 2021), low revenue generation (Kicheleri <i>et al.</i> , 2018)
4	Enduimet	Longido	751.4	Operational	Regulation and policy issues (Dekker, 2020), HWC, food insecurity, poor governance, low revenue generation, and unfair benefit sharing (Kicheleri <i>et al.</i> , 2018; Osipova <i>et al.</i> , 2018; Salerno <i>et al.</i> , 2016)through focus group discussions, key informant interviews, a questionnaire survey, and literature review, we found that the participation of local people in both the establishment and management of the WMA was limited and rife with conflict. While benefits have materialized at the communal level, local people saw neither value nor benefit of the WMA to their livelihoods. Specifically, local people's access to natural resources got worse while private eco-tourism investors and the central government have gained financially. Contrary to the livelihood enhancing WMA rhetoric, top-down institutional choices have sidelined democratically elected Village Governments and successive legislative adjustments disenfranchised and dispossessed them and their constituencies.

SN	Name	District	Size (km ²)	Status	Challenges/ Problems
					We conclude that village governments should consistently demand for their legal rights to the resources on their land until the WMA approach to conservation and development is democratized.”, ”container-title”: ”Sage Open”, ”DOI”: ”10.1177/2158244018774382”, ”ISSN”: ”2158-2440, 2158-2440”, ”issue”: ”2”, ”journalAbbreviation”: ”Sage Open”, ”language”: ”en”, ”page”: ”2158244018774382”, ”source”: ”DOI.org (Crossref
5	Randilen	Monduli	312.0	Operational	Environmental degradation, boundary disputes, loss of wildlife corridors, HWC, increased poverty level and inadequate Income Generating Activities (IGA), Unplanned intensive road network, and unreliable telecommunication (Penrod <i>et al.</i> , 2022); general lack of inclusion of communities in the preparation and implementation hence misunderstanding between villagers and WMA management (Kicheleri <i>et al.</i> , 2018)
6	Makame	Kiteto	5, 372.0	Operational	Food insecurity, HWC, (Salerno <i>et al.</i> , 2016) efforts focus on balancing trade-offs in local communities near the borders of protected areas. Devolving rights and control over certain resources to communities is increasingly considered necessary, but decades of attempts have yielded limited success and few lessons on how such interventions could be successful in improving livelihoods. We investigated a key feature of household well-being, the experience of food insecurity, in villages across Tanzania’s northern wildlife tourist circuit. Using a sample of 2,499 primarily livestock-keeping households we compared food insecurity in villages participating in the country’s principal community-based conservation strategy with nearby control areas. We tested whether community-based projects could offset the central costs experienced by households near strictly protected areas (i.e. frequent human–wildlife conflict and restricted access to resources, Low revenue generation and lack of community involvement (Kicheleri <i>et al.</i> , 2018)
7	Ukutu	Morogoro Rural	639.0	Operational	Low revenue generation (Kicheleri <i>et al.</i> , 2018)
8	Iluma	Kilombero & Ulanga	509.0	Operational	Low revenue due to lack of investors and boundary conflicts (Kicheleri <i>et al.</i> , 2018).
9	Ngarambe-Tapika	Rufiji	767.0	Operational	Managerial challenges (Katandukila & Chibwana, 2022), HWC (crop damage and livestock depredation), economic loss due to opportunity cost, (Kicheleri <i>et al.</i> , 2018; Mdendemi & Nzunda, 2021; Noe <i>et al.</i> , 2019) and power struggles of conservation partnership (John, 2021)
10	Juhiwangumwa	Rufiji	497.0	Operational	HWC, (Noe <i>et al.</i> , 2019), power struggles of conservation partnership (John, 2021)
11	Waga	Mufindi, Iringa & Mbarali	315.0	Operational	Vegetation changes due to climate change (Komba <i>et al.</i> , 2021), poor socioeconomic performance, and boundary conflicts
12	Umemaruwa	Mbarali & Wanging’ombe	61.0	Operational	Vegetation changes due to climate change (Komba <i>et al.</i> , 2021), low revenue, boundary conflicts, little collaboration between WMA and the surrounding protected areas, lack of awareness about WMA operations and future plans by village members, poor governance and management, human-related disturbances threatening ecological integrity

SN	Name	District	Size (km ²)	Status	Challenges/ Problems
13	Pagawa-Idodi	Iringa	777.0	Operational	Vegetation changes due to climate change (Komba <i>et al.</i> , 2021), human wildlife conflict, low revenue due to lack of investors, boundary conflicts, poor governance, and gender inequality, and low revenue generation (Kicheleri <i>et al.</i> , 2018)
14	Tunduru	Tunduru	1391.0	Operational	Low revenue generation (Kicheleri <i>et al.</i> , 2018)
15	Mbarang'andu	Namtumbo	2471.0	Operational	(HWC - crop destruction, human and animal injuries, and deaths) (Mugisha, 2022), low revenue (Mawi & Mashenene, 2020), low revenue, overlapping and incompatible legal frameworks where the same land is gazetted for mining and WMA (Kicheleri <i>et al.</i> , 2018)
16	Ching'oli	Tunduru	938.0	Operational	Low revenue generation (Kicheleri <i>et al.</i> , 2018; Mawi & Mashenene, 2020)
17	Kisungule	Namtumbo	1345.0	Operational	Low revenue due to a lack of investors, poor governance and management, poor socioeconomic performance and boundary conflicts (Kicheleri <i>et al.</i> , 2018; Mawi & Mashenene, 2020)
18	Kimbanda	Namtumbo	2150.0	Operational	Low revenue generation (Kicheleri <i>et al.</i> , 2018; Mawi & Mashenene, 2020)
19	Liwale	Liwale	4515.0	Operational	Low revenue generation (Kicheleri <i>et al.</i> , 2018)
20	Uyumbu	Urambo	839.0	Operational	Low revenue generation, HWC (crop raiding) (Kicheleri <i>et al.</i> , 2018)
21	IPole	Sikonge	2406.0	Operational	Poor governance, low revenue generation, HWC (crop raiding) (Kicheleri <i>et al.</i> , 2018)

CONCLUSION AND RECOMMENDATIONS

Community-based conservation aims to foster rural transformation and human-elephant coexistence by empowering local communities. Shifting from exclusionary models, it seeks tangible benefits for communities bearing wildlife costs. However, emphasis on communal infrastructure overlooks individual burdens from HEC (crop damage, injuries, deaths). No reviewed programs or governments offer robust compensation. Global bans on consumptive use allow elephant populations to grow, intensifying localized conflicts amid human expansion and habitat loss.

To promote rural transformations with elephants as assets:

- i. Engage conservation authorities, local communities, and ministries (wildlife, agriculture, land, minerals, local government) in prioritizing land use where wildlife offers viable income, balancing competition, Discuss with local communities and authorities' sustainable wildlife uses, addressing population growth and non-consumptive limitations,
- ii. Develop optimal interventions for problem animal control, focusing on HEC, and
- iii. Recommend strategies positioning wildlife, especially elephants, as catalysts for individual rural benefits, diverging from community-focused approaches.

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FACTORS INFLUENCING COMMUNITY PERSPECTIVES ON ELEPHANT CROP DAMAGE AND CONSERVATION SUPPORT IN TANZANIA

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ABSTRACT

Human–elephant interactions remain a major conservation and livelihood challenge in many parts of sub-Saharan Africa, particularly for rural communities living near protected areas. This study examined how socio-demographic and contextual factors shape local perceptions of elephant crop raiding, perceived changes in crop raiding over the previous five years, and support for elephant conservation in two contrasting protected areas in northern Tanzania. Data were collected through structured household questionnaires administered to 240 respondents across eight villages adjacent to Lake Manyara National Park and Swagaswaga Game Reserve between 2022 and 2023. Ordinal probit and ordinal logistic Generalised Linear Mixed Models were applied to analyse perceptions of crop raiding intensity and trends, while binary logistic regression was used to assess determinants of support for elephant conservation. The type of protected area emerged as a strong predictor of conservation support, with residents near the game reserve showing significantly lower support than those near the national park. Village location, education level, and age group also significantly influenced conservation attitudes. These findings demonstrate that perceptions of human–elephant interactions and conservation support are shaped by both social characteristics and the institutional context of protected areas. Tailored, location-specific conservation strategies that address local concerns and enhance tangible benefits are therefore essential for promoting sustainable coexistence between people and elephants.

Keywords: Community engagement, Conservation attitudes, Crop damage, Elephants, Human-wildlife coexistence

INTRODUCTION

Interactions between humans and wildlife are increasingly recognised as complex social–ecological processes resulting from overlapping demands for space, resources, and livelihoods (Hariohay *et al.*, 2020; Banerjee & Aiyadurai, 2021; de la Torre *et al.*, 2021; Malley & Gorenflo, 2023; Bajuta *et al.*, 2025). As human populations expand and land-use patterns intensify, encounters between people and wildlife have become more frequent, particularly in landscapes surrounding protected areas (Warner, 2008; Walker, 2012; de la Torre *et al.*, 2021; Montgomery *et al.*, 2022). In many regions of Africa, these interactions are dominated by negative outcomes, including crop damage, livestock losses, injuries, and loss of human life, which collectively undermine

local support for wildlife conservation (Walker, 2012; Runyoro *et al.*, 2018; Mmbaga, 2022; Montgomery *et al.*, 2022).

Among conflict-prone species, elephants pose a particularly serious challenge due to their large body size, extensive home ranges, and high daily food requirements (Montgomery *et al.*, 2022; Chang’a & Kifaro, 2023). When elephants move beyond protected area boundaries into agricultural land, the resulting crop damage can be severe, rapid, and difficult for farmers to prevent (Laizer, 2024). Such incidents often generate fear, economic hardship, and resentment among affected households, shaping negative attitudes toward both elephants and conservation authorities (Warner, 2008; Hariohay *et al.*, 2017; Runyoro *et al.*, 2018). Consequently, human–elephant

conflict has become a central obstacle to the long-term conservation of elephant populations across much of their range (Warner, 2008).

Perceptions held by local communities play a critical role in determining the success or failure of conservation initiatives (Mudimba, 2019). Attitudes toward wildlife and conservation are not uniform but are influenced by a range of socio-demographic factors, including age, education, livelihood strategies, length of residence, and proximity to protected areas (Nsonsi *et al.*, 2018). Individuals who depend heavily on farming and experience frequent crop losses may be more likely to view elephants as threats rather than assets (Walker, 2012; Mbise & Sosiya, 2023). Conversely, communities that receive tangible benefits from conservation such as employment, infrastructure, or tourism revenue may be more willing to tolerate wildlife presence and support conservation efforts.

In Tanzania, where a large proportion of protected areas are bordered by rural settlements, human–elephant conflict remains widespread (Kaswamila, 2009; Chang’a & Kifaro, 2023). Communities living adjacent to national parks and game reserves often face similar risks from crop raiding but may experience different governance arrangements, benefit-sharing mechanisms, and levels of engagement with conservation institutions (De Boer & Ntumi, 2001; Lamarque *et al.*, 2009; Kamweya *et al.*, 2012). These differences can strongly influence how local people perceive wildlife impacts and whether they are willing to support conservation objectives (Mudimba, 2019).

Understanding how social characteristics and local context shape perceptions of elephant crop raiding and conservation support is therefore essential for designing effective mitigation and outreach strategies (Warner, 2008; Blackie & Sowa, 2019; Mbise & Sosiya, 2023). This study investigates these relationships by comparing communities located near Lake Manyara National Park (LMNP) and Swagaswaga Game Reserve (SGR). Specifically, it addresses three

questions: (i) how local residents perceive the severity of elephant crop raiding, (ii) how they perceive changes in crop raiding over the past five years, and (iii) how these perceptions, together with socio-demographic factors, influence support for elephant conservation. By identifying the key determinants of local attitudes, the study contributes to the development of more inclusive and context-sensitive conservation approaches.

MATERIALS AND METHODS

Study area

The research was undertaken in northern and central Tanzania in communities located adjacent to two distinct protected areas: Lake Manyara National Park (LMNP) and Swagaswaga Game Reserve (SGR). These sites were deliberately selected to represent different protected-area categories and management contexts, allowing comparison of community perceptions under contrasting conservation regimes. LMNP is situated within the Manyara Region and partially extends into Monduli District of the Arusha Region (Mikole *et al.*, 2024). The surrounding landscape is characterised by a mosaic of pastoral land, small-scale agriculture, human settlements, and conservation areas. Livestock keeping, particularly cattle, goats, and sheep, is the dominant livelihood, although crop cultivation (notably maize, beans, millet, rice, bananas, and vegetables) is increasingly practised, especially in villages located closer to park boundaries (NBS, 2022). The area experiences a semi-arid to semi-humid tropical climate, with average annual rainfall of approximately 600 mm and temperatures ranging between 14 °C and 30 °C. Rainfall occurs in two distinct seasons: short rains from November to January and long rains from February to May (Deus *et al.*, 2013). The proximity of LMNP to other protected areas, including Tarangire National Park and Makuyuni Wildlife Park, places the region within an important wildlife movement landscape, intensifying human–wildlife interactions.

SGR is located in Kondo District, Dodoma Region, within a predominantly semi-arid environment. The area experiences a prolonged dry season extending from April to December, followed by a shorter rainy season between December and April. Annual rainfall varies substantially, ranging from approximately 400 mm in plateau areas to about 1000 mm in the highlands, with average temperatures between 15 °C and 20 °C (Senkondo, 2023). Local livelihoods in villages surrounding SGR are primarily based on subsistence farming and livestock rearing, supplemented by activities such as beekeeping and small-scale timber use (Mbiase & Senkondo, 2024). Compared with

LMNP, SGR is less integrated into tourism circuits, which may influence local perceptions of conservation benefits.

Eight villages were included in the study: four adjacent to LMNP (Oltukay, Esilalei, Naiti, and Mbuyuni) and four adjacent to SGR (Misirai, Hurumbi, Sogasoga, and Singida Ndogo). These villages are located at varying distances from the respective protected-area boundaries (ranging from approximately 1.3 km to over 30 km), providing a spatial gradient for examining how proximity influences perceptions of elephant crop raiding and conservation support (Fig.1).

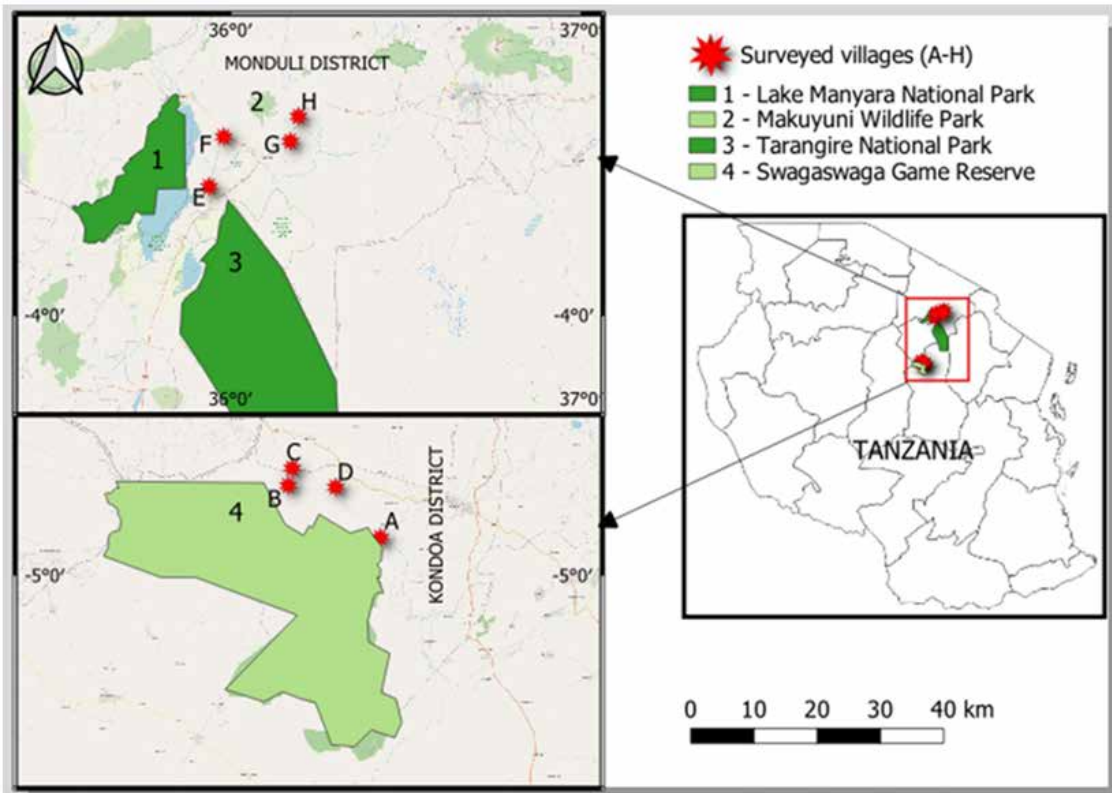


Figure 1: Map showing the surveyed villages (A-H) in Monduli and Kondo Districts.

Data collection

Data were collected through a household-based questionnaire survey administered to residents living in villages neighbouring LMNP and SGR. A total of 240 households were surveyed, with 30 households selected from each village

to ensure balanced representation across study sites. Households were selected using random sampling procedures within each village. Prior to data collection, permission was obtained from local authorities, including village leaders, and respondents were informed about the purpose of the study. Participation was voluntary,

and verbal consent was obtained from all respondents before interviews commenced. Data collection around SGR was conducted between March and April 2022, while surveys in villages surrounding LMNP were carried out between March and April 2023. The questionnaire was administered through face-to-face interviews and consisted of closed-ended questions designed to capture socio-demographic characteristics, experiences with elephant crop raiding, perceptions of changes in crop-raiding frequency, and attitudes toward elephant conservation. Respondents were asked three key perception-based questions: (i). *To what extent are your crops raided by elephants?* (response options: extremely high, very high, moderate, undecided); (ii). *How would you describe the trend of elephant crop raiding over the past five years?* (increasing, stable, decreasing, undecided); (iii). *Do you consider elephant conservation important for biodiversity conservation?* (yes/no). To enhance consistency in responses, explanatory guidelines were provided for the crop-raiding intensity categories. These guidelines clarified the approximate frequency and severity associated with each response option, ensuring that respondents shared a common understanding of the terms used.

Data analysis

Completed questionnaires were coded and entered into SPSS version 27 for statistical analysis. Initial data screening and descriptive statistics were performed to summarise socio-demographic characteristics and overall response patterns. To examine factors influencing perceptions of elephant crop raiding, two Generalized Linear Mixed Models (GLMMs) were fitted using ordinal regression techniques. The first model applied an ordinal probit link function to assess the influence of socio-demographic variables on perceived levels of crop raiding intensity. The second model employed an ordinal logistic link function to analyse perceptions of changes in crop-raiding frequency over the preceding five years. For the third research question, which

focused on support for elephant conservation, a binary logistic regression model was used. The dependent variable was coded as 1 for respondents who supported elephant conservation and 0 for those who did not. Predictor variables across all models included resident status, economic activity, education level, age category, gender, village location, and protected-area type. Statistical significance was evaluated using Wald chi-square tests, with a significance threshold set at $p < 0.05$. Model outputs were interpreted in terms of regression coefficients, confidence intervals, and odds ratios where applicable.

RESULTS

Socio-demographic characteristics of respondents

The socio-demographic profiles of respondents from communities neighbouring LMNP and SGR reveal both similarities and notable contrasts. Perceptions regarding changes in elephant crop raiding over the past five years differed markedly between the two protected areas. In villages surrounding SGR, responses were relatively evenly distributed: 40.0% of respondents perceived an increase in crop raiding, 35.8% believed the situation had remained stable, and 24.2% were uncertain. In contrast, a substantially larger proportion of respondents living near LMNP (74.2%) reported an increase in elephant crop raiding, while only 15.0% perceived no change and 10.8% expressed uncertainty (Table 1).

No statistically significant difference was observed in the length of residence between the two study areas ($\chi^2 = 3.900$, $p = 0.273$). Across both protected areas, the largest proportion of respondents fell within the 21–40-year age category. Sampling across the eight villages was evenly distributed, with each village contributing an equal proportion of respondents, resulting in a highly significant difference in village representation by design ($\chi^2 = 240.00$, $p < 0.001$) (Table 1).

Economic activities differed significantly between the two areas ($\chi^2 = 9.603$, $p = 0.002$). Farming was the dominant livelihood among households near SGR, whereas livestock keeping was more prevalent among respondents living adjacent to LMNP. Educational attainment did not differ significantly between the two sites ($\chi^2 = 0.089$, $p = 0.766$), with

the majority of respondents in both areas reporting primary education as their highest level achieved. Similarly, age distribution and gender composition did not vary significantly between the two protected areas ($p > 0.05$), although males constituted a higher proportion of respondents in both locations (Table 1).

Table 1: Socio-demographic characteristics of respondents from villages adjacent to Swagaswaga Game Reserve (SGR) and Lake Manyara National Park (LMNP), Tanzania

Variable	Category	SGR (%)	LMNP (%)	χ^2	p -value
Resident status (years)	< 10	21.7	32.5		
	10–20	25.0	21.7		
	21–40	34.2	31.7		
	> 41	19.2	14.2	3.900	0.273
Village distance (km)	1.3	12.5			
	1.9	12.5			
	3.9	12.5			
	5.7	12.5			
	6.6		12.5		
	11.3		12.5		
	27.4		12.5		
	31.6		12.5	240.000	<0.001***
Economic activity	Farming	60.8	40.8		
	Livestock keeping	39.2	59.2	9.603	0.002**
Education level	Primary	74.2	75.8		
	Secondary or above	25.8	24.2	0.089	0.766
Age category (years)	18–30	24.2	25.0		
	31–40	32.5	33.3		
	41–50	15.0	19.2		
	> 50	28.3	22.5	1.443	0.696
Gender	Male	65.8	58.3		
	Female	34.2	41.7	1.434	0.231

Socio-demographic influences on perceived levels of elephant crop raiding

The ordinal probit model identified threshold parameters that distinguish between perceived levels of elephant crop raiding. The threshold

separating the “Very high” category from lower categories was statistically significant, as was the threshold for the “Moderate” category, indicating clear differentiation in respondents’ perceptions across these levels. In contrast, the threshold associated with the “Extremely

high” category was not statistically significant (Table 2). None of the socio-demographic predictors included in the model significantly influenced respondents’ perceived intensity of elephant crop raiding. Length of residence showed no meaningful effect across all residence categories when compared to long-term residents. Similarly, economic activity

did not significantly affect the likelihood of reporting higher or lower levels of crop raiding. Educational attainment, age group, and gender were also non-significant predictors, suggesting that perceptions of crop raiding intensity were broadly shared across demographic groups rather than being shaped by individual socio-economic characteristics (Table 2).

Table 2: Ordinal probit GLMM results assessing socio-demographic determinants of perceived intensity of elephant crop raiding

Predictor	B	SE	95% CI (Lower–Upper)	Wald χ^2	p-value
Thresholds					
Extremely high	0.460	0.331	–0.189 – 1.110	1.932	0.164
Very high	1.048	0.335	0.390 – 1.705	9.761	0.002**
Moderate	1.405	0.340	0.739 – 2.072	17.082	<0.001***
Resident status (ref: >41 years)					
<10 years	0.220	0.241	–0.253 – 0.693	0.832	0.362
10–20 years	0.329	0.246	–0.153 – 0.810	1.787	0.181
21–40 years	0.287	0.238	–0.180 – 0.753	1.451	0.228
Economic activity (ref: Livestock)					
Education (ref: Secondary+)	0.200	0.151	–0.096 – 0.497	1.757	0.185
Age category (ref: >50)					
18–30	–0.059	0.216	–0.483 – 0.365	0.075	0.784
31–40	–0.214	0.211	–0.628 – 0.199	1.031	0.310
41–50	–0.194	0.239	–0.663 – 0.275	0.657	0.418
Gender (ref: Female)					
	–0.077	0.156	–0.383 – 0.230	0.240	0.624

Perceived trends in elephant crop raiding over the past five years

The ordinal logistic regression model assessing perceived trends in elephant crop raiding revealed a significant upper threshold, indicating a clear distinction between respondents who perceived an increasing trend and those who did not. The lower threshold, however, was not statistically significant (Table 3). Among the explanatory variables, economic activity emerged as a significant predictor. Respondents engaged primarily in farming were more likely to perceive an increase in elephant crop raiding over the previous five years compared

to those whose main livelihood was livestock keeping. Education level approached statistical significance, with respondents holding primary education tending to report increasing crop raiding more frequently than those with higher educational attainment (Table 3). Length of residence, age category, and gender did not significantly influence perceptions of crop raiding trends. These findings suggest that recent experiences linked to livelihood activities, rather than demographic attributes, play a greater role in shaping perceptions of temporal changes in elephant crop raiding (Table 3).

Table 3: Ordinal logistic regression examining factors influencing perceived trends in elephant crop raiding over the past five years

Predictor	B	SE	95% CI (Lower–Upper)	Wald χ^2	p-value
Resident status (ref: >41 years)					
<10 years	−0.329	0.397	−1.106 – 0.449	0.685	0.408
10–20 years	−0.738	0.422	−1.564 – 0.089	3.062	0.080
21–40 years	−0.499	0.394	−1.272 – 0.274	1.602	0.206
Economic activity (ref: Livestock)	0.665	0.262	0.151 – 1.179	6.439	0.011*
Education (ref: Secondary+)	0.577	0.310	−0.030 – 1.183	3.471	0.062
Age category (ref: >50)					
18–30	−0.037	0.364	−0.749 – 0.676	0.010	0.919
31–40	0.103	0.354	−0.590 – 0.796	0.085	0.771
41–50	−0.727	0.436	−1.582 – 0.127	2.781	0.095
Gender (ref: Female)	0.178	0.276	−0.362 – 0.718	0.416	0.519

Determinants of local support for elephant conservation

The binary logistic regression model demonstrated that several contextual and socio-demographic factors significantly influenced support for elephant conservation. The model constant was statistically significant, indicating a generally high baseline probability of supporting conservation (Table 4). Type of protected area was a strong predictor of conservation support. Respondents living adjacent to the Game Reserve were significantly less likely to support elephant conservation compared to those residing near the National Park. Village location also exerted a significant overall effect, with marked variation in support levels across villages. Residents of some villages showed substantially higher odds of supporting conservation, while others were

significantly less supportive compared to the reference village (Table 4).

Education level was negatively associated with conservation support, with respondents who had attained only primary education being less likely to support elephant conservation than those with secondary education or higher. Age also influenced conservation attitudes, as younger respondents (18–30 years) were significantly less supportive of conservation initiatives compared to older respondents (Table 4). In contrast, length of residence, economic activity, and gender did not significantly affect the likelihood of supporting elephant conservation. Overall, the results indicate that conservation support is shaped primarily by spatial context, education, and age rather than by livelihood type or duration of residence (Table 4).

Table 4: Binary logistic regression analysing determinants of local support for elephant conservation

Predictor	B	SE	Wald	p-value	Exp(B)
Constant	4.356	1.022	18.164	<0.001	77.920
Protected area (ref: National Park)	−2.867	0.788	13.241	<0.001***	0.057
Resident status (ref: >41 years)					
<10 years	−0.371	0.526	0.497	0.481	0.690
10–20 years	−0.717	0.526	1.853	0.173	0.488
21–40 years	−0.938	0.501	3.504	0.061	0.391

Predictor	B	SE	Wald	p-value	Exp(B)
Village distance (ref: 6.6 km)					
1.3 km	0.983	0.581	2.861	0.091	2.672
5.7 km	1.627	0.625	6.777	0.009**	5.087
3.9 km	0.562	0.588	0.914	0.339	1.755
1.9 km	-0.904	0.826	1.198	0.274	0.405
11.3 km	-0.377	0.831	0.206	0.650	0.686
27.4 km	-2.301	0.762	9.112	0.003**	0.100
Economic activity (ref: Livestock)	0.198	0.351	0.317	0.573	1.219
Education (ref: Secondary+)	-1.397	0.449	9.699	0.002**	0.247
Age category (ref: >50)					
18–30	-1.439	0.462	9.696	0.002**	0.237
31–40	-0.192	0.448	0.184	0.668	0.825
41–50	0.107	0.529	0.041	0.840	1.113
Gender (ref: Female)	0.086	0.330	0.067	0.795	1.090

DISCUSSION

This study provides important insights into how socio-demographic and contextual factors shape local communities' perceptions of elephant crop raiding and their willingness to support elephant conservation in areas adjacent to protected landscapes in Tanzania. The findings highlight the complexity of human–elephant interactions and demonstrate that conservation attitudes are not driven by a single factor, but rather by an interplay of livelihoods, education, age, spatial proximity to protected areas, and the type of protected area itself.

The type of protected area emerged as a key determinant of conservation support. Communities neighbouring LMNP showed a significantly higher likelihood of supporting elephant conservation compared to those living near SGR. This disparity is likely linked to differences in the visibility and distribution of benefits derived from conservation. National Parks in Tanzania are often closely associated with tourism revenue, employment opportunities, and community development initiatives, which can positively influence local perceptions of wildlife conservation. In contrast, Game Reserves tend to generate fewer direct benefits for surrounding communities, which may foster feelings of exclusion and

reduce incentives to support conservation initiatives (Mojo *et al.*, 2020; Ranke *et al.*, 2023). When conservation is perceived as imposing costs without corresponding benefits, local resistance is more likely to occur.

Village location and distance from protected areas also played a significant role in shaping conservation attitudes (Andersson *et al.*, 2017). Communities located at moderate distances from protected areas demonstrated higher levels of support for elephant conservation than those either very close to or far from reserve boundaries. Villages situated close to protected areas often experience frequent and severe human–elephant conflicts, particularly crop raiding, which can overshadow any conservation benefits. Conversely, communities located farther away may have limited interaction with wildlife and reduced exposure to conservation programmes, leading to weaker conservation engagement. These findings suggest that spatial context strongly mediates how communities perceive both the costs and benefits of living alongside protected areas.

Education level was another influential factor, with respondents who had attained only primary education showing lower levels of support for elephant conservation. This pattern suggests that limited formal education may constrain

understanding of the broader ecological and long-term benefits of wildlife conservation. Individuals with lower educational attainment may prioritise immediate livelihood security over abstract conservation goals, particularly when wildlife is perceived as a direct threat to food production and income. This underscores the importance of conservation education and targeted awareness programmes that communicate conservation benefits in locally relevant and accessible ways (Mbise *et al.*, 2021; Ardoin *et al.*, 2023).

Age differences further influenced attitudes toward conservation (Mbise & Sosiya, 2023). Younger respondents were less likely to support elephant conservation compared to older individuals. This may reflect generational differences in priorities and experiences (Liu *et al.*, 2022). Younger people may face greater economic pressures, land scarcity, and employment challenges, making them more sensitive to the immediate costs associated with wildlife presence. Older community members, on the other hand, may possess greater historical knowledge of environmental change and a stronger appreciation of the long-term value of conserving wildlife and ecosystems. These findings highlight the need for conservation strategies that resonate with younger generations by linking conservation goals to tangible socio-economic opportunities. Although gender did not significantly influence conservation support in this study, the overall patterns observed suggest that human–elephant conflict remains a central driver of local perceptions. Frequent crop raiding undermines household food security and economic stability, which in turn erodes positive attitudes toward conservation. Without effective mitigation measures and fair compensation mechanisms, local tolerance of elephants is likely to decline, regardless of broader conservation objectives.

CONCLUSION

In conclusion, this study provides an in-depth understanding of the social and contextual factors that shape local communities'

perceptions of elephant crop raiding and their support for elephant conservation in areas neighbouring protected landscapes in Tanzania. By comparing communities adjacent to a National Park and a Game Reserve, the study demonstrates that conservation attitudes are neither uniform nor random, but are strongly influenced by the nature of the protected area, spatial proximity, and key socio-demographic characteristics.

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ASSESSMENT ON IMPACTS OF CROP RAIDING ELEPHANTS TO FOOD AVAILABILITY AND ACCESSIBILITY TO MIKUMI FARMERS

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ABSTRACT

Local communities living in elephant-dominated landscapes bear significant costs, primarily from crop raiding elephants which potentially impact food availability and access for farmers. Crop losses from elephant damage have been well documented in villages adjacent to Mikumi National Park. However, empirical data on the impacts of crop damage on food availability and access remain limited. This paper examines the impacts of elephant crop damage on food availability and access among local communities. Primary data on crop damage were collected from July to August 2023 from 405 households in 10 villages adjacent to Mikumi National Park. Descriptive analysis was conducted to produce a summary table and frequency distribution graphs for visualizing the extent of crop damage and the proportions of respondents who replanted damaged crops. The study found that the most affected crops by acreage were maize (102.5 acres), pigeon peas (56.7 acres), sorghum (36.3 acres), and paddy rice (28.35 acres). The mean crop loss per household was 27.31 acres, representing 42.06% of cultivated farms. There was no statistically significant relationship between the cumulative area damaged and crop preferences. Only 7% (n=28) of the affected households replanted their crops. These findings suggest that crop losses and limited crop replanting may reduce households' food availability and access, potentially prompting farmers to shift toward less palatable crops as a trade-off between food security and elephant conservation.

Keywords: Crop damage, Crop replanting, Food accessibility, Food availability, Reduced agricultural Output.

INTRODUCTION

Human–elephant conflicts (HEC) remain among the most pressing conservation and livelihood challenges across Africa, particularly in areas where agricultural land borders protected areas (Hoare, 2015; Nyhus, 2016). Among the various forms of HEC, crop raiding by African elephants (*Loxodonta africana*) is the most prevalent and economically damaging, threatening both rural livelihoods and community support for wildlife conservation (Songhurst *et al.*, 2016; Mukenka *et al.*, 2018). In Tanzania, crop raiding is widespread in villages adjacent to elephant dominated ranges, where elephants frequently move beyond core protected areas in search of food and water (TAWIRI, 2020; Mutanga *et al.*, 2022).

Crop loss through elephant damages directly undermines food security, which is referred to as a situation of having access to quantity and quality food (FAO, 2009). Food security rests on availability, accessibility, utilization, and stability; all of which can be disrupted by elephant crop damage. The most immediate effects of crop damage are food availability (quantity) and food accessibility (the ability of households to obtain food through own production and or/ purchase) (WFP, 2017).

When elephants destroy staple crops such as maize, rice, or bananas which are key components of household diets in rural Tanzania results in significant yield loss, sometimes exceeding 30–50% in high-conflict zones (Mabula, 2017; Mayengo *et al.*, 2017; Mutanga

et al., 2022). These losses reduce household food stocks and often force families to purchase food at higher market prices, thereby reducing accessibility, particularly among resource poor farmers (Hoare, 2015; Hahn *et al.*, 2017). Moreover, repeated crop losses can diminish household resilience, leading to chronic food insecurity and dependence on food aid during lean seasons (Kitsuki and Sakurai, 2023; Tofu *et al.*, 2023).

In human-elephant dominated ecosystems, villages bordering the park experience frequent raids, often occurring at night and during crop maturation periods (Mukeka *et al.*, 2018; Mutanga *et al.*, 2022). For farming households around Mikumi National Park (MNP), crop damage by elephants represents loss of staple crops reducing food self-sufficiency and disrupts seasonal food supply cycles.

MNP, part of the expansive Nyerere-Selous-Mikumi ecosystem, hosts one of the largest populations of elephants (Lohay *et al.*, 2020), and experienced a rapidly expanding agriculture and other human activities (Malley and Gorenflo, 2023; Rutta, 2023). Despite growing documentation of human–elephant conflicts in Tanzania, few studies have quantitatively assessed how crop-raiding affects household-level food availability and access in the area. Addressing this gap is crucial for designing effective mitigation on food security by evaluating how elephant-induced crop losses influence food production, household food stocks, and access to alternative food sources among Mikumi farmers. These losses directly undermine food availability, as households lose their primary source of sustenance, and indirectly affect food accessibility, as reduced harvests limit household purchasing power and force reliance on food markets (FAO, 2009; WFP, 2017).

This study seeks to assess the impacts of crop-raiding elephants on food availability and accessibility among farming communities adjacent to Mikumi National Park, Tanzania by; (1) evaluating elephant induced crop losses, (2) Assessing replanting damaged crops on household food availability and accessibility as important dimensions towards food availability and accessibility in the farming communities.

METHODOLOGY

Study area

Mikumi National Park lies in south-eastern Tanzania in Morogoro Region, forming part of the greater Selous–Mikumi landscape and connecting to neighboring eastern-arc mountain blocks (Fig. 1). Its position in the Selous ecosystem makes Mikumi an important node for regional elephant movement and human–wildlife interactions (Gunn *et al.*, 2013). The climate is tropical with a clear wet season spanning from December to May and a pronounced dry season spanning from June to November. Communities bordering Mikumi are predominantly smallholder farmers practicing rain-fed subsistence agriculture supplemented by petty trade, casual labour and small livestock (Mayengo *et al.*, 2017). Common food and cash crops reported in the literature and government baseline assessments include maize, paddy rice, sorghum, cassava, beans, groundnuts, and some horticultural/cash species (sesame, sunflower); cropping patterns vary by elevation and access to wetland farm sites (lowland rice vs upland maize/cassava) (Kahimba *et al.*, 2015).

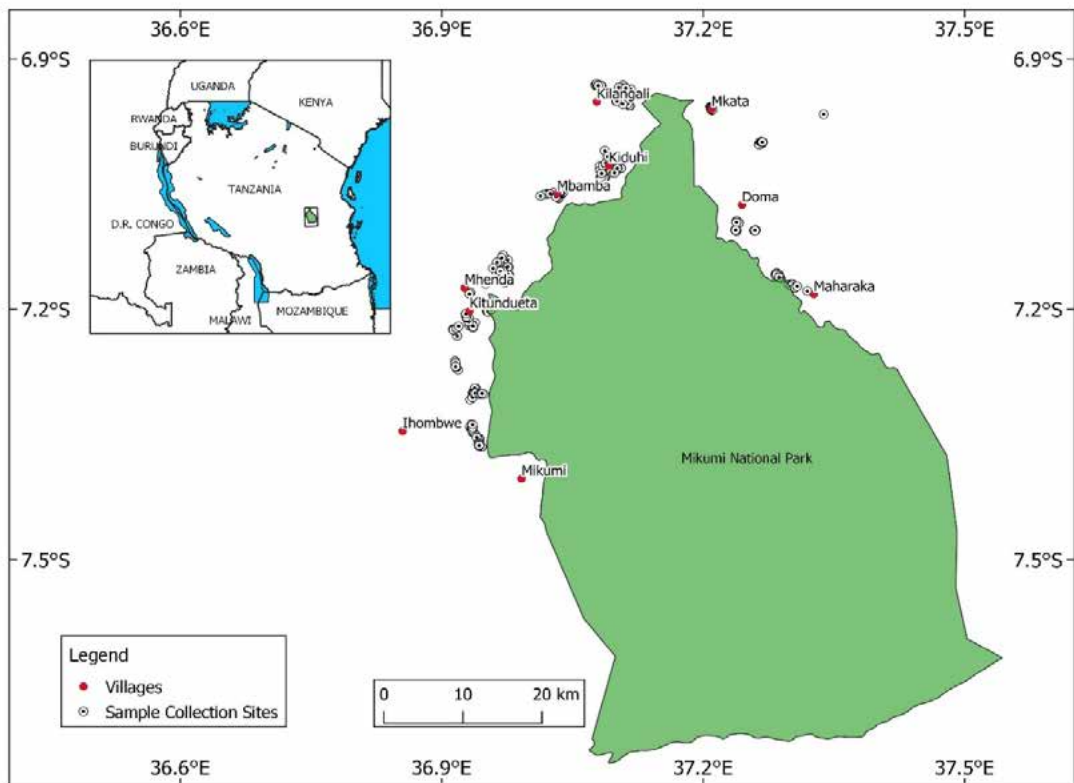


Figure 1: Locations of sampled households in ten surveyed villages adjacent to Mikumi National Park (July -August 2023)

Study design

This study adopted a cross-sectional observation study design using closed and open-ended questionnaires through household questionnaire surveys in July-August 2023 immediately after the main cropping season (typically December-June), when farmers can recall and estimate crop damage levels and food stocks to be able to assess the impacts of elephant crop-raiding on household food availability and accessibility among communities adjacent to Mikumi National Park. Similar designs have been used in related studies across Tanzania, Kenya, and Botswana to capture both the measurable extent of crop loss and the socio-economic perceptions of affected households (Graham *et al.*, 2010; Songhurst *et al.*, 2016; Mutanga *et al.*, 2022). We sampled 405 households using Cochran's formula (1977), $N_0 = (Z^2 * P * Q) / e^2$ where N_0 is required the sample size, ($Z=1.96$ at 95% confidence), P is the estimated

proportion of the population (0.5) to maximize the sample size, ($Q = 1 - P$), e is the margin of error (0.05) at 5% (Uarkarn *et al.*, 2021). Households were picked through systematic random sampling from village registers, with sampling intervals determined by dividing the number of households per village approximated to be 40. Household selection started with the nearest household to the village office from village register, following sampling intervals for each village to attain the desired sample. Questionnaires were pretested to 20 village leaders' households in Mikumi, Maharaka, and Mhenda villages; these households were not involved during data collection. Further, questionnaires were translated into Kiswahili to ensure clear communication with participants. Interviews were conducted to either household heads, spouses or any adult over 18 years who had resided with the household for at least one year.

Data Collection Methods

Households questionnaires collected information on respondents and household demographics, and respondents were asked whether their households had experienced crop raiding by elephants. For households which experienced crop raiding were asked on the proportion of the farm (acres) destroyed by elephants in relation to the size of the farms cultivated. Further, victims whose crops were destroyed were asked if they had replanted the destroyed crops and further probed where households sourced food for the rest of the season following elephant destruction of their farms. Respondents were further asked to state elephant preference by ranking the crops cultivated on a 5-point Likert scale from 1 – Highly preferred by elephants to 5 – Not preferred at all to assist in relating elephant crop raiding and preference.

Data Analysis

A descriptive analysis was conducted using R version 4.4.2, graphical representations were generated for enhanced visualization and clarity (McGuinness and Sheppard, 2021), to create a summary table for visualization of the proportions of crop farms destroyed. Further frequency distribution charts on proportions of respondents who replanted damaged crops, and sources of foods where households obtained foods for the rest of the season/year. Shapiro-

Wilk test was used to confirm the normality assumption of data. The Spearman's Correlation test was used to test the relationship between the cumulative area damaged and elephant crops preference rankings.

RESULTS

Crop losses from elephants

Table 1 shows quantitative estimates from household assessments of the proportions of crop fields destroyed by elephants in Mikumi villages. Main staple foods destroyed included Maize 105.52 acres making 60.4% of the cultivated area, Maize 186.1 acres (60.4% of the cultivated farms), Pigeon peas 56.36 acres (56.73% of the cultivated area), Sorghum 36.25 acres (44.34%), and Paddy 28.35 acres (45.73%). Overall crop damage by raiding elephants was estimated at 36.67 acres which accounted for 45.6% of the cultivated farms (80.49 acres). Most households reported significant reductions in harvests and household food outputs representing a substantial decline in available food reserves. The Spearman's correlation test on the relationship between the cumulative area damaged and elephant crops preferences ($\rho=0.066$, $P=0.187$), indicating a weak positive and statistically non-significant relationship. These results suggest that the size of the crop damaged was not significantly associated with elephants' preference rankings in the surveyed area.

Table 1: Proportions of crop farms destroyed by elephants in Mikumi villages (July-August 2023)

Crop	Size of farm cultivated (Acres)	Size of farm destroyed (Acres)	Percentage of farm destroyed (%)
Maize	169.74	102.52	60.40
Paddy	61.99	28.35	45.73
Sunflower	84.5	10.75	12.72
Beans	9	3.55	39.44
Tomato	68.1	34.26	50.31
Cow peas	9.5	5.8	61.05
Banana	42.8	17.97	41.99

Crop	Size of farm cultivated (Acres)	Size of farm destroyed (Acres)	Percentage of farm destroyed (%)
Pigeon peas	120.75	56.73	46.98
Papaya	7.25	2.94	40.55
Sorghum	81.75	36.25	44.34
Sugarcane	36	14.75	40.97
Cassava	68.54	13.88	20.25
Overall Mean	63.33	27.31	42.06

Replanting damaged crops

Figure 2 shows proportions of the respondents who replanted the damaged crops in Mikumi villages after experiencing losses. Majority of the respondents (93%, n=377) did not replant the damaged crops, while only 7% (n=28) did. Further, Figure 3 shows the proportions of respondents showing the sources of food for households after elephant crop damage. Results indicate that 55% (n=223) of respondents could afford to buy food and supplement with leftover foods in the field, while 45% (n= 182) depend entirely on remaining crops in the field. The findings revealed that replanting damaged crops was an obviously ineffective strategy among farmers affected by elephant raids in Mikumi villages. Almost all the respondents indicated that replanting efforts were constrained by seasonal timing in terms of rain, resource limitations, and repeated raids, leaving most farmers unable to recover lost yields with food scarcity and accessibility.

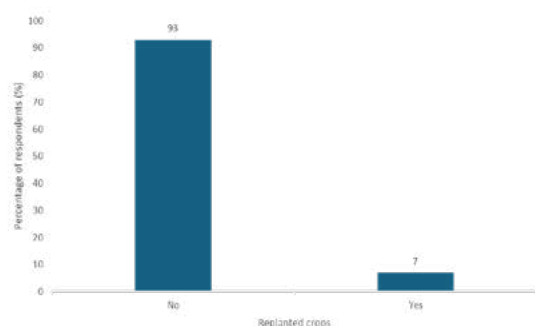


Figure 2: Proportions of respondents who replanted damaged crops in villages adjacent to Mikumi National Park (July-August 2023)

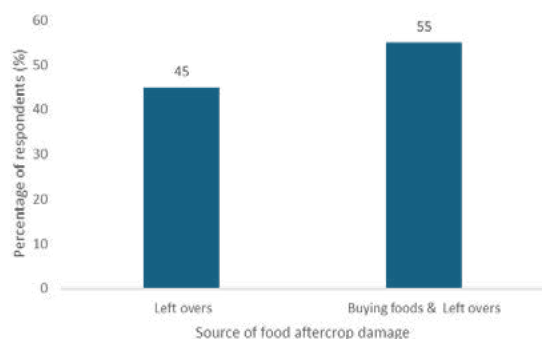


Figure 3: Respondents' sources of food after elephant crop raiding by Mikumi farmers (July – August 2023)

DISCUSSION

Crop losses from elephants

Results from this study suggest that crop losses result in economic and food insecurity outcomes affecting the wellbeing of households and communities in Mikumi villages. Therefore, elephant-induced losses translate directly into reduced food supply and immediate shortages, forcing early depletion of reserves and dependence on food purchases or borrowing (FAO, 2009; WFP, 2017). These results concur with studies demonstrating that elephants cause disproportionately high agricultural losses, both in frequency and in the magnitude of damage per incident (Hoare, 2015; Mukeka *et al.*, 2018; Mutanga *et al.*, 2022). Further empirical evidence across Africa indicates that elephant raids can destroy between 20% and 70% of household harvests, depending on proximity to park boundaries, crop type, and season (Graham *et al.*, 2010; Songhurst *et al.*, 2016).

In Tanzanian contexts such as Kilombero and Ruaha, farmers report losing entire fields of maize, rice, or bananas within a single night raid (Mabula, 2017; Mutanga *et al.*, 2022). Beyond such immediate losses, elephants also cause secondary effects on agricultural productivity where frequent raids discourage farmers from cultivating high-value or high-risk crops, leading to reduced planting area, delayed sowing, or complete abandonment of fields near the park (Naughton-Treves, 1998; Hoare, 2015). A study by Graham *et al.* (2010) in Kenya and Hahn *et al.* (2017) in Tanzania found that farmers in high-conflict zones cultivate up to 30% less land than those in safer areas, directly affecting local food availability.

Furthermore, elephants often trample crops without consuming them, compact soil, and damage irrigation infrastructure or grain storage, further reducing the effective yield. Such cumulative impacts reinforce the food insecurity cycle in elephant-adjacent communities like Mikumi. The consequences of elephant raids extend beyond physical food availability to food accessibility, defined as the ability of households to obtain sufficient food through production, purchase, or social networks (FAO, 2009). Elephant-induced crop losses diminish household incomes and purchasing power by eroding the primary source of cash and subsistence food. Studies in Tanzania, Botswana, and Zimbabwe have demonstrated that households experiencing repeated raids are more likely to sell assets, reduce meal frequency, or rely on relief assistance to bridge food gaps (Nyhus, 2016; Songhurst *et al.*, 2016).

Over time, the repeated crop raiding have implications for being long term on structural decline in agricultural productivity, lowering regional food output, undermining both subsistence food stocks and market earnings, forcing households to rely on food purchases, borrow grain, or reduce meal frequency.

Replanting damaged crops

The study learnt that most households depend on rain-fed agriculture as their primary source of food, replanting might not be in good timing of the rainy season, still replanting could not ensure growth of the planted crops. Only 7% (n= 28) of the surveyed farmers managed to replant the damaged crops. It was observed that elephants frequently destroy maturing crops during critical growing stages, where replanting is often a short-term response to an ongoing threat, rather than a sustainable solution to food insecurity.

Similar trends were reported in other Tanzanian protected area interfaces of Ruaha and Tarangire, where less than one-fifth of farmers succeeded in replanting damaged crops after raids (Hoare, 2015; Mutanga *et al.*, 2022). Replanting crops following elephant raids is one of the most common coping mechanisms used by farmers living adjacent to protected areas in Africa. However, the effectiveness of replanting as a recovery strategy is highly influenced by the timing of the damage, resource availability, and the frequency of raids (Hoare, 2015; Graham *et al.*, 2010). Most elephant incursions occur at the middle or late stages of crop development, when crops like maize, rice, or bananas are nearing maturity (Chiyo *et al.*, 2012; Hahn *et al.*, 2017).

However, replanting after such damage is constrained by seasonal rainfall patterns. In rain-fed agricultural systems like Mikumi, elephant raids that occur late in the season leave insufficient time for crops to mature before the rains cease (Mukeka *et al.*, 2018; Mutanga *et al.*, 2022). Apart from crop damage, replanting failures further reduce households' food availability and accessibility deepening vulnerability to hunger.

CONCLUSION AND RECOMMENDATIONS

This study reveals that crop raiding by elephants imposes substantial and

multidimensional challenges to food security and livelihoods of Mikumi farmers. It reduces household food production, depletes food reserves, and limits both physical and economic access to food for farming communities. Overall, the findings confirm that crop-raiding elephants are a significant driver of rural food insecurity in Mikumi. Without effective mitigation, households remain trapped in cycles of production loss, forcing many households to adopt short-term coping mechanisms such as replanting damaged fields, reducing food insecurity and/or increasing reliance on purchased food. This demonstrates that elephant-induced crop losses are not only ecological nuisances, but they are also central drivers of livelihood vulnerability and nutritional deprivation.

Addressing this challenge requires integrated strategies that balance conservation objectives with human livelihoods needs. Potential mitigation measures include encouraging farmers to cultivate less palatable crops as a trade-off between food security and elephant conservation.

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COMMUNITY ENGAGEMENT THROUGH OUTREACH INITIATIVES FOR VILLAGES LIVING ADJACENT TO TARANGIRE NATIONAL PARK

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ABSTRACT

Outreach programmes have increasingly been adopted as a strategy to strengthen relationships between protected areas and neighbouring communities, particularly in regions where conservation objectives intersect with local livelihoods. This study assessed the influence of outreach initiatives implemented by Tarangire National Park on local people's perceptions, attitudes, and levels of cooperation with park management in northern Tanzania. Data were collected through household surveys conducted in villages adjacent to the park, complemented by key informant interviews. Descriptive statistics and inferential analyses were used to examine community awareness of outreach activities, perceived benefits, and overall attitudes towards the park. The findings indicate that a majority of respondents were aware of outreach programmes and associated them with improvements in social services such as education, water supply, and health facilities. Households that reported direct or indirect benefits from outreach interventions exhibited more positive attitudes towards the park and expressed greater support for conservation efforts. However, perceptions varied among villages, with communities located closer to the park or more frequently engaged in outreach activities reporting higher levels of satisfaction. Despite these positive outcomes, some respondents felt that benefits were unevenly distributed and insufficient to fully offset conservation-related costs. The study concludes that outreach programmes play a significant role in enhancing park–community relations, but their effectiveness depends on transparency, equitable benefit sharing, and sustained engagement with local stakeholders.

Keywords: Benefit sharing, Community engagement, Conservation support, Outreach initiatives, Tarangire ecosystem

INTRODUCTION

Protected areas are a central pillar of biodiversity conservation worldwide, yet their success is often influenced by the nature of relationships between park authorities and neighbouring communities (Figueroa & Aronson, 2006; Mavah *et al.*, 2018). In many parts of sub-Saharan Africa, national parks are embedded within landscapes that support human settlements and livelihood activities, creating conditions in which conservation goals may conflict with local socio-economic needs (Kideghesho, 2008). Where such tensions exist, negative attitudes towards

protected areas can undermine conservation effectiveness and contribute to resource-use conflicts (Dickman, 2010; Htay *et al.*, 2022). To address these challenges, conservation agencies have increasingly adopted community outreach programmes aimed at improving park–people relationships (Kideghesho, 2010; Bennett, 2016;). Outreach initiatives typically include investments in social infrastructure, education, health services, and awareness campaigns designed to enhance local understanding of conservation objectives (Emerton & Mfunda, 1999; Kideghesho, 2006). By delivering tangible benefits to adjacent communities, outreach programmes seek to foster positive

perceptions of protected areas, encourage compliance with conservation regulations, and reduce antagonism arising from restrictions on resource access or wildlife-related costs (Mbise & Røskoft, 2021).

In Tanzania, outreach programmes have become an integral component of national park management, particularly in areas where local livelihoods are closely linked to natural resources (Newmark & Hough, 2000). Tarangire National Park (TNP) represents a prominent example, as it is surrounded by villages that depend on agro-pastoral activities and are directly affected by wildlife presence and land-use regulations (Mbise *et al.*, 2021). The park has implemented various outreach interventions over time, with the intention of improving community welfare while strengthening local support for conservation.

Despite the widespread adoption of outreach strategies, empirical evidence on their effectiveness remains mixed. Some studies report improved attitudes and enhanced cooperation between communities and park authorities (Ebua *et al.*, 2011; Burgoyne & Mearns, 2017; Abukari & Mwalyosi, 2020), while others highlight persistent dissatisfaction, perceived inequities, or limited awareness of outreach benefits (Hall, 2007; Karanth *et al.*, 2012; Kicheleri *et al.*, 2021).

Therefore, understanding how local people perceive outreach initiatives, and whether these perceptions translate into positive attitudes towards conservation, is therefore critical for improving programme design and implementation.

MATERIAL AND METHODS

Study area

The study was conducted in villages (Kwaraa, Mwikantsi, Nkaiti, and Olasiti) located adjacent to TNP (Figure 1). The park forms part of a broader savannah ecosystem characterised by seasonal rainfall, with a distinct wet season and a prolonged dry period (TANAPA, 2021). Tarangire National Park supports a wide range of wildlife species and is surrounded by rural communities whose livelihoods are largely dependent on agro-pastoral activities, including crop cultivation and livestock keeping. The neighbouring villages experience varying degrees of interaction with the park due to differences in distance from the park boundary, access routes, and levels of engagement with park outreach programmes (Mbise *et al.*, 2021). These spatial differences provided an appropriate setting for examining how outreach interventions influence community perceptions and attitudes towards the park (Fig 1).

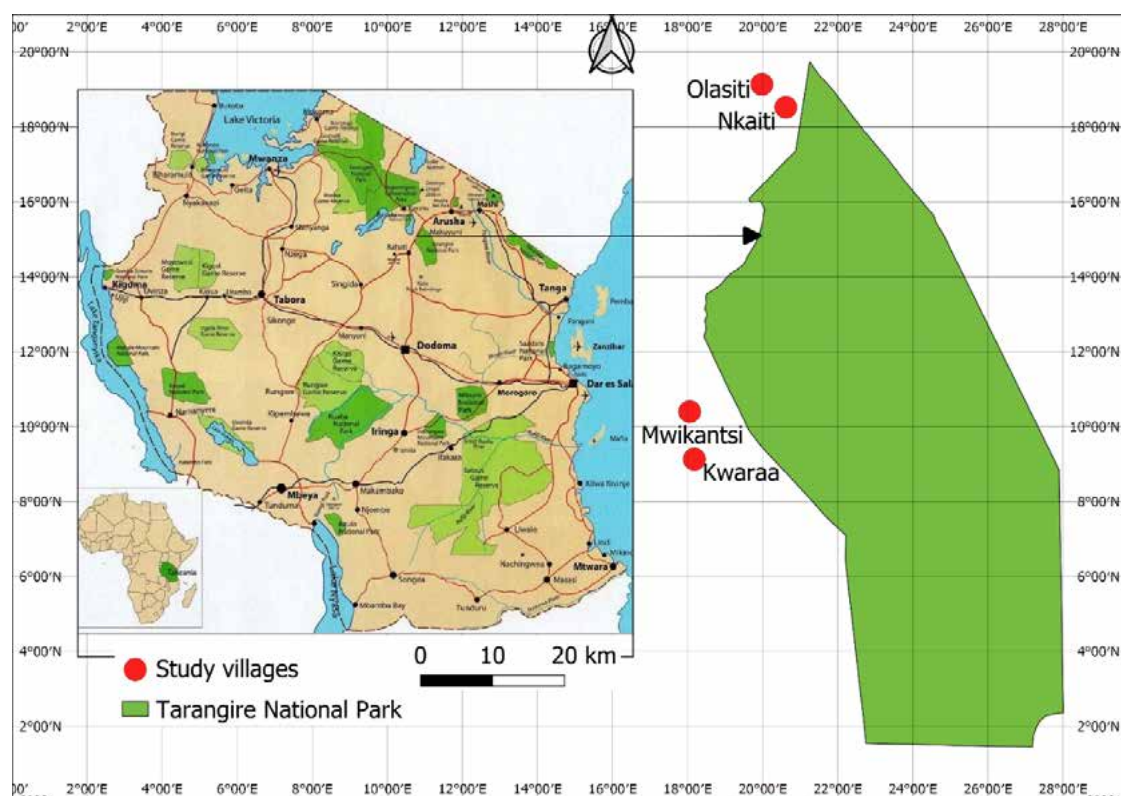


Figure 1: Map of Tarangire National Park and study villages

Data collection

Primary data were collected through household surveys using structured questionnaires administered via face-to-face interviews. Households were selected using a systematic sampling approach within each village to ensure broad representation. Only adult respondents aged 18 years and above were interviewed, and participation was voluntary.

The questionnaire captured information on respondents' socio-demographic characteristics (Table 1), awareness of park outreach programmes, perceived benefits received from these initiatives, and overall attitudes towards TNP. In addition, key informant interviews were conducted with village leaders and park officials to supplement household-level data and provide contextual insights into outreach implementation.

Table 1: Socio-demographic characteristics of respondents in villages adjacent to Tarangire National Park (N = 180)

Variable	Category	Frequency (n)	Percentage (%)
Gender	Male	122	67.8
	Female	58	32.2
Age group (years)	18–30	49	27.2
	31–40	61	33.9
	41–50	38	21.1
	≥51	32	17.8

Variable	Category	Frequency (n)	Percentage (%)
Education level	No formal education	6	3.3
	Primary education	71	39.4
	Secondary education	86	47.8
	College/University	17	9.5
Main livelihood	Crop farming	52	28.9
	Livestock keeping	48	26.7
	Mixed farming	80	44.4

Notes: Percentages are calculated based on total respondents (N = 180).

Each respondent was classified into one category per variable.

Data analysis

Collected data were coded and analysed using statistical software. Descriptive statistics were used to summarise respondent characteristics, levels of awareness of outreach programmes, and perceived benefits. Frequencies and percentages were calculated to illustrate patterns of community engagement and attitudes towards the park. Inferential analyses were conducted to examine relationships between awareness of outreach programmes, perceived benefits, and attitudes towards conservation. Comparisons among villages were used to assess spatial variation in perceptions, while significance testing was applied to identify meaningful differences across respondent groups. Qualitative information obtained from key informant interviews was analysed thematically and used to support the interpretation of quantitative findings.

RESULTS

Awareness of outreach programmes

The majority of respondents indicated that they were aware of outreach initiatives implemented by TNP (74.4%, n=180, Table 2). Awareness levels, however, varied among villages, with communities located closer to the park or more frequently engaged by park staff reporting higher levels of familiarity with outreach activities. Respondents identified several forms of outreach support, including assistance

to schools, improvement of water sources, provision of health-related services, and conservation education (Table 3). Education-related support, such as classroom construction and provision of learning materials, was among the most frequently mentioned benefits (38.3%). Water supply projects, including boreholes and water tanks, were also commonly cited (28.9%), particularly in villages experiencing seasonal water shortages. Health-related assistance was less frequently reported but was nonetheless viewed positively where it occurred.

Table 2: Community awareness of outreach programmes implemented by Tarangire National Park (TNP)

Response	Frequency (n)	Percentage (%)
Aware of outreach programmes	134	74.4
Not aware	46	25.6

Table 3: Perceived benefits of outreach programmes reported by respondents

Type of benefit	Frequency (n)	Percentage (%)
Education support	69	38.3
Water supply projects	52	28.9
Health services	31	17.2
Conservation education	28	15.6

Perceptions on benefit satisfaction

The binary logistic regression analysis identified several socio-demographic factors influencing respondents' satisfaction with conservation activities implemented through Community Outreach. Age and education level emerged as the most important predictors, while village distance, gender, and economic activity showed no significant effects (Table 4). Age was negatively associated with satisfaction. Compared to younger respondents, middle-aged individuals were less likely to be satisfied with conservation activities, with

the effect being marginally significant ($p = 0.054$). The effect was stronger and statistically significant for elders ($p = 0.008$). Education level also had a significant negative influence on satisfaction. The strongest effect was observed among primary school graduates ($p = 0.001$), suggesting that increased educational attainment may enhance understanding of conservation objectives and perceived benefits, thereby improving satisfaction with outreach initiatives. However, village distance from the protected area, gender, and economic activity did not significantly influence satisfaction.

Table 4: Binary logistic regression results examining determinants of respondents' satisfaction with conservation activities implemented through Community Outreach.

Predictor	Estimate	SE	z value	p-value
Intercept	1.34	0.78	2.92	0.087
Village distance (km)	-0.09	0.15	0.40	0.524
Age category				
Middle-aged	-1.33	0.69	3.70	0.054*
Elder	-1.97	0.74	7.07	0.008**
Gender (Male)	0.59	0.39	2.31	0.128
Education level				
Never attended school	-1.91	0.77	6.16	0.013*
Primary graduate	-1.40	0.43	10.79	0.001**
Economic activity				
Farming	-0.46	0.72	0.41	0.522
Pastoralism	-1.48	1.15	1.65	0.199

Relationship between outreach initiatives and conservation support

Analysis revealed a positive association between awareness of outreach programmes

and support for conservation. Villagers with primary education or higher were more likely to support conservation initiatives and were more willing to report poaching activities to park authorities (Table 5).

Table 5: Results from a binary logistic regression model examining factors influencing whether local residents are willing to report poaching activities to park authorities (coded as yes/no).

Variable	Estimate	SE	z	P value
Intercept	-1.98	0.68	8.36	0.004
Distance from village (km)	0.23	0.12	3.60	0.058

Variable	Estimate	SE	z	P value
Age group (reference: Youth)				
Middle-aged	0.87	0.59	2.16	0.141
Elder	0.82	0.61	1.79	0.181
Gender (reference: Female)				
	0.47	0.34	1.88	0.170
Education level (reference: Secondary graduate)				
No formal schooling	0.95	0.57	2.79	0.094
Primary education	0.83	0.37	4.82	0.028*
Primary livelihood (reference: Other ¹)				
Farming	-1.07	0.59	3.28	0.070
Pastoralism	-0.39	0.69	0.32	0.569

DISCUSSION

The results of this study indicate that outreach programmes implemented by TNP play an important role in shaping local community perceptions and attitudes towards conservation. High levels of awareness of outreach initiatives among respondents suggest that the park has made visible efforts to engage neighbouring villages. Such engagement appears to contribute to more favourable views of the park, particularly where communities perceive tangible benefits that address local development needs.

Education and water-related interventions emerged as the most recognised forms of outreach support (Ardoin *et al.*, 2023). These types of investments are highly valued in rural settings where access to basic social services is often limited. By responding to community priorities, outreach programmes may help to offset some of the perceived costs associated with living near a protected area (Mbise & Røskaft, 2021), such as restricted access to resources or wildlife-related impacts (Ulimboka *et al.*, 2025). This finding aligns with broader conservation literature emphasising the importance of linking conservation objectives with local development outcomes (Dickman, 2010; Karanth *et al.*, 2012; Ardoin *et al.*, 2020; Htay, 2020).

Despite generally positive perceptions, the findings also highlight concerns regarding the distribution and adequacy of outreach benefits (Kideghesho, 2008; Kicheleri *et al.*, 2021). Some respondents expressed dissatisfaction with the level of support received or perceived inequities among villages (Kicheleri *et al.*, 2021). Such perceptions can undermine trust and weaken the effectiveness of outreach programmes if not addressed. Uneven benefit sharing has been identified in other protected area contexts as a source of persistent tension between conservation authorities and local communities (Abukari & Mwalyosi, 2020).

Variation in attitudes among villages underscores the influence of spatial and institutional factors on community–park relations (Mbise *et al.*, 2021). Villages with closer proximity to the park or more frequent interactions with park staff reported stronger positive attitudes, suggesting that sustained engagement and communication are critical for building trust. On the other hand, limited contact with park authorities may reduce awareness of outreach activities and diminish their perceived relevance.

The observed positive association between outreach awareness and support for conservation suggests that outreach programmes can contribute to improved cooperation between

communities and park management (Holmes, 2003; Mashauri, 2017). However, outreach alone is unlikely to fully resolve conservation-related conflicts. Long-term success depends on transparent decision-making, inclusive participation, and continuous dialogue that allows communities to voice concerns and influence programme design.

CONCLUSION

In conclusion, outreach initiatives represent a vital tool for improving park–community relations and promoting conservation in human-dominated landscapes. Enhancing their effectiveness requires sustained investment, inclusive planning, and adaptive management that responds to community needs. Such approaches are likely to contribute to more resilient conservation outcomes and improved coexistence between protected areas and surrounding communities in northern Tanzania and beyond.

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INTEGRATING AI TECHNOLOGIES WITH INDIGENOUS PRACTICES TO MITIGATE HUMAN-WILDLIFE CONFLICTS IN COMMUNITIES ADJACENT TO RUAHA NATIONAL PARK

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ABSTRACT

Human-Wildlife conflicts (HWC) near Ruaha National Park endanger local communities, jeopardising food security, safety, and environmental balance. Climate change induced changes such as unpredictable rainfall, deforestation, land use disputes, and unpeaceful human-wildlife interactions all contribute to escalation of these conflicts. Despite increased acknowledgement of Artificial Intelligence (AI) technology's potential, few studies have experimentally investigated its use in HWC management, particularly in protected areas such as Ruaha National Park. This study investigates how Artificial intelligence techniques can be used to manage elephant movements, protect crops, and encourage sustainable land-use practices, ultimately lowering crop loss and increasing climate resilience in the region. Using a mixed-methods approach, data were obtained through interviews, focus group discussions, surveys, and direct observations, resulting in a thorough grasp of the region's socioeconomic and environmental dynamics. The findings indicate that combining AI technologies with conventional knowledge systems can improve human-animal coexistence, enable adaptive agriculture methods, and promote more effective wildlife management. This study adds to the growing body of literature on creative HWC solutions by showing the potential of AI to support sustainable livelihoods and ecological stability in the Ruaha-Rungwa ecosystem.

Keywords: AI Technologies, Indigenous Practices, Human-Wildlife Conflicts, Communities, Ruaha National Park

INTRODUCTION

Human-Wildlife Conflict (HWC) poses significant challenges to both farmers' livelihoods and wildlife conservation worldwide (Liang, Wang and Xie, 2025). With rapid human population growth, the expansion of agricultural frontiers and intensified nature conservation efforts, the frequency and severity of HWC have escalated, making it a pressing issue in wildlife management governance worldwide (Mekonen, 2020; Li *et al.*, 2023). Fundamentally, HWC arises when the needs and behaviors of wildlife negatively affect humans, or when human activities threaten wildlife survival (Barua *et al.*, 2013). The nature of human-wildlife co-existence depends upon their interaction, which may range from

positive to neutral to negative (Frank, Glikman & Marchini, 2019). It is a negative interaction where anthropogenic losses or losses of wildlife species occur, generally referred to as conflict or human-wildlife conflict (HWC) (Peterson *et al.*, 2010; Nyhus, 2016; Bhatia *et al.*, 2010).

Rapidly increasing human populations and their needs have led to large-scale land-use changes. Land use here entails farming, pastoralism, irrigation and fishing, to list some. Landscape-level changes and the transformation of habitats are significant concerns for large and wide-ranging mammals such as tigers, leopards, and elephants. However, in current scenario of increasing anthropogenic pressure on forests and loss of traditional ways of subsistence has led to strenuous conditions for human-wildlife

co-existence, sometimes turning to conflict due to negative human wildlife interaction (HWI). Protecting wide-ranging large carnivores is difficult because they frequently prey on valuable livestock on community lands and areas close to protected areas (Holland *et al.*, 2018; Mustăţea & Pătru-Stupariu, 2021; Brackowski, 2023).

Large carnivores are territorial species and occur at low population densities, requiring vast extant habitats, thus becoming more vulnerable to habitat loss (Poor *et al.*, 2020). Studies have highlighted the importance of designation of multi-use landscapes for conserving large carnivores, especially in regions where carnivore ranges overlap with high human-density areas (Kshetry *et al.*, 2017; Naha, *et al.*, 2020). One of the critical tasks in carnivore conservation is identifying priority human-carnivore conflict sites and their underlying reasons for effective mitigation strategies (Miller, 2015).

This study investigates the integration of AI Technologies with indigenous practices to mitigate Human–Wildlife Conflicts as we hypothesise that human-induced habitat changes are one of the critical drivers and factors for negative HWC. We assume that rapid changes in habitat features and quality induces unfavourable conditions for large mammals, increasing the chances of negative interaction with humans. This paper address three specific objectives; to assess the predominant types and nature of human-wildlife conflicts experienced by communities surrounding Ruaha National Park, to document indigenous knowledge practices currently employed to manage and prevent wildlife incursions, and to evaluate the effectiveness of AI-based technologies when integrated with indigenous practices in reducing human-wildlife conflicts. Understanding the relationship between habitat modifications and HWI is essential to strategize the effective and efficient long-term conservation and co-existence of humans and large mammals.

THEORETICAL LITERATURE REVIEW

The theories underpinning this study are diffusion of innovation theory and the Human-Environment Interaction Theory. This paper adopted diffusion of innovation theory propounded by Everett Rogers in 1962. This theory describes how innovation (new ideas, technologies, product or practice is perceived by the society) is communicated, adopted and spread within a social system over time. The theory is guided by several assumptions: (i) An innovation must be perceived by individuals within a social system to diffuse and use it regardless of how long it has existed; (ii) The spread of an innovation depends on the way information is spread through various channels such mass media, interpersonal networks, and digital platforms; (iii) The norms, values, and structure of a social system influence how quickly and widely an innovation is adopted, (iv) Individuals adopt innovations at different rates such as early adopters, late adopters and laggards.

Despite the assumption of the Diffusion of Innovations Theory, the criticisms of the theory are: (i) The theory assumes that all innovations are beneficial and should be adopted, overlooking the possibility that some innovations may be harmful, unnecessary, or culturally inappropriate, (ii) It tends to place responsibility on individuals for not adopting innovations, rather than considering systemic barriers such as poverty, lack of access, or institutional resistance, (iii) The theory often applies a Western-centric lens and may not account for cultural, social, or economic differences that influence adoption in diverse communities, (iv) The model presents a linear and predictable adoption curve, which may not reflect the complex, iterative, and sometimes chaotic nature of real-world innovation adoption, (v) It underemphasizes how users often adapt or modify innovations to suit their needs and how these adaptations can influence the innovation's evolution and further diffusion. Diffusion of Innovations Theory advocates that, in agriculture, innovation promote adoption of

new farming techniques, seeds or equipment. Innovation helps predict droughts, determine resistance crops and plan interventions to accelerate adoption.

How diffusion of innovation theory fit in this study

The Diffusion of Innovations Theory provides a valuable framework for understanding how technological interventions can be adopted by farmers adapting to climate change and wildlife challenges in the Ruaha-Rungwa ecosystem. In the context of this study, innovation refers to the technological tools and practices introduced to help farmers mitigate the impacts of climate variability and human-wildlife conflict. These could include early warning systems, climate-smart agriculture techniques, mobile-based weather forecasting, or digital platforms for market access and wildlife tracking. Farmers adopt innovation if it gives them the intended benefits such as reducing wild eating their crops, damage their crops, damage properties and killing people. If farmers see clear benefits, such as reduced crop loss from wildlife or improved yields despite erratic rainfall, they are more likely to adopt and sustain innovations. By aligning the intervention with community values and involving local stakeholders in the dissemination process, this study will enhance the effectiveness and sustainability of its outcomes. Thus, the Diffusion of Innovations Theory offers a strategic lens to guide the design, implementation, and scaling of technology-driven solutions in the Ruaha-Rungwa Ecosystem.

The second theory for this study is Human-Environment Interaction Theory. The theory emphasizes the reciprocal relationship between human actions and the environment. It recognizes that human activities can affect the environment, and conversely, environmental changes can affect human societies. In the context of Climate Smart Agriculture (CSA) theory it highlights the importance of

considering social, economic, and cultural factors in the adoption and implementation of climate-smart technologies and practices. Human-Environment Interaction Theory explores how human activities affect and is affected by the environment. This theory is relevant for understanding how farmers interact with wildlife and climate conditions in the Ruaha-Rungwa ecosystem. Human-environment interaction theory helps us understand the reciprocal relationship between humans and their environment, guiding the development of CSA practices that are both effective and socially acceptable.

MATERIALS AND METHODS

Description of the study area

This study was undertaken in the villages surrounding Ruaha National Park (RUNAPA) (Fig.1). The Ruaha-Rungwa Ecosystem is being targeted because the ecosystem is a critical elephant stronghold and home to Tanzania's largest elephant population (Juma *et al.*, 2025). The RUNAPA is located between 7°00' S 8°00'9 S, and 33000' E 36000' E (George and Kangalawe, 2024). Ruaha National Park is among Tanzania's largest national parks and one of the most significant ecoregions in the world given its abundance and diversity of plants and animals (Msilu, 2021). The climate in the area ranges from semi-arid to desert, with dry seasons from May to November and wet seasons from December to April (Kabuga, 2018). The average annual rainfall is 500 mm, and temperatures range between 15°C and 35°C (Kabuga, 2018). The park's principal commercial activity is photography tourism, which attracts approximately 21,000 people each year, many of whom come to see iconic wildlife species such as the African lion (Msilu, 2021). Furthermore, the area is plagued by wildlife-induced crop damage, wildlife-related mortality, deforestation, encroachments, and clashes between residents and protected area managers (Sirima, 2016).

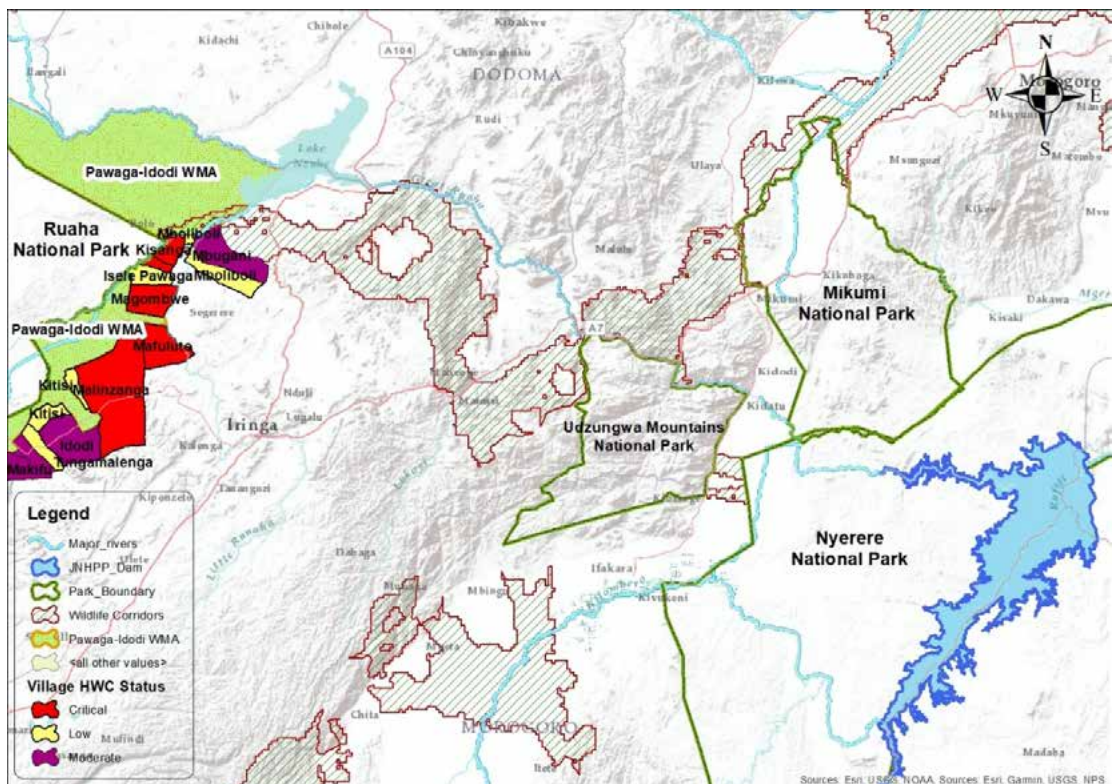


Figure 1: Map of the study area

Ruaha National Park was established in 1964 and is one of Africa's largest protected areas, comprising approximately 20,226 square kilometres (Barnes, 1983). The park is part of the larger Rungwa-Ruaha ecosystem, which encompasses the Rungwa and Usangu Game Reserves, resulting in a broad protected area that supports a wide range of species (Roug *et al.*, 2020). The park's ecosystems include miombo woodlands, riverine forests, grasslands, and wetlands, which support a diverse range of wildlife such as elephants, lions, buffaloes, hippos, and numerous bird species (Kimaro *et al.*, 2022).

A mixed-methods research approach was used, combining qualitative and quantitative data analysis to ensure a thorough grasp of the research problem. Data were acquired using four major approaches: (i) Surveys and questionnaires for heads of households and community leaders to assess conflict frequency and mitigation strategies. (ii) Focus Group Discussions (FGDs) with elders to explore

indigenous practices, traditional knowledge, and community attitudes towards conflict management. (iii) Key informant interviews (KIIs) with local communities, park officials, conservation NGO representatives, and technologists to gain insights.

Two sampling procedures were used: purposive sampling and stratified random sampling. Purposive sampling was used in the FGDs, KIIs, and observations to ensure the inclusion of people with relevant expertise and experience. Survey participation was targeted at household heads using stratified random sampling, allowing data to be collected from a wide range of socioeconomic and geographical origins. All research operations met accepted ethical norms. All participants provided informed consent, and cultural sensitivities and intellectual property rights associated with indigenous knowledge were fully honoured. Throughout the study, local institutions worked together to foster transparency, mutual respect, and equitable benefit sharing.

Data analysis

The project team applied SPSS for Windows version 20 and MS Excel to support data analysis for this research assignment to (i) work efficiently (ii) save time (iii) quickly organize, store and retrieve data (iv) Uncover connections in ways that are not possible manually (v) Rigorously back-up findings with evidence. SPSS and MS Excel are software that supports qualitative and mixed research data analysis. It is designed to help organize, analyse and find insights in qualitative data like interviews, open-ended survey responses and focus group discussions. Both qualitative and quantitative data were analysed with appropriate procedures. The quantitative data were analysed using descriptive statistics, while qualitative data from interviews and discussions were interpreted using thematic coding and content analysis.

RESULTS AND DISCUSSION

Demographic characteristics of the respondents

According to the 2022 census, the district's 12 villages have a total population of 44,002 people, with 53% women and 47% men spread over 8,122 households. This corresponds to an average household size of 5.4 people, which is approximately 17.4% larger than Tanzania's national average of 4.6 people per home (NBS, 2023). Most villages have a modest female majority, which reflects the district-wide trend. Notably, Kinyika Village has the highest proportion of women (59%), whereas Tungamalenga Village is the only one with more males than women (53%). These demographic patterns serve as a solid foundation for informed community planning and resource distribution.

Crops grown in the study villages

Rice is the most common crop in the area, grown by around 43.4% of farmers. It is typically planted with maize, sesame,

or sunflower. Maize is the second most important crop, cultivated as a single crop by approximately 11% of farmers and frequently coupled with sesame, peanuts, or rice in mixed farming systems. Sesame is also frequently cultivated, often grown alongside maize or rice. Sunflowers and peanuts are commonly seen in intercropped fields rather than as stand-alone crops. Other crops, such as cassava, sugarcane, bananas, and sorghum, are less common and typically grown alongside staple crops. Overall, local farming systems rely heavily on cereal production (rice and maize), supplemented by oilseeds (sesame and sunflower) and legumes (peanuts), which increase nutritional and livelihood diversification.

Patterns and drivers of Human–Wildlife Conflicts

Wildlife conflicts were identified as one of the most serious issues affecting the villages bordering Ruaha National Park through interviews, focus groups, and field observations. People in all studied villages identified crop raiding and animal predation as the most common conflicts, followed by human attacks. Respondents picked elephants (*Loxodonta africana*) as the most detrimental to crops, followed by lions (*Panthera leo*), hyenas (*Crocuta crocuta*), and leopards (*Panthera pardus*).

In focus group discussions in the March 2025 Tungamalenga village, some individuals described the behaviour of the issue elephants as follows: “Elephants often enter farms at night, especially during the dry season when food and water is scarce inside the park.” Interviews with farmers in Idodi and Msembe yielded similar experiences, with an emphasis on crop loss during critical periods. “When elephants come, they can destroy an entire field in one night; even shouting or lighting fires does not stop them,” explained one of the farmers, capturing not only crop destruction but also the unpredictability of the attacks and the resource-depleting behaviour of the raiding elephants, contrary to popular belief that problem animals

and smallholders are victimised populations. Geographically, disputes were found to be more common in villages closer to the park boundary.

Over the last decade, community members have reported major climate-related changes, with droughts (73%), rising temperatures (63%), and water scarcity (61%) emerging as the most common issues. Erratic rainfall (59%) and flooding (54%) were also commonly mentioned, indicating a rise in climate instability. Furthermore, more than a third (36%) of respondents reported an increase in climate-related diseases, most likely due to harsh weather and water scarcity.

During the March 2025 interviews, participants repeatedly stated that droughts exacerbate human-wildlife conflicts. They added that during the rainy season, wild animals avoid approaching settlements since water and food are abundant in the park. However, during the dry season, the shortage of these supplies attracts wildlife to adjacent settlements and farms. Elephants appear to know their usual paths, with community residents reporting that they frequently return to the same sites each dry season. Lions were also recorded to enter towns in search of food and water, occasionally killing animals like cows and goats. According to one response, nine cows died and three were damaged in March 2025 alone. KII-2 explained the issue as follows:

“During the dry season, wild animals migrate to adjacent settlements in search of food and water. This causes human-wildlife conflicts due to crop destruction and livestock losses. Elephants, gazelles, and tandalas (eland) are the most commonly involved species. Elephants eat maize and ruin paddy fields by trampling them”.

Overall, the interviews show how seasonal droughts increase contacts between humans and wildlife, posing threats to both livelihoods and safety in impacted communities. KII-3 provided a similar explanation:

“During the dry season, animals congregate around villages to search food and water. This leads to confrontations when crops are destroyed and cattle are attacked”.

This study’s findings support prior research undertaken in areas surrounding Ruaha National Park and other parts of Tanzania, which have also identified elephant crop raiding as a major source of human-wildlife conflict. For example, Kioko *et al.* (2006), Kideghesho *et al.* (2020), and Nyhus (2016) found elephants to be the most destructive species to agricultural fields, especially during the dry season when food and water sources within protected areas become low. Consistent with previous findings, our data suggest that elephants routinely attack farms near settlements, resulting in severe crop losses and compromising household food security. Farmers in the study area reported utilising similar deterrence techniques, such as chilli fences, noise, and night patrols, which have previously been demonstrated to be labour-intensive and only partially effective.

Furthermore, as with other studies, our findings indicate the growing contradiction between conservation goals and community livelihoods as a result of insufficient compensation and ineffective institutional response mechanisms (He *et al.*, 2023; Persson *et al.*, 2021). These similarities indicate that elephant crop raiding is a continuous, structural issue that necessitates collaborative, community-based mitigating strategies integrated into conservation management plans. Similar studies were undertaken in Limpopo National Park in Mozambique, and it was discovered the difficulties of integrating conservation efforts with local livelihoods, as communities face growing human-wildlife conflicts, notably agricultural loss due by elephants (Givá and Raitio, 2017).

Indigenous Knowledge and Local Strategies for Conflict Mitigation

Interviews and focus group discussions revealed that communities living adjacent to

Ruaha National Park continue to rely on a wide range of indigenous strategies to chase wildlife and protect their livelihoods. These include night guarding, burning dry grasses to produce smoke, beating metal objects, and using thorn fences to protect livestock enclosures. In several villages, participants described how they climbed on trees to monitor fields during harvest time, especially in areas prone to elephant incursions. A youth group in Idodi explained that “we take turns at night making noise with drums and tins when elephants come close,” while another participant from Tungamalenga added that “old men teach us how to make fire torches and wave them to scare animals away.” Such practices are deeply embedded in local routines and often combine practical defense with symbolic meaning. Fire and noise are not only deterrents but are believed to communicate territorial boundaries understood by animals. Observations during fieldwork confirmed the use of thorny branches of *Acacia tortilis* and *Commiphora africana* for fencing, indicating the persistence of local ecological knowledge about plant species effective for constructing durable barriers.

Cultural beliefs and collective decision-making also play a vital role in conflict mitigation. Elders interviewed in Msembe and Mapogoro emphasized that wildlife is part of the ancestral landscape and should be treated with respect rather than hostility. Some taboos prohibit the unnecessary killing of wild animals, particularly elephants and lions, which are viewed as spiritual messengers or symbols of strength. In one FGD, women participants explained that “when elders meet under the tree to discuss wildlife problems, they do not decide alone; they consult the spirits and ancestors before taking action.” Such processes guide decisions about when and how to guard, and whether to relocate livestock temporarily. Furthermore, communities use local ecological indicators such as the movement of birds, the flowering of certain trees, and the behaviour of insects to predict wildlife movements and seasonal changes.

For example, several respondents associated the appearance of specific bird species along the Ruaha River with the onset of elephant migration. These findings demonstrate that indigenous knowledge operates as an early warning system, developed through generations of observation and interaction with the environment. Integrating these insights with AI-based prediction models could strengthen real-time monitoring and improve community preparedness for human–wildlife conflicts.

Previous research in Tanzania and East Africa shows that indigenous knowledge and local methods play an important role in reducing human-wildlife conflicts, such as elephant crop raiding. According to Gillingham and Lee (2003), Kideghesho (2008), and Omondi *et al.* (2021), rural communities have long used locally developed methods to protect their crops and livestock, such as building thorn fences, making noise through drumming and shouting, lighting fires, and using chili-based deterrents. These tactics are based on generations of experience studying animal behaviour, migration patterns, and seasonal cycles. While modern conservation initiatives increasingly emphasise technological interventions (Ullah *et al.*, 2025; Marvin *et al.*, 2016), these studies show that indigenous approaches are still practical, adaptive, and culturally embedded, providing valuable insights for designing participatory and sustainable conflict-mitigation frameworks that combine traditional knowledge with modern wildlife management practices.

Application and Performance of AI Technologies

Interviews with park officers, conservation partners, and selected community representatives revealed that AI-powered technologies are now being in Ruaha National Park as part of efforts to monitor wildlife migration and decrease conflicts. The most frequently stated tools include video traps, drones, GPS-collared animals. Park rangers highlighted the use of AI-assisted drones

that help to chase elephants from the farms. According to a Ruaha national park officers interviewed in February 2025, “the drones and collars help us to know when elephants move outside the park so we can warn the nearby villages on time.” Community members in Tungamalenga and Msembe acknowledged getting early warnings via mobile messages relayed by project officers, which they characterised as “helpful, especially during the dry season when elephants often cross the farms.” These examples demonstrate AI technologies’ growing potential to provide real-time monitoring and improve early warning systems for reducing human-wildlife conflicts.

However, various limits and concerns were identified with regard to the performance and accessibility of these technologies. While AI-based systems have proven efficient for data gathering and species tracking, their application and interpretation are mostly limited to trained conservation personnel, restricting community ownership and knowledge. Participants in focus groups raised worries about the lack of feedback and transparency in how AI data is used, with one farmer in Idodi observing that “we hear about drones and collars, but we don’t know what information they take or how it helps us directly.” Technical difficulties, such as poor network connectivity, insufficient power supply, and high device maintenance costs, were also highlighted as impediments to long-term use. In some situations, wildlife damaged or stole camera traps, limiting data reliability. Despite these constraints, both park authorities and residents acknowledged that when coupled with indigenous tracking and observation systems, AI technologies hold significant promise for creating participatory, data-informed approaches to conflict management. Improving local capacity to evaluate and implement AI findings remains critical for achieving significant, equitable, and long-term impact.

This study’s findings support prior research that has shown that Artificial Intelligence (AI) technologies have a favourable impact

on reducing human-wildlife conflict. Previous research has shown that AI tools such as drones, camera traps, predictive algorithms, and acoustic sensors play an important role in detecting and tracking wildlife movements in real time, lowering the likelihood of animal incursions into human settlements and farmlands (Aliane, 2025; Rathi, 2025). Consistent with these findings, this study discovered that AI-powered early warning systems and predictive models improve the ability of communities and wildlife authorities to forecast conflict-prone locations and take proactive measures, such as placing deterrents or upgrading barriers. Furthermore, as stated by Mburu and Mutinda (2020), the usage of AI-supported communication networks was found to improve cooperation among rangers, farmers, and conservation officers, resulting in shorter response times and better conflict management. Overall, the use of AI technology into conservation efforts is consistent with current literature, which emphasises their effectiveness in improving monitoring, lowering crop and livestock losses, and fostering cohabitation between humans and wildlife.

Community Perceptions and Acceptance of AI Technologies

Interviews, surveys, and focus group discussions in the villages surrounding Ruaha National Park revealed that people’s attitudes towards and adoption of Artificial Intelligence (AI) technologies range significantly. Many of the participants were unfamiliar with how AI tools such as drones, camera traps, and tracking devices are utilised to monitor and manage wildlife. However, when the aim of these technologies was described, most people expressed cautious optimism because they saw how they could assist lessen conflicts between humans and wildlife (HWC), particularly when crops are destroyed or animals are lost. Some members of the community suggested that video traps and drones may be used to detect where animals are travelling early on, allowing people to respond promptly. Even while many people had nice things to say about AI technologies,

a considerable majority of them continued to distrust them. They perceived them as tools of outside control, deployed by conservation agencies rather than community-owned assets. Focus group talks revealed that this lack of trust stems from poor communication and a lack of involvement in decisions about how to utilise and manage these types of technologies.

The statistics also revealed that people of varying ages and genders held significantly different perspectives about AI. Younger participants, notably students and youths involved in conservation activities, were more responsive and curious about technology's role in environmental protection and livelihood enhancement. Older respondents, particularly elders and traditional leaders, placed a higher emphasis on indigenous knowledge systems such as tracking animal activity, seasonal trends, and community warning networks, believing them to be more dependable and culturally appropriate. Women who took part had differing perspectives. Some enjoyed how AI could warn people about wildlife risks and safeguard crops, but they didn't like not being invited to meetings and training sessions where technology was discussed or shown. All parties raised ethical and cultural concerns regarding data ownership, the prospect of replacing traditional ways, and the reality that locals lack the skills to use these technologies. The findings indicate that in order for AI technologies to be more generally accepted in Ruaha's border villages; they must be presented in a way that prioritises community involvement, openness, gender equality, and the incorporation of indigenous ecological knowledge into technology design and use.

RECOMMENDATION AND CONCLUSION

The study's findings show that human-wildlife conflict (HWC) worsens during droughts, when wild animals migrate from protected areas to surrounding communities in search of food and water. Elephants, lions, gazelles, and tandala were identified as the primary species

responsible for crop devastation and livestock predation, with elephants wreaking havoc on maize and paddy fields. The trend demonstrates the strong relationship between climatic stress, resource constraint, and conflict frequency. To address these issues, community-based conservation strategies should be developed through partnership among local inhabitants, park officials, and conservation organisations. Improved early warning systems, targeted water provisioning for wildlife within park boundaries, and the use of indigenous deterrent tactics, in conjunction with modern monitoring technologies such as AI-enabled video traps and drones, could all help to reduce conflict events. Furthermore, empowering people through awareness campaigns and compensation procedures would foster cohabitation, assuring both livelihood stability and animal protection.

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SUB-THEME: BEE ECOLOGY AND POLLINATORS

WILD EDIBLE INSECTS ENHANCE FOOD SECURITY AND PROMOTE CULTURAL HERITAGE IN THE LAKE ZONE, TANZANIA

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ABSTRACT

Insects are part of traditional diets to many ethnic groups in Tanzania. With increasing global attention in sustainable food systems, entomophagy “the practice of consuming insects” offers a valuable opportunity to address food security, promote cultural heritage, and support environmental sustainability. This study investigated the socio-economic, cultural and nutritional dimensions of edible wild insect consumption in Kagera and Mwanza regions in Tanzania. Using semi-structured questionnaire for household survey (n=233) and key informant interviews (n=10), the study assessed demographic influences, consumption patterns, motivations, harvesting and preparation methods, and perceptions of environmental changes affecting insect availability. Winged termites (87.1%) and longhorn grasshoppers (82.4%) were the most commonly consumed species, with cultural significance emerging as the leading motivation across all age and sex groups. Seasonal availability strongly influenced consumption patterns, with 83.7% of respondents consuming insects during rainy season. Habitat loss (82.4%), climate change (75.5%), and unsustainable harvesting (69.1%) were reported as key factors affecting insects’ availability. Hand-picking and frying were predominant harvesting and preparation methods respectively, and did not vary significantly between the regions (harvesting: $p = 0.224$; preparation: $p = 0.315$). These findings underscore the significance of entomophagy in the region and its potential contributions to food security, cultural heritage, and rural livelihoods.

Keywords: Entomophagy; Food security; Insects availability; Longhorn grasshoppers; Winged termites

Global climatic changes have impacted food production placing increased demand on agricultural systems and natural resources. As the global population continues to rise and demand for nutritious food increase, edible wild insects have the potential role to play in addressing food security in both rural and urban communities (Lange & Nakamura, 2021). Wild insects encompass diverse group of insects that lives in natural habitats such as forests, wetlands, grassland and desert, in contrast to those that are domesticated or reared in captivity (Samways, 2005). Insects such as beetles, bugs, bees, ants, wasps, flies, butterflies are well recognized due to the significant roles they play in ecosystems, including pollination, decomposition and nutrient cycling, soil

aeration, regulation, and a crucial food source for other wildlife (Sheikh *et al.*, 2017; Van Huis, 2021). Apart from being source of protein to human and other animals, some species are efficient recyclers of organic waste, contributing to waste reduction and improving soil fertility in agro-ecosystems (Sharma *et al.*, 2019). The larvae of black soldier flies, for example, have been used to process food waste and animal manure, converting them into usable form as high-quality manure (Siddiqui *et al.*, 2022). On the other hand, the larvae themselves are protein-rich and can be used as fresh or dried animal feed, while simultaneously reducing the environmental impact of waste disposal (Van Huis & Gasco, 2023).

Globally, insects are part of traditional diets to many indigenous groups and are often prepared as separate snacks or included in the traditional dishes (Gabrys *et al.*, 2008). Many societies have a long history of consuming insects as a source of food, a practice commonly known as entomophagy (Rumpold & Schlüter, 2015). Over 2 billion people worldwide are estimated to consume wild insects regularly, with more than 2,000 species recognized as edible (van Huis, 2020). In Africa, insects such as termites, crickets, grasshoppers, ants, and caterpillars are among the most widely consumed mainly in rural areas due to their high nutritional value (Kelemu *et al.*, 2015; Douglas, 2017; Lange & Nakamura, 2021). These insects are often harvested and collected from their natural habitats during specific seasons of the year and are either fried, dried, or used in stews (Matandirotya *et al.*, 2022). They offer a sustainable and nutritious food source, packed with protein, vitamins, and minerals (Liceaga *et al.*, 2022).

In Tanzania, the practice is increasingly recognized for its potential to contribute to sustainable nutrition and food security (Kimambo, 2016). The availability of edible wild insects among ethnic groups in the Lake Zone of Tanzania is influenced by a combination of ecological, socioeconomic, and cultural factors (Mmari *et al.*, 2017). Socio-economically, factors such as market demand, local agricultural practices, and economic conditions play crucial roles in affecting availability of edible insects (Anagonou *et al.*, 2023). Additionally, the growing demand for sustainable food sources and increasing awareness of the nutritional benefits of edible insects are driving their availability, while challenges such as habitat destruction, overharvesting, and the need for better preservation and storage methods may limit access for some communities. Economic factors, such as income levels and market access, further impact the affordability and availability

of edible insects among different ethnic groups (Ochieng *et al.*, 2023). Understanding the socio-economic contribution, consumption patterns and cultural practices among ethnic groups is crucial to promoting sustainable entomophagy to improve food security while conserving biodiversity across the country.

MATERIALS AND METHODS

Description of the study area

The research was conducted in Mwanza and Kagera regions, located in the lake zone of north western Tanzania (Fig 1). These regions have diverse ecological zones, including woodlands, grasslands, banana plantations, and wetlands, which are ideal habitats for various species of wild edible insects such as winged termites, longhorn grasshoppers and lake fly (non-biting midges) (Ayieko & Tamale, 2010). Mwanza features both urban and rural settings, with rural districts being dominated by the Sukuma, Kerewe, Jita and Nyamwezi people (Wijsen & Tanner, 2002). And Kagera is predominantly rural and inhabited by Haya, Nyambo, and Nyayangiro, who possess rich traditional knowledge on the harvesting and consumption of wild edible insects (Kimambo, 2016). Both regions experience a tropical climate with bimodal rainfall patterns, supporting seasonal availability of edible wild insects (Mmari *et al.*, 2017). The residents in both regions relies heavily on subsistence agriculture, fishing, and small-scale trade, where insects play a role both as a food source and as an income-generating activity. The study areas were purposively selected due to their cultural diversity, ecological richness, and the strong presence of indigenous practices related to insect utilization, making them ideal for assessing the consumption status and socio-economic significance of edible wild insects.

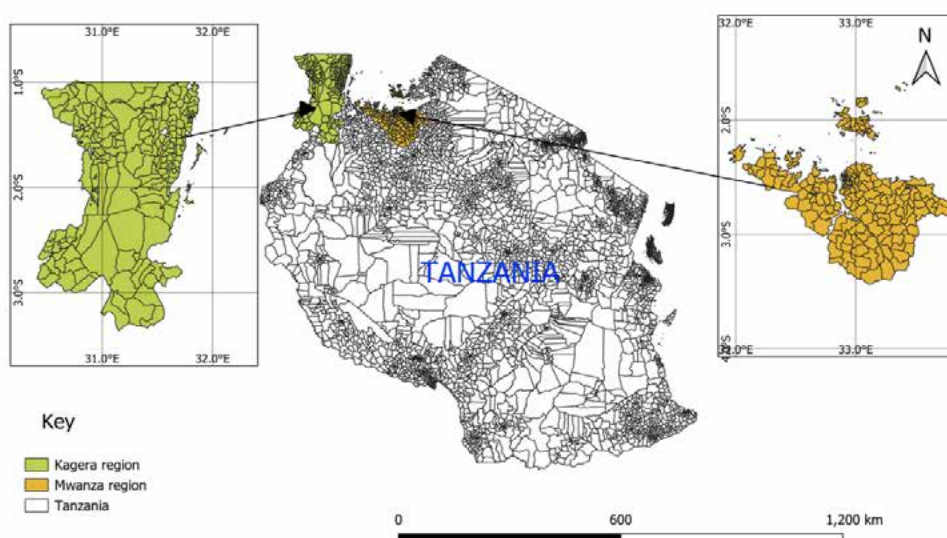


Figure 13: A map of the study area. Created by author, 2025

DATA AND DATA SOURCE

Training of interviewers

A team of three research assistants conducted the interviews in the study area. Prior to commencement of actual data collection, all interviewers were trained by researchers for three days at The University of Dodoma on ethics and data collection techniques. The aim was to ensure collection of quality data while maintaining a good relationship with the respondents and selected key informants. After training, the assistants undertook two days pilot sessions in Ng'onghona village, Dodoma so as to test the questionnaires and familiarize with the questions.

Sampling and sample size

The sample size ($n=233$) for household questionnaire surveys was determined using a simple random sampling technique. Due to the reason that the current population of insect consumers in the study regions were unknown by the period of this study, the fisher Andrew's formula was used to obtain the sample size (Jung, 2014)

$$n = \frac{Z^2 \cdot P (1 - P)}{E^2}$$

Where by z -value for 95% confidence level, the Z - value is 1.96, P is proportional of the population exhibiting the consumption of wild insects, where by there is no estimate available so 0.5 will be used. E is marginal error typically is 5% based on the confidence level of 95%. After calculating the sample size, the study shows 384 participants but alternative to adjust sample size, 233 households as sample size was selected to ensure reliability and the main criterion being insects' consumers. Simple random sampling used to ensure that different indigenous groups, income levels, were well represented. Additionally, 10 key informants were purposefully selected based on their experience and expertise on entomophagy.

Data collection

Data from the selected respondents were gathered via the semi-structured household questionnaire surveys (HQS) and Key Informant Interviews (KIIs). The HQS was chosen as the major data collection tool since it enables gathering of more information on personal opinions and views about a particular subject (Creswell, 2014). Also, in-depth interviews with selected key informants such as local elders, nutritionists, natural resource

officers, agricultural extension officers, and market vendors were carried out to enable a thorough investigation of the subject matter. Both HQS and KIIs focused on collecting information on the consumption status, cultural practices, current threats, availability, health and socio-economic benefits of the edible insects across each ethnic group. The checklist of questions for the tools were pre-developed and tested, but some flexibility was allowed during actual data collection to ensure that more information was gathered from the participants. Only respondents who were at least 18 years old provided the information, assuming that at this age range they are actively participating in socio-economic activities and possess knowledge of entomophagy.

Data analysis

Excel, SPSS, and STATA software were used to perform both descriptive and inferential statistics for the collected data. Frequency, percentage, and means were used to find the consumption status and demographic data among the respondents. Inferential statistics including Chi-square tests were used to assess the relationship between categorical variables including ethnic groups, income levels, consumption patterns, cultural perceptions for insects. Binary logistic regression was used to examine the influence of edible wild insects on socio-economic benefits within and between the ethnic groups.

RESULTS

Socio-demographic characteristics of respondents

Out of 233 respondents interviewed, 53.9% were males and 46.1% were females, and they differed significantly between the two regions ($p = 0.038$). Kagera region contributed a total of 48.1% ($n=112$) while Mwanza contributed a total of 51.9% ($n=121$) of all respondents. The age distribution was relatively balanced across the regions, with the largest group being 18–27 years (25.9%), followed by 38–47 years (22.2%) and 48–57 years (21.4%), indicating similar age engagement in both areas. Majority of respondents (77.8%) had formal education while 22.2% attained informal education. Individuals with informal education refers to those who have not attended planned or structured classroom settings but have acquired knowledge through real-life experiences. Most participants were self-employed (75.7%), with significant variation by region ($p < 0.001$), highlighting differing economic dependence on natural resources. Ethnically, Haya (47.0%) and Nyambo (19.2%) dominated, with notable regional differences ($p < 0.001$), and since cultural background plays a crucial role in entomophagy practices, this diversity may influence both consumption habits and knowledge systems related to wild edible insects across the two regions.

Table1: Sociodemographic characteristics of all respondents interviewed ($n = 233$) in the two regions

Variable	Categories	Kagera ($n=112$)	Mwanza ($n=121$)	Total (%) ($n=233$)	Test statistic (t)	P-value
Gender	Male	60.7	47.1	53.9	2.092	0.038
	Female	39.3	52.9	46.1		
Age group	18 to 27	30.4	21.5	25.9	-0.304	0.762
	28 to 37	13.4	18.2	15.8		
	38 to 47	18.8	25.6	22.2		
	48 to 57	20.5	22.3	21.4		
	58+	17.0	12.4	14.7		

Education level	Formal education	68.8	86.8	77.8	3.392	0.001
	Informal education	31.3	13.2	22.2		
Occupation	Self employed	90.2	61.2	75.7	-5.938	<0.001
	Employed	0.9	26.4	13.7		
	Unemployed	0.0	5.0	2.5		
	Students	8.9	7.4	8.2		
Ethnic group	Nyayangiro	10.7	0.0	5.4	-4.253	<0.001
	Haya	63.4	19.8	41.6		
	Nyambo	25.9	30.6	28.2		
	Jita	0	12.4	6.2		
	Sukuma	0	22.3	11.2		
	Kerewe	0	9.1	4.5		
	Nyamwezi	0	5.8	2.9		

Common wild edible insects consumed by ethnic groups

The consumption of wild edible insects is being practiced by major ethnic groups in the study area. Winged termites (87.1%) were the most widely consumed insects while locusts were less commonly eaten, reported by only 9.9% of respondents. The Haya people followed by Nyambo people in Kagera region were most ethnic groups with high consumption acceptances of Longhorn grasshopper, winged termites and locust than other ethnic groups in Mwanza Region (Fig 2).

Analysis by sex revealed that both males and females consumed wild edible insects for various reasons, but cultural significance was the most cited motivation across both groups (75% of females and 67.6% of males consuming for cultural reasons). The reasons for consumption did not differ significantly between sex ($\chi^2 = 8.70$, $df = 3$, $p = 0.833$). Similarly, when evaluating community perceptions toward insect consumption, males and females showed similar views, with the majority indicating cultural acceptance (58.4%) with the Chi-square test yielded a value of ($\chi^2 = 8.598$, $df = 3$, $p = 0.035$) which were statistically significant due to other perception views.

Cross-tabulation by age demonstrated that younger (18–27) and middle-aged (28–47) respondents consumed insects largely for cultural and nutritional reasons, while older respondents (48+) cited cultural significance most often. The association between age and reason for consumption was not statistically significant ($\chi^2 = 15.102$, $df = 12$, $p = 0.236$). However, age was more influential in shaping community perceptions, with cultural acceptance being dominant across all age groups although it was not statistically significant ($\chi^2 = 16.175$, $df = 12$, $p = 0.183$).

In term of ethnic groups, the Haya and Nyambo were the most represented and showed strong cultural acceptance of insect consumption. Among the 95 Haya respondents, 52 respondents consumed insects for cultural reasons. It was found that the differences in reasons for consumption and community perceptions were not statistically significant between the groups (reason: $\chi^2 = 4.260$, $df = 6$, $p = 0.642$; perception: $\chi^2 = 23.506$, $df = 15$, $p = 0.074$). The study also found that the majority of respondents (83.7%) reported consuming wild edible insects seasonally, suggesting that consumption is closely tied to seasonal availability rather than being a regular dietary habit. Only 16.3% of the respondents explained that they can access the insects for consumption

occasionally. Notably, consumption patterns varied significantly by region ($\chi^2 = 9.391$, $df = 2$, $p = 0.009$): in Kagera, nearly all respondents consumed seasonally (91%), with very few reporting occasional or monthly use (4.5% each), while Mwanza showed more

variation with 76.9% of respondents consumed seasonally, 15.7% reported occasionally while 7.4% monthly consumption. This regional difference reflects ecological factors, cultural norms, or differences in insect availability across the two areas.

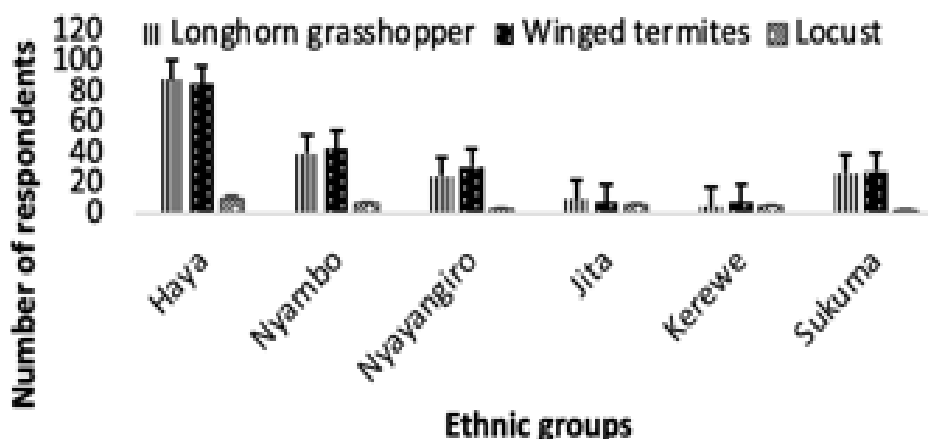


Figure 14: Consumption pattern of edible wild insects across the ethnic groups.

Factors influencing the availability, distribution and diversity of edible wild insects

Key factors reported to influence the availability, distribution, and diversity of edible insects were habitat loss (82.4%), climate change (75.5%), and overharvesting (69.1%) (Fig 3). When comparing study areas, the perception of change in insect availability over the past ten

years significantly differed between the regions ($\chi^2 = 6.169$, $df = 2$, $p = 0.045$), with more respondents in Mwanza reporting high change (53.7%) compared to Kagera (41%). However, there were no significant differences between Kagera and Mwanza on how they perceived the impact of habitat loss ($\chi^2 = 0.346$, $p = 0.656$), climate change ($\chi^2 = 0.034$, $p = 0.855$), or overharvesting ($\chi^2 = 0.012$, $p = 0.912$).

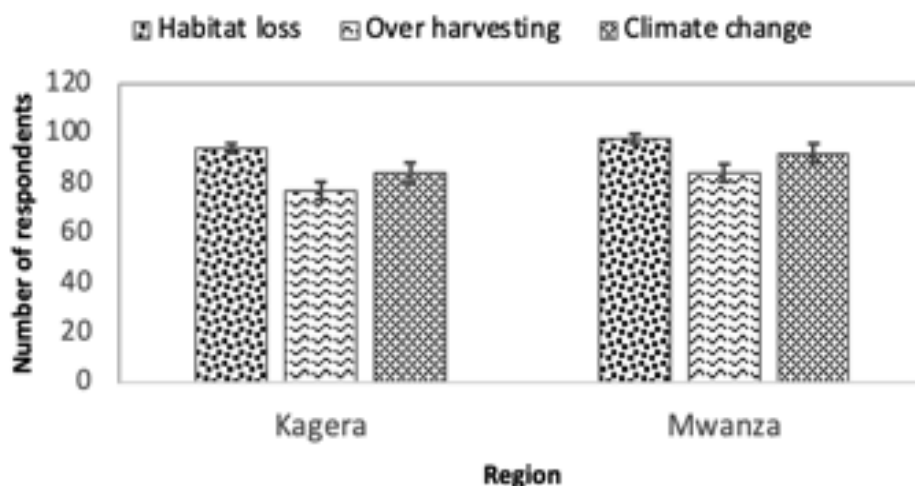


Figure 15: Factors affecting the availability, distribution and diversity of wild edible insects

Perceived nutritional benefits of wild edible insects

During interviews, the respondents identified several nutritional benefits of consuming wild edible insects, with fat content being the most frequently recognized (63.9%, $n = 149$), followed by minerals and vitamins (44.2%, $n = 103$), protein (42.5%, $n = 99$), and dietary fibre (24.5%, $n = 57$) being the least. To further explore these perceptions, a binary

logistic regression was conducted to assess the influence of education, location, sex, and age on awareness of fat content as shown in Table. 02. The findings revealed that both education ($p = 0.026$, $\text{Exp}(B) = 2.268$) and location ($p = 0.023$, $\text{Exp}(B) = 2.005$) were important predictors. Respondents with higher education and particularly those from Mwanza region perceived fat as the key nutritional benefit. In contrast, perceptions did not vary significantly between the sex ($p = 0.998$) and age ($p = 0.112$).

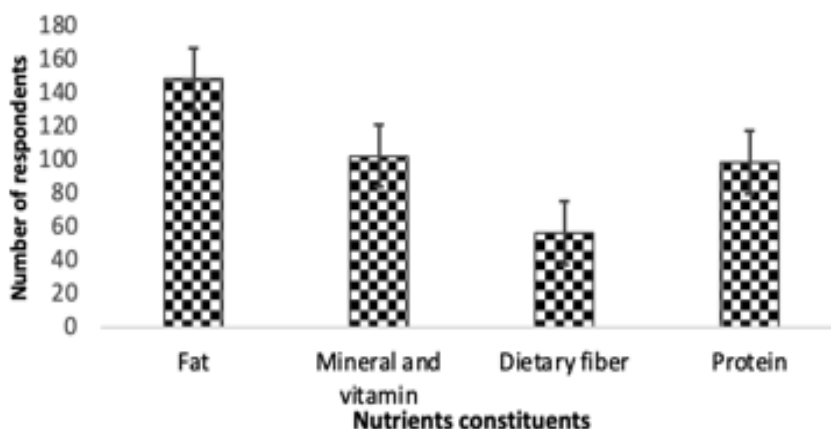


Figure 16: Perceived nutritional benefits of wild edible insects by respondents in Mwanza and Kagera regions

Harvesting and preparation methods practiced by ethnic groups in lake zone,

Hand-picking was the most dominant harvesting method (70.0%, $n = 163$) followed by light traps (23.6%, $n = 55$) and sweep nets (6.4%, $n = 15$) being the least (Fig 5). Similarly, the most common method practiced

by ethnic groups when processing the collected wild edible insects was frying (70.0%, $n = 163$), followed by smoking (14.2%, $n = 33$), boiling (8.6%, $n = 20$), and roasting (7.3%, $n = 17$) being the least (Fig 6). Neither preparation ($\chi^2 = 1.84$, $p = 0.912$) nor preparation methods ($\chi^2 = 2.988$, $p = 0.224$) that showed significant differences between the regions.

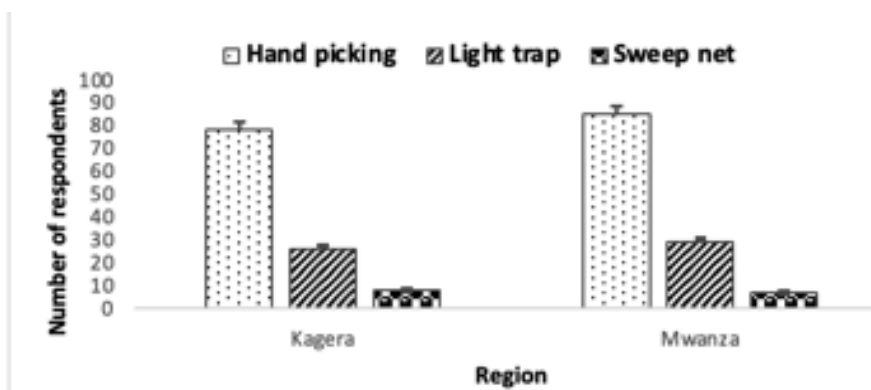


Figure 17: Harvesting methods practiced by ethnic groups in lake zone

Management strategies to ensure long-term availability of wild edible insects

To have detailed information on management strategies that work better in the study area, we conducted interview with key informants including local elders, nutritionists, natural

resource officers, agricultural extension officers, and market vendors. Generally, a range of management strategies were reported, such as abiding to food safety regulations, research and monitoring, market improvement, education and public awareness, sustainable harvesting practices, and habitat restoration.

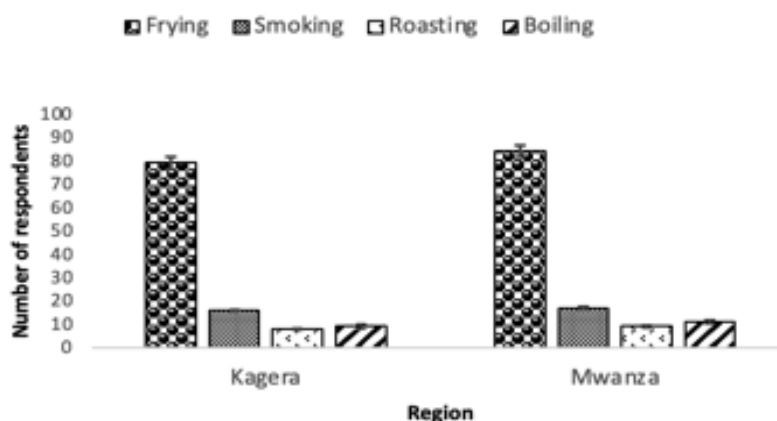


Figure 18: Preparation methods practiced by ethnic groups in lake zone

DISCUSSION

The study provides an in-depth understanding of the socio-cultural, ecological, economical and nutritional dimensions of wild edible insect consumption in Kagera and Mwanza regions, north-western Tanzania. It offers valuable insights into how demographic factors, cultural beliefs, ecological awareness, and nutritional perceptions influence the utilization of edible insects, contributing to sustainable food systems and biodiversity conservation (Lange & Nakamura, 2023). Furthermore, the demographic profile of respondents underscores the significance of gender and regional patterns among others, which influence the consumption of wild edible insects. A statistically significant difference by gender ($p = 0.038$) across the two regions suggests a strong influence in insect-related activities. This aligns with Okia *et al.*, (2017), who reported gender positions in insect harvesting and preparation across East Africa. The males often engage in harvesting while females normally dominate the preparation and marketing, highlighting the need for gender

sensitive policy and strategies when initiating for entomophagy.

The age distribution was relatively balanced as we found no significant difference between age groups across the regions, suggesting similar levels of age-related engagement in the area. Interestingly, youths (age between 18 and 27 years) were the most represented group in this study, which may indicate the shifting of livelihood strategies and income diversification in response to environmental and economic constraints. Similar findings were reported in Uganda and western Kenya where youths were found to involve in alternative livelihood strategies like insect rearing and processing (Kelemu *et al.*, 2015). Education had a substantial impact, with majority of interviewees having formal education, particularly in Mwanza region. This is crucial since formal education was associated with greater awareness of insects' nutritional benefits, as confirmed by logistic regression findings. One of the previous studies reported that education enhances knowledge

and willingness to consume alternative protein sources (Rumpold & van Hui, 2021). These findings suggest that educational interventions could significantly boost informed and sustainable consumption of wild edible insects, particularly through curriculum integration and community outreach. Ethnicity emerged as a key cultural determinant, with the Haya and Nyambo dominating the sample (Fig 2). These groups, particularly in Kagera, reported higher rates of insect consumption, notably winged termites and longhorn grasshoppers. This cultural influence was also highlighted by Mmari *et al.*, (2017), who showed that entomophagy was deeply rooted in cultural heritage, especially among Bantu-speaking groups in East African countries. Despite this, Chi-square tests revealed no statistically significant association between ethnicity and reasons for consumption as well as perceptions. This indicates a broad cultural acceptance of entomophagy across the groups, suggesting that while some groups may have slight variation in their practices, but consumption was not exclusive to specific ethnic group and may spread through shared practices and education.

The importance of cultural preservation was strongly expressed by one of the community elders in Kagera during in-depth interviews that; *“Senene (Swahili name for longhorn grasshopper) is not just a food, it’s a cultural treasure, it brings people together; it is a social event. In the past, senene was reserved for elders and men during special occasions, but now everyone enjoys it. I feel proud that something from our land is helping the youth to survive economically and nutritionally”*. This aligns with previous studies showing that “senene is becoming increasingly popular, providing opportunity for local businesses” (Mmari *et al.* 2017, p. 60). The elders’ view that senene supports both culture and livelihoods is also supported by research reporting that “the trade in edible insects carries economic potential important for improving incomes and the livelihoods of women and smallholder farmers” (Mmari *et al.* 2017, p. 60). This supports the idea that senene is now valued

not only culturally but also economically and nutritionally among communities in Kagera.

Consumption of wild edible insects was widespread, with winged termites and longhorn grasshoppers being the most consumed species (Fig 2). This conforms with the findings by (Ayieko & Tamale, 2010) in northwestern Tanzania, who identified termites and grasshoppers as the most commonly harvested edible insects due to their abundance and palatability. Cultural significance was the leading motivation for consumption, surpassing nutritional or economic drivers. Although no significant association was found between sex and consumption reasons, this cultural dimension was critical. Cultural beliefs dictate dietary norms, particularly in rural East African societies, where insects are often tied to rituals, taboos, and seasonal ceremonies (FAO, 2013). As such, these findings highlight the importance of respecting cultural perspectives when designing entomophagy promotion programs.

Cross-tabulations by age group indicated that younger and middle-aged respondents consume insects for both nutritional and cultural reasons, while elders predominantly consume for cultural reasons. This pattern, though not statistically significant, suggests intergenerational diffusion of entomophagy practices, with younger generations integrating modern knowledge (e.g., nutrition) into traditional habits (Moruzzo *et al.*, 2021). This fusion of knowledge presents opportunities for enhancing community health through targeted education that builds on existing beliefs among ethnic groups (Gahukar, 2020). Furthermore, majority of respondents consumed insects seasonally, with significant regional differences. Mwanza residents reported more frequent (monthly or occasional) consumption, possibly due to differing agroecological zones or availability of forest patches supporting diverse insect populations. Seasonal consumption was also reported by (Ishara *et al.*, 2023), who linked insect abundance with rainy seasons and traditional harvesting calendars in similar area.

These results support the notion that ecological cycles strongly influence entomophagy (Anagonou *et al.*, 2023b). Respondents consistently cited habitat loss, climate change, and overharvesting as threats to edible insect availability. These perceptions were statistically consistent across regions, except for perceived overall change over the past decade, with Mwanza residents more likely to report major declines (Byamukama *et al.*, 2025). This reflects localized environmental degradation patterns, including deforestation and land-use change, which were prominent in urbanizing areas of Mwanza.

An agricultural officer from Nyakato Ward, Kagera, warned, *“Agricultural activity destructs habitat of wild edible insects, especially winged termites, and pine tree plantations reduce longhorn grasshopper availability during across seasons. To promote their availability, sustainable technology should be promoted for harvesting, processing, and preparation of wild edible insects.”* This ecological awareness suggests a strong potential for community-based conservation strategies that incorporate traditional ecological knowledge, as emphasized by Kelemu *et al.* (2015). Integration of insect conservation into local environmental policies and community resource management plans can enhance both biodiversity protection and sustainable food sources.

Moreover, fat was the most cited nutritional benefit of consuming wild edible insects, followed by minerals/vitamins and protein. This aligns with compositional studies such as those (Odongo *et al.*, 2018), which found high lipid and protein content in East African termite and grasshopper species. Awareness of these benefits can be a powerful tool in promoting entomophagy as a viable solution to malnutrition. Education and regions as significant predictors of fat content awareness. Educated respondents and those from Mwanza were more likely to perceive fat as a benefit, reinforcing the role of formal knowledge and environmental exposure in shaping nutritional

literacy. Conversely, neither sex nor age significantly predicted fat awareness, reflecting earlier findings that nutritional knowledge is not strongly gender- or age-dependent in rural Tanzanian settings.

A ward health officer reinforced the nutritional importance of edible insects: *“Senene and other edible insects are a good source of protein and micronutrients, especially in areas where people can’t afford meat or fish regularly. However, we emphasize hygiene during processing and selling to avoid health risks”*. Traditional methods such as frying and hand-picking were the most common preparation and harvesting methods. These techniques were consistent across sex and region, as shown by non-significant Chi-squared values for both variables (harvesting: $p = 0.224$; preparation: $p = 0.315$). This reflects deeply embedded indigenous knowledge systems and the adaptation of local communities to available tools and resources, also reported by (Gahukar, 2020). Low-tech practices such as light traps and sweep nets were used less frequently, possibly due to limited access to such equipment or a preference for labor-intensive but familiar methods. This supports the findings of (Banjo *et al.*, 2006) who noted that most rural communities in East Africa use low-cost, sustainable insect collection techniques. Promoting improved tools and training could enhance efficiency and hygiene, particularly for income-generating activities. A senene vendor summarized the socio-economic impact as follows: *“Selling senene has been my main source of income for the past eight years. During the harvesting season, I can earn between TZS 150,000 (USD 60) to 300,000 ((USD 120) in a week. We use the income to pay school fees, buy household items, and even support other small businesses”*.

Implications to conservation

The findings of this study highlight the critical need to protect the natural habitats that support edible wild insect populations in the Lake Zone. With habitat loss, climate change, and

overharvesting reported as major threats, the long-term availability of insects such as winged termites and longhorn grasshoppers is increasingly at risk. Protecting forests, grasslands, and termite mounds is essential not only for sustaining insect diversity but also for preserving the cultural traditions and food security of local communities. Strengthening community-based resource management, restoring degraded landscapes, and promoting environmentally friendly harvesting practices can help maintain stable insect populations. In addition, integrating insect conservation into local agricultural and environmental policies will ensure that both biodiversity and livelihoods are safeguarded. Ensuring sustainable entomophagy practices is therefore key to supporting ecological resilience, cultural heritage, and the nutritional well-being of future generations.

CONCLUSION AND RECOMMENDATION

This study shows that edible wild insects are culturally, nutritionally, and economically important among communities in Kagera and Mwanza. Winged termites and longhorn grasshoppers were the most consumed species, with seasonal availability strongly shaping consumption habits. Communities recognized habitat loss, climate change, and overharvesting as major threats to insect abundance. Edible insects were also valued for their high fat, protein, and micronutrient content, while harvesting and trading provided meaningful income, especially for women and youth. These findings highlight the need to support entomophagy as part of sustainable nutrition, cultural preservation, and biodiversity conservation.

To ensure long-term availability and safe use of edible insects, the study recommends promoting sustainable harvesting practices, restoring habitats, and integrating insect conservation into local environmental planning. Public awareness should be strengthened to improve knowledge on nutrition, hygiene, and safe

processing. Support for local markets through better processing, packaging, and food safety guidelines is needed to enhance economic benefits. Further research and monitoring should document population trends and indigenous knowledge. Policymakers should also include edible insects in food security strategies and encourage gender-inclusive programs that benefit women and youth involved in insect-related activities.

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PRELIMINARY SURVEY OF BUTTERFLIES IN MOYOWOSI GAME RESERVE, WESTERN TANZANIA.

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ABSTRACT

This study aimed at determining butterfly species composition, diversity and distribution of the Moyowosi Game Reserve (MGR). The study employed a transect survey with hand-held butterfly sweep nets, visual observations and butterfly baited traps methods for the data collection. The results indicated that about 912 butterfly individuals were recorded belonging to 5 families, 58 genera and 98 species. The higher butterflies' species richness (54.28%), mean abundance (7 ± 1.24), and diversity ($H' = 3.51$) were found more frequently in the closed miombo woodland habitat type. Moreover, the Nymphalidae family was the common butterfly group dominated the area. Again, the results indicated that more than half (55.1%) of butterfly species were habitat-specific such that (27.6%) showed more specificity to closed miombo woodland habitat than other habitats in the area. Therefore, implying that habitat monitoring was very important for the butterfly conservation in the MGR. Furthermore, future conservation and management plans for MGR should focus conserving these refuge habitats for continued existence of butterfly species for achieving biodiversity conservation at a large scale.

Key words: Butterfly, Composition, Miombo, Woodland,

INTRODUCTION

Conservation of biodiversity should involve the protection and scientific management of all levels of biological diversity in order to maintain it at a certain threshold level and to derive sustainable benefits for present and future generations (Nicholls and Altieri, 2013; Goldstein, 2017; Archer, 2021; Ren, 2021). However, most studies in Tanzania have focused on large charismatic species of herbivores and carnivores, leaving most of the invertebrate species including insects under represented despite their huge abundance and diversity (Fitzherbert *et al.*, 2006, Nkwabi *et al.*, 2021). Still, today there is a global alert on the small animal taxa including insects. Overall

insects represent 56% of the 1.7 million species that have been described in the world (Siemann and Weisser, 2008; Stork, 2018). Recognizing the insects as an important component in the ecosystem, this study assessed the species abundance, diversity, and composition of insects in the group of butterflies in different habitats of Moyowosi Game Reserve (MGR).

Butterflies are the second largest, the most widespread, and recognizable group of arthropods in the order Lepidoptera (Devries *et al.*, 1997; Devries *et al.*, 1999). They can be visually identified to species level even when they are flying or landing on a surface because they have relatively large size, are diurnal, have a colourful appearance, have ease

of sampling, and have broad appeal (Devries *et al.*, 1997; Devries *et al.*, 1999; Freitas *et al.*, 2021; Himmelman, 2023). In addition, butterfly populations are strongly influenced by local weather conditions, microclimate, and light levels, marking them as good indicators for environmental change (Dennis *et al.*, 2013, Nkwabi *et al.*, 2017, Schmitt *et al.*, 2021, Sharma and Sharma 2021, Rija 2022). Currently, these small animal taxa have received increased attention as key indicators in ecological and conservation monitoring, as they are more sensitive indicators of habitat pollution and changes than large animals (Fitzherbert *et al.*, 2006, Notø 2014, Nkwabi *et al.*, 2021).

Butterfly composition varies significantly across different habitats, influenced by ecological integrity, human activity, and environmental conditions. There is a positive correlation between habitat diversity and butterfly abundance, for instance Munisi *et al.*, (2024); Nkwabi *et al.*, (2017) and Fitzherbert *et al.*, (2006) showed that miombo woodland and riverine forests exhibits higher species richness and abundance, while grasslands and cultivated areas recorded the lowest diversity of butterflies (Fitzherbert *et al.*, 2006). This indicates that butterfly community composition is negatively affected by various anthropogenic disturbances such as agriculture and habitat destruction, leading to decreased species richness and diversity (Fitzherbert *et al.*, 2006; Eskildsen *et al.*, 2015; Archer *et al.*, 2021; Kasisi *et al.*, 2024). Our study assessed the community composition of butterflies in four habitat types in Moyowosi Game Reserve namely; open miombo woodland, closed miombo woodland, grassland, and forest. Overall, the study emphasizes that protecting natural habitats is crucial for maintaining butterfly diversity in protected and open areas. The conservation of butterflies is crucial due to their ecological significance, aesthetic value, and the threats they face from human activities. Butterflies serve as vital pollinators, supporting plant reproduction and ecosystem stability (Jarpla, 2024; Kumar *et al.*, 2024) and indicators of environmental

health, making their preservation essential for maintaining biodiversity and ecosystem functionality. As sensitive species, butterflies reflect the health of their habitats, indicating broader environmental changes (Jarpla, 2024; Kumar *et al.*, 2024), hence conserving diverse habitats for butterfly populations are vital.

MATERIALS AND METHODS

Study Area

The study was conducted in the MGR which is in the western part of Tanzania, located between 30°30'00" to 31°30'00" East and 3°15'00" to 5°0'00" South with a size of 11,408 km². The climate and rainfall of Moyowosi Game Reserve in western Tanzania has an average of 29°C is characterized by two rainfall peaks in February and November, and a dry season from May to October. The game reserve obtains rainfall between 1000 mm and 1,500mm per annum. Precipitation peaks in November to December and April to May. The vegetation of the area consisted of four main habitat types namely: Open and closed miombo woodland, grassland and forest. In each habitat type was consisted of more than 90% cover of one of the selected species. The open and closed Miombo woodland harboured a variety of trees species including *Brachystegia spiciformis*, *Julbernadia globiflora*, *Pterocarpus angolensis*, *Combretum psidioides*, *Cassipourea mollis*, *Gardenia ternifolia*, *Catunaregam taylorii* and *Phyllanthus inflata*. Also, the ground floor was devoid of grass cover including *Themeda triandra*, *Hypparrhenia* species and *Andropogon* species which in turn when dominated to 90% created grassland habitats (Musika *et al.* 2021). The forest habitat was mainly observed along rivers and was dominated by *Borassus* and *Phoenix* palms with stands of *Vachellia* woodland along the margins of the floodplain (Musika *et al.* 2021).

The MGR is wealthy in biodiversity and harbours species preferring wetland-habitats like the Nile crocodile (*Crocodylus niloticus*) and African slender-snouted crocodile (*Crocodylus*

cataphractus), and bird species include the shoebill (*Balaeniceps rex*) and the wattled crane (*Buggeranus caruncuta*). The MGR also harbor the mammalian species such as African elephant (*Loxodonta africana*), lion (*Panthera leo*), giraffe (*Giraffa camelopardalis*), African buffalo (*Syncerus caffer*), lion (*Panthera leo*), wild dog (*Lycaon pictus*) and greater kudu (*Tragelaphus strepsiceros*).

Data collection design

The survey was conducted from 10th - 25th April 2018 and from 3rd - 16th April, 2024 using hand-held butterfly sweep nets, visual observations and butterfly baited traps. A transect of 5km long was established and crossed all the habitat types in the area (i.e. open and closed Miombo woodland, grassland, and forest) to capture as many species as possible. In each transect, twenty-five sampling points were spaced at every 200m making a total of 150 sampling points within six established transects in the area. Each point was sampled using two complementary methods. First, transect method through direct visual observation was used to sample butterflies by observing of common flying individuals along transect. In transect method involved counting the number of common and easy to identify flying butterflies that crossed a strip of known length (between 30 m and 60 m) and 20 m wide for 10 minutes, and time lapsed was recorded by a handheld Smartphone. Secondly, butterflies were collected by sweeping with a hand-held butterfly net (35 cm in diameter) following Sinclair *et al.* (2015). Butterfly searches were conducted by four collectors from morning (09:00 hrs. to 11:00 hrs.) when butterflies were active and evening from 15:00 hrs to 17.00 hrs. Species types and associated habitat types were recorded on standardized datasheets. Individuals not identified in the field were kept in special envelopes and later identified to the species level with the aid of field guidebooks (Kielland 1990, Larsen 2005, Martins and Collins 2016), collected butterflies were kept in the laboratory for further references.

Data analysis

The Kruskal-Wallis (KW) test was used to examine differences between multiple independent samples among habitat types. Estimates of species richness were obtained following Seaby and Henderson (2007). This index uses three biodiversity indices including, richness, diversity, and abundance to explain which habitat types had the greatest biodiversity for the butterfly community.

Diversity of butterfly species was determined by the Shannon-Wiener Diversity (H') index denoted as;

$$H' = - \sum_{i=1}^k p_i \ln p_i$$

Where k is the total number of species and p_i is the proportion of individuals found in the i^{th} species

Jaccard Similarity Index (J_i) was calculated in species composition using the presence-absence method (Magurran 2004). The J_i compares the proportions of individual species occurring in two different habitats or communities: Using the formula; $A/(A+B+C)$, where A represents the number of species occurring in both communities, B represents the number of individuals found only in community B, and C refers to the number of individuals occurring in only community C (Magurran 2004). The proportion was multiplied by 100 to easily interpret the Jaccard similarity index (J_i), which varies between 0 and 1 such that values close to 1 refer to highly similar species composition. In our case, J_i varies from 0 to 100.

RESULTS

Species composition

We recorded 912 butterfly individuals belonging to 5 families, 58 genera, and 98 species in four habitat types: forest, closed woodland, open woodland, and grassland (Appendix 1). Across habitat types, 54.28% of butterflies were in closed woodland, 17.32% in open woodland,

17.21% in grassland, and 11.18% in the forest. Nymphalidae had the highest number of species (47), followed by Hesperidae (21), Pieridae (13), and Lycaenidae (11), while Papilionidae had only 6 species. Species richness differed significantly across habitat types Kruskal-Wallis rank-sum test, ($\chi^2 = 12.01$, $df = 5$, $P < 0.004$).

Abundance of butterfly

The mean abundance of individuals did not differ across families ($\chi^2 = 3.56$, $df = 19$, $P < 0.314$) but differed significantly across habitat types ($\chi^2 = 8.65$, $df = 180$, $P < 0.034$). The

abundance (mean $\pm$ SE) was higher in closed woodland 7 ± 1.24 , followed by forest 4.43 ± 0.68 , open woodland 3.6 ± 0.53 , and least in grassland 3.41 ± 0.81 (Table 1; Figure 1). The abundance (mean $\pm$ SE) was relatively higher; Lycaenidae was abundant in closed woodland (11.25 ± 4.97) compared to other families, followed by Pieridae in closed woodland (9.0 ± 4.77), and lastly Nymphalidae (8.60 ± 1.97) (Table 1; Figure 1). Generally, Pieridae was observed in all habitats, and the least mean abundance was in open woodland and grassland each with two individuals of Papilionidae, and two 2 individuals of Hesperidae in forest (Fig.1; Table 1).

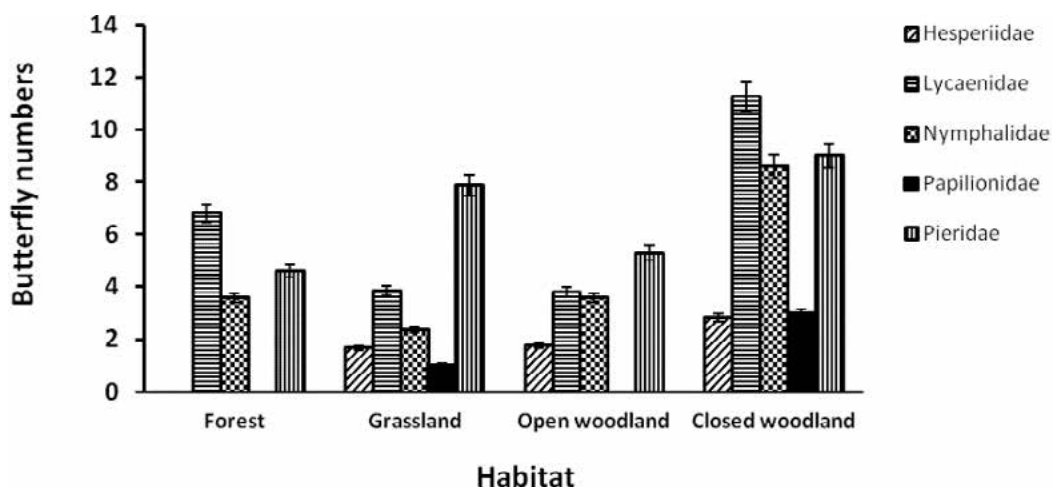


Figure 1: Graph illustrating the species diversity of butterfly in different family and habitat types in Moyowosi Game Reserve

Table 1: Butterflies individuals in different family and habitat types in Moyowosi Game Reserve, in brackets are mean abundance and their standard error (open and closed Miombo woodland).

Family	Habitat types				Total	No. of species	% capture
	Closed Woodland	Forest	Grassland	Open Woodland			
Hesperidae	51 (2.83 $\pm$ 0.51)	2 (0)	5 (1.67 $\pm$ 0.67)	7 (1.75 $\pm$ 0.48)	65	21	7.13
Lycaenidae	90 (11.25 $\pm$ 4.97)	34 (6.8 $\pm$ 1.88)	23 (3.83 $\pm$ 1.14)	19 (3.80 $\pm$ 1.16)	166	11	18.20
Nymphalidae	252 (8.60 $\pm$ 1.97)	43 (3.58 $\pm$ 0.62)	64 (2.37 $\pm$ 0.40)	93 (3.58 $\pm$ 0.66)	452	47	49.56
Papilionidae	12 (3.0 $\pm$ 1.41)	-	2 (1 $\pm$ 0)	2 (0)	16	6	1.75
Pieridae	90 (9.0 $\pm$ 4.77)	23 (4.6 $\pm$ 1.83)	63 (7.88 $\pm$ 4.25)	37 (5.29 $\pm$ 1.96)	213	13	23.36
Total by habitat	495 (7 $\pm$ 1.24)	102 (4.43 $\pm$ 0.68)	157 (3.41 $\pm$ 0.81)	158 (3.6 $\pm$ 0.53)	912	98	100
% capture	54.28	11.18	17.21	17.32	100		

Species diversity

The highest Shannon-Weaver Index (H') of 3.51 was recorded in closed miombo woodland, followed by open miombo woodland ($H' = 3.401$), grassland ($H' = 3.401$) while forest had the lowest H' (2.897). Across families, diversity of Nymphalidae was highest ($H' = 3.071$), followed by Hesperidae ($H' = 2.815$), Lycaenidae ($H' = 1.746$), Pieridae ($H' = 1.55$), and least diversity in Papilionidae ($H' = 1.44$). Across habitat types, Nymphalidae presented higher diversity in grassland ($H' = 3.005$), open woodland ($H' = 2.92$) and closed woodland ($H' = 2.756$), followed by Hesperidae ($H' = 2.629$) in closed woodland. No records of

Hesperidae and Papilionidae were observed in forest habitat and open woodland respectively. Results indicated that butterfly species found in open woodland, forest, and grassland were more closely related compared to those found in closed woodland (Fig. 2). Results in Figure 3 indicated that more than half (55.1%) of butterfly species were habitat-specific such that (27.6%) showed specificity to closed woodland habitat followed by grassland (13.3%), open woodland (12.2%), and forest (2%). Further results showed that only 14.3% of butterfly species were not habitat-specific. The list of species in different habitats is shown in Appendix 1.

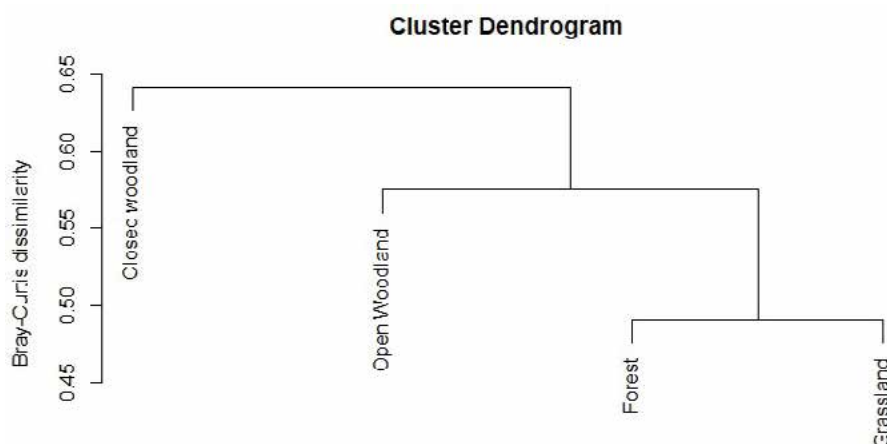


Figure 2: Cluster analysis of butterflies' overall species composition in different habitats in Moyowosi Game Reserve.

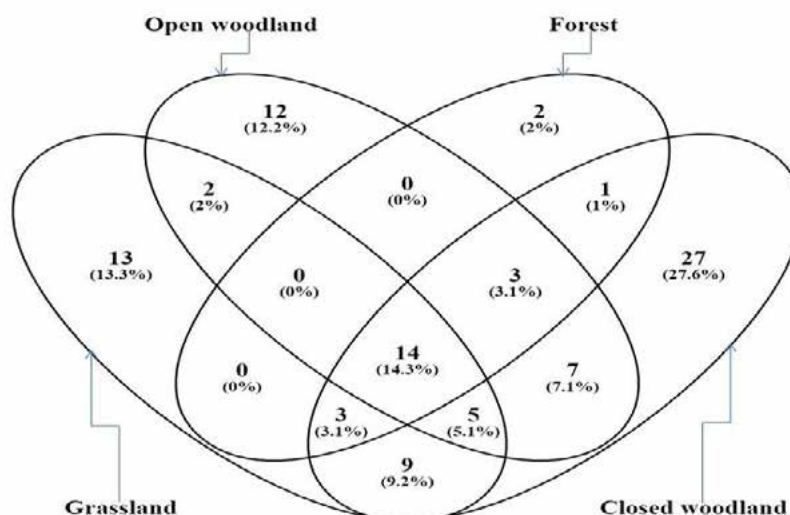


Figure 3: Overall butterflies' species distribution in different habitats of Moyowosi Game Reserve, western Tanzania

We found variability in species composition similarities among habitat types (Table 2). Open woodland and grassland had a higher Jaccard Similarity Index ($J_i \times 100$), and lowest between forest and closed woodland (Table 2). Even though the scale of Jaccard similarity index (J_i)

varies from 0 to 100 where values closer to 100 means higher similarities (Magurran 2004). These results show that most comparisons were below 0.5 (50%) similarities in species composition.

Table 2: Jaccard Similarity Index ($J_i \times 100$) of butterfly species between habitat types

Habitat type	Closed woodland	Open woodland	Forest	Grassland
Closed woodland				
Open woodland	32			
Forest	28	35		
Grassland	34	41	31	

DISCUSSION

Butterflies species abundance

Our results revealed the existence of 5 families in the MGR, while in the whole of Tanzania there are a total of 7 butterfly families (Kielland 1990). This finding suggests that the game reserve possesses healthy and heterogeneous habitats for butterfly conservation. A similar pattern was observed by previous studies in other countries (Tiple and Khurad 2009, Bonebrake *et al.*, 2010). Our study recorded higher species richness and diversity of Nymphalids in grassland habitats than Pierids. In addition, Nymphalids were more abundant and diverse than Pierids across all habitat types. Pierids took the second level of dominance across habitat types. These disparities may be caused by the level of disturbance observed in the two areas. The species abundance was lower in grassland compared to other habitats because of the low abundance of food plant resources for butterflies in the area. The study by Fitzherbert *et al.* (2006) in Katavi National Park highlighted similar factors: they reported that most species in the Pieridae family are generally grassland specialists and are more abundant and diverse in grassland habitats. However, this is contrary to our observation in Moyowosi Game Reserve, though both

of the two sites are in western Tanzania. The huge number of Pierids indicates disturbance and the presence of Nymphalids indicates less disturbances. Disturbance in habitats can be caused by anthropogenic activities or natural origins. Examples of anthropogenic disturbance include frequent burning, burning was the major problem observed in the miombo of Moyowosi Game Reserve. Burning favours grassland; in our study, burning created wooded grassland (open woodland) dominated by sparse miombo trees which may favour Nymphalids. This indicates that miombo in Moyowosi Game Reserve still receiving moderate disturbance. However, this may need further research to establish the cause of the observed variations.

Butterfly species diversity

This study recorded low diversity and abundance of butterflies in forest habitats presenting a slightly different pattern from other studies. These findings contradict many studies (Brown and Hutchings. 1997, Fitzherbert *et al.*, 2006, Moyer *et al.*, 2006, Munisi *et al.*, 2024) who documented that forests are highly diverse taxonomically compared to other habitat types, example study conducted in Katavi National Park (Fitzherbert *et al.*, 2006) and Ntakata Forest (Nkwabi *et al.* 2021) confirmed presence of higher diversity of butterfly in

forested habitat. On the other hand, in this study overall butterfly species diversity was highest in closed and open miombo woodland implying more availability of resources in that habitat; these findings concurred with previous studies (Jew *et al.*, 2015, Norfolk *et al.*, 2017, Topp, and Loos, 2019, Munisi *et al.*, 2024). However, these results provided confirmation that closed and open miombo woodland habitats accommodated higher species diversity number of habitat-restricted species. Such results support the theory that the higher tree abundance and canopy cover in an area contributes more biodiversity conservation than in treeless areas such as grassland habitat (Moreno *et al.*, 2016). However more research to establish the critical reasons for this is needed, for example by standardizing sampling efforts between habitat types rather than having plots sampled as forest being a smaller percentage compared to woodland plots. Habitat specificity of more than half of the overall butterfly species in the study area signifies the importance of maintaining the habitats in pristine conditions. This is because either destruction or changes due to anthropogenic or climatic effects can even result in local extinction of more than half of butterflies' species, as reported by Jew *et al.* (2015). The only butterfly species that might be resilient to habitat disturbances in the study area include *Mylothris agathina*, *Hamanumida daedalus*, *Leptotes pirithous*, *Neptis penningtoni*, *Eurema regularis regularis*, *Pseudargynnis hegemone*, *Pentila pauli clarensis*, *Precis archesia*, *Eurema upembana*, *Precis octavia sesamus*, *Uranothauma cordatus*, *Precis ceryne ceryne*, *Catopsilia florella* and *Danaus chrysippus chrysippus*. The species are disturbance-adaptable species, or generalists, meaning that the species are able to use a wider range of host plants as resources. It has been reported that disturbance-adaptable species possess more than one annual generation in different varieties of host plants for survival (Hogsden and Hutchinson 2004). Therefore, any change in habitat requirement will not cause species local extinction.

In addition, our finding showed there are some species of butterfly occurring in all habitats identified in the study area, indicating that the observed species are habitat generalists. The same pattern was observed with several previous studies (Hamer *et al.*, 2015, Schmitt *et al.*, 2021; Munisi *et al.*, 2024). The presence of generalist butterfly species would profit them by exploiting available resource where by ecologically benefiting in maintaining a better population size (Topp, and Loos, 2019). Despite of that our finding showed that some butterfly species are also habitat specialists, as three species were recorded including *Belenois calypso minor*, *Junonia artaxia* and *Precis ceryne ceryne*. These species only forage in miombo woodland (Jew *et al.*, 2015). Therefore, these findings suggest that butterflies are either habitat specialists or generalists (Musarandega, 2015), because many factors including climate, resource availability, and habitat diversity are determinants of this distributional pattern. According to Menéndez *et al.* (2007), climate variables have strong effects on generalists whereas the host-plant richness and habitat diversity contribute to habitat specialists.

CONCLUSION AND RECOMMENDATIONS

We have demonstrated that MGR is relatively rich, diverse, and has high butterfly abundance. The observed pattern in species richness, abundance, and diversity consistently shows the remarkable species' distribution across habitat types indicating a healthy baseline for the ecosystem. The pattern of butterflies' distribution and species composition could be due to vegetation heterogeneity. We also show that butterflies had strong species composition similarities between different habitat types. The presence of many flowering plants across the area habitats provides food requirements, cover, microclimates, and supports larvae as well as adults to survive. We suggest further studies in MGR should consider different seasons and habitats and should be repeated to monitor changes in species composition and diversity; our study was conducted in a

wet season without repeated sampling efforts. From the results obtained MGR should focus on preserving the Miombo woodland (open and closed) by sustaining the habitat steadiness as accommodates a large number of butterfly species diversity and abundance compared to other habitat types. To make sure the conservation of these unique habitats is taken care of by game reserve managers as well as conservationists they should start a long-term monitoring program and research to examine any changes in butterfly species over time and space.

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Appendix 1: Butterfly species observed in different habita types and family (identification based on Kielland 1990, Larsen 1996, 2005, Martins and Collins 2016)

No.	Family and Latin name	Habitat type						
		Miombo specialists	Closed Miombo woodland	Forest	Grassland	Open Miombo Woodland	Total	% capture
Hesperiidae			51	2	5	7	65	7.13
1	<i>Acada biseriatus</i>		5			2	7	0.77
2	<i>Andronymus neander neander</i>		7				7	0.77
3	<i>Borbo fatuellus fatuellus</i>		1				1	0.11
4	<i>Chondrolepis telisignata</i>		2		1	1	4	0.44
5	<i>Coeliades libeon</i>		1				1	0.11
6	<i>Eretis umbra maculifera</i>				3		3	0.33
7	<i>Fresna cojo</i>		1				1	0.11
8	<i>Fresna netopha</i>		4				4	0.44
9	<i>Fresna nyassae</i>		6				6	0.66
10	<i>Gegenes niso brevicornis</i>		2				2	0.22
11	<i>Hypoleucis tripunctata draga</i>		1				1	0.11
12	<i>Metisella midas midas</i>			2			2	0.22
13	<i>Monza alberti</i>		5				5	0.55
14	<i>Pelopidas mathias</i>		1				1	0.11
15	<i>Prosopalpus styła</i>		1				1	0.11
16	<i>Sarangesa brigida atra</i>					1	1	0.11
17	<i>Sarangesa motozi</i>		2		1		3	0.33
18	<i>Tagiades flesus</i>		1				1	0.11
19	<i>Teniorhinus herilus</i>		1			3	4	0.44
20	<i>Teniorhinus horona</i>		6				6	0.66
21	<i>Zenonia zeno</i>		4				4	0.44
Lycaenidae			90	34	23	19	166	18.20
22	<i>Axiocerses tjoane</i>		3				3	0.33
23	<i>Baliochila hildegarda</i>		7	9	8		24	2.63
24	<i>Cigaritis nyassae</i>		1				1	0.11
25	<i>Euchrysops malathana</i>		3	2		3	8	0.88
26	<i>Euchrysops osiris</i>				1		1	0.11
27	<i>Hypolycaena hatita ugandae</i>				1		1	0.11
28	<i>Leptotes pirithous</i>		18	12	6	7	43	4.71
29	<i>Lolaus frater frater</i>					1	1	0.11
30	<i>Ornipholidotos peucetia peucedata</i>		12				12	1.32
31	<i>Pentila pauli clarensis</i>		43	8	4	2	57	6.25
32	<i>Uranothauma cordatus</i>		3	3	3	6	15	1.64

No.	Family and Latin name	Habitat type						
		Miombo specialists	Closed Miombo woodland	Forest	Grassland	Open Miombo Woodland	Total	% capture
Nymphalidae			252	43	64	93	452	49.56
33	<i>Acraea caecilia</i>		1		1		2	0.22
34	<i>Acraea egina</i>		3		1	4	8	0.88
35	<i>Acraea encedana</i>		1				1	0.11
36	<i>Acraea eponina eponina</i>		1		1		2	0.22
37	<i>Acraea leucographa</i>					2	2	0.22
38	<i>Acraea neobule neobule</i>				1		1	0.11
39	<i>Acraea zetes</i>				1		1	0.11
40	<i>Amauris oscarus oscarus</i>		1			4	5	0.55
41	<i>Amauris tartarea</i>		2				2	0.22
42	<i>Aterica galene</i>		2		3	1	6	0.66
43	<i>Bematistes poggei</i>				1		1	0.11
44	<i>Bicyclus buea</i>		38		3	2	43	4.71
45	<i>Bicyclus mollitia</i>		3				3	0.33
46	<i>Bicyclus safitza safitza</i>		23	7		4	34	3.73
47	<i>Byblia anvatara acheloia</i>		34		3	14	51	5.59
48	<i>Catacroptera cloanthe cloanthe</i>		1			1	2	0.22
49	<i>Charaxes achaemenes</i>					2	2	0.22
50	<i>Charaxes brutus</i>					4	4	0.44
51	<i>Charaxes etheocles evansi</i>					1	1	0.11
52	<i>Charaxes protoclea azota</i>					1	1	0.11
53	<i>Cymothoe coranus</i>					1	1	0.11
54	<i>Danaus chrysippus chrysippus</i>		2	1	2	4	9	0.99
55	<i>Gnophodes betsimena diversa</i>		2			1	3	0.33
56	<i>Hamanumida daedalus</i>		8	2	8	6	24	2.63
57	<i>Junonia artaxia</i>	V	4	4	1		9	0.99
58	<i>Junonia natalica</i>		9	2	2		13	1.43
59	<i>Junonia oenone oenone</i>		1		1		2	0.22
60	<i>Junonia orithya madagascariensis</i>				1		1	0.11
61	<i>Kallimoides rumia ratrayi</i>				1		1	0.11
62	<i>Melanitis leda</i>		8	2		1	11	1.21
63	<i>Melanitis libya</i>				1		1	0.11
64	<i>Neptidopsis ophione velleda</i>				1		1	0.11
65	<i>Neptis gratiosa</i>		9		2		11	1.21
66	<i>Neptis nemetes nemetes</i>					2	2	0.22
67	<i>Neptis penningtoni</i>		24	8	4	6	42	4.61
68	<i>Neptis poultoni</i>		1				1	0.11

No.	Family and Latin name	Habitat type						
		Miombo specialists	Closed Miombo woodland	Forest	Grassland	Open Miombo Woodland	Total	% capture
69	<i>Phalanta phalantha</i>		1		2		3	0.33
70	<i>Physcaeneura leda</i>		1				1	0.11
71	<i>Precis antilope</i>				1		1	0.11
72	<i>Precis archesia</i>		21	2	4	6	33	3.62
73	<i>Precis ceryne ceryne</i>	V	22	4	6	3	35	3.84
74	<i>Precis octavia sesamus</i>		13	5	8	5	31	3.40
75	<i>Precis tugela</i>					2	2	0.22
76	<i>Protonotomorphia parhassus parhassus</i>				1	1	2	0.22
77	<i>Pseudacraea deludens echerioides</i>		3	3			6	0.66
78	<i>Pseudacraea lucretia protracta</i>					2	2	0.22
79	<i>Pseudargynnis hegemon</i>		13	3	3	13	32	3.51
Papilionidae			12		2	2	16	1.75
80	<i>Graphium angolanus</i>					2	2	0.22
81	<i>Graphium polices</i>				1		1	0.11
82	<i>Papilio dardanus</i>		1				1	0.11
83	<i>Papilio nireus lyaeus</i>		3				3	0.33
84	<i>Papilio phorcas</i>		7		1		8	0.88
85	<i>Papilio phorcas tenuifasciatus</i>		1				1	0.11
Pieridae			90	23	63	37	213	23.36
86	<i>Appias sabina phoebe</i>				1		1	0.11
87	<i>Belenois crawshayi</i>				1	1	2	0.22
88	<i>Belenois calypso minor</i>	V		2			2	0.22
89	<i>Belenois sudanensis pseudodentigera</i>		4		4		8	0.88
90	<i>Belenois thysa</i>		1			1	2	0.22
91	<i>Catopsilia florella</i>		1	1	2	1	5	0.55
92	<i>Colotis rogersi</i>		2		1		3	0.33
93	<i>Eurema hapale</i>		1			9	10	1.10
94	<i>Eurema hecabe solifera</i>		1				1	0.11
95	<i>Eurema regularis regularis</i>		37	10	34	13	94	10.31
96	<i>Eurema upembana</i>		4	2	2	10	18	1.97
97	<i>Mylothris agathina</i>		38	8	18	2	66	7.24
98	<i>Mylothris rueppellii tirikensis</i>		1				1	0.11
Total			495	102	157	158	912	100

BUTTERFLY SPECIES COMPOSITION IN VARIOUS HABITAT TYPES IN THE AGA KHAN UNIVERSITY'S ARUSHA CLIMATE AND ENVIRONMENTAL RESEARCH CENTRE

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ABSTRACT

This study assessed the composition, richness, and diversity of butterfly species across different habitat types within the Aga Khan University's Arusha Climate and Environmental Research Centre. Surveys were conducted between December 2023 and January 2024, using a sweep net and baited butterfly traps across thirteen transects representing forest, riverine forest, grassland, and cultivated habitats. A total of 161 butterfly individuals belonging to five families and 55 species were recorded. Riverine forest habitats, particularly Transect Riverine Forest 2, supported the highest richness and diversity of species, while cultivated habitats recorded the lowest. Family Nymphalidae was the most abundant and widespread across all habitats, whereas Papilionidae and Hesperidae were the least represented and restricted to specific habitats. Species diversity was consistently higher in structurally complex and less disturbed habitats compared to simplified agricultural landscapes. These findings highlight the importance of heterogeneous habitats, especially riverine and mixed forest systems, in sustaining butterfly diversity within peri-urban landscapes.

Keywords: biodiversity, butterfly, diversity, conservation, habitats

INTRODUCTION

Insects play a fundamental role in maintaining ecosystem functioning through processes such as pollination, nutrient cycling, herbivory, and serving as a key food source for higher trophic levels. Butterflies belonging to Order Lepidoptera are particularly important due to their dual ecological roles as pollinators in the adult stage and herbivores during the larval stage. They are the second largest, the most widespread and mostly widely recognizable arthropods in the phylum Arthropoda in the order Lepidoptera. Beyond their ecological

significance, they are widely recognized as ideal bio-indicators for ecosystem function, habitat quality, potential land use change and climate change (Munisi *et al.*, 2024; Ramola *et al.*, 2024; Syaripuddin *et al.*, 2015; Pe'er & Settele, 2008; and Parmesan *et al.*, 2006). They are considered to be good ecological indicators for other invertebrate taxa (Ramola *et al.*, 2024; Kumar *et al.*, 2009; Kremen 1994; & Kremen 1992) and surrogate representatives of environmental quality changes. They are used to predict changes in plant populations (Kremen, 1992). On biological grounds, butterfly diversity could be used as an index

of plant diversity (Amaliyah *et al.*, 2024; Pyle, 1980), considering the existence of co-evolution between butterflies and their larval host plants, microclimate, and habitat structure. There is also a positive correlation between birds and butterflies due mainly to the food chain relationship. The presence of high abundance and diversity of butterflies leads to high abundance and species richness of birds too. In general, butterflies can be termed as very sensitive to changes in the environment. Even a slight change can alter their population assemblage and species richness, which in turn affect all other associated organisms within the same ecosystem.

Butterfly diversity and distribution are strongly influenced by habitat structure, vegetation composition, microclimate, and land-use practices (Guderjan *et al.*, 2023). Habitats with high plant species richness and structural complexity generally support greater butterfly diversity by providing abundant nectar resources, larval host plants, shelter, and suitable breeding sites (Habel *et al.*, 2021). By contrast, habitat simplification resulting from agricultural expansion, urbanization, monoculture plantations, and land degradation often leads to declines in butterfly abundance and species richness. These changes disproportionately affect habitat specialists and moisture-dependent species, while favoring a few disturbance-tolerant generalists (Chaianunporn & Chaianunporn, 2019; Szyszko-Podgórska *et al.*, 2023).

Globally, butterflies have been used extensively to monitor biodiversity trends and assess ecosystem health, particularly in landscapes undergoing rapid transformation (Ombugadu *et al.*, 2023). In tropical regions where insect diversity is exceptionally high but poorly documented, baseline data on butterfly communities remain limited. This knowledge gap is especially pronounced in East Africa where increasing human population pressure, land-use change, and climate variability are reshaping natural and semi-natural habitats at unprecedented rates (Ndandika *et al.*, 2025). Establishing reliable baseline information

on butterfly diversity is, therefore, critical for tracking ecological change, guiding conservation planning, and evaluating the effectiveness of habitat management and restoration initiatives.

In Tanzania, peri-urban landscapes are experiencing rapid conversion driven by agriculture, infrastructure development, and settlement expansion (Komba B.B, 2014; Haule, M. J. 2014). These landscapes often represent mosaics of natural, semi-natural, and heavily modified habitats, creating both challenges and opportunities for biodiversity conservation. While such areas may still support significant insect diversity, continued disturbance and habitat fragmentation threaten the persistence of many species (Ndandika *et al.*, 2025). Understanding how butterflies respond to different habitat types within these mixed-use landscapes is essential for integrating biodiversity conservation into land-use planning and sustainable development strategies.

The Aga Khan University's Arusha Climate and Environmental Research Centre (AKU-ACER) provides a unique setting for examining butterfly diversity across contrasting habitat types. The centre functions as a "living laboratory," encompassing mixed forests, riverine forests, grasslands, plantations, and cultivated farmland within a relatively confined peri-urban environment. Ongoing land-use activities, including agriculture and reforestation efforts, have created a heterogeneous landscape that is ideal for assessing how habitat characteristics influence butterfly assemblages. Despite this, systematic studies documenting butterfly species composition, richness, and diversity within the Arusha Climate and Environmental Research Centre (ACER) landscape have been lacking.

Riverine forests within peri-urban settings are of particular ecological importance, as they often act as biodiversity refugia by maintaining higher moisture levels, stable microclimates, and diverse plant communities (Munisi *et al.*, 2024; Livesley, *et al.*, 2016). These habitats

may support butterfly species that are otherwise absent from surrounding dry or disturbed areas. Conversely, cultivated lands and plantations are typically characterized by reduced vegetation diversity and frequent disturbance, potentially limiting their capacity to support diverse butterfly communities (Kwatrina *et al.*, 2018). Comparative studies across such habitat types are, therefore, essential for identifying areas of high conservation value and informing site-specific management interventions.

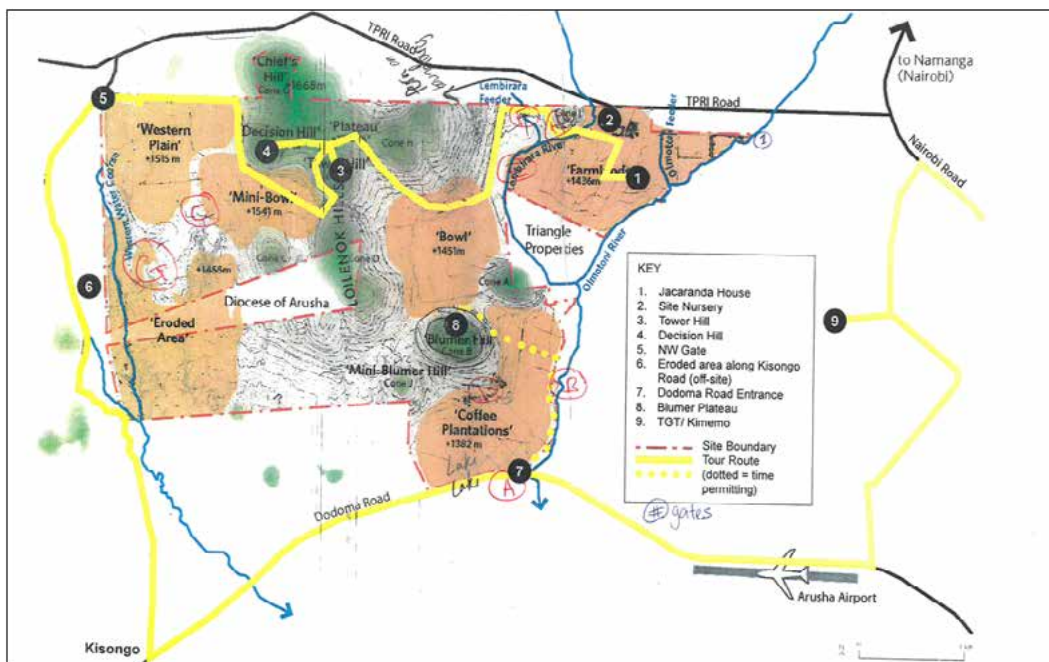
Against this background, the present study aimed to assess the composition, richness, diversity, and abundance of butterfly species across different habitat types within the Manyara and Two Rivers sites of ACER. Specifically, the study sought to document butterfly species occurring in forested, riverine, grassland, and cultivated habitats; compare patterns of species richness, diversity, and abundance among habitat types; and evaluate the influence of habitat structure and land-use practices on butterfly assemblages. The findings of this study provide baseline information that can support long-term biodiversity monitoring,

guide habitat management and restoration efforts, and contribute to broader conservation planning in peri-urban landscapes of Northern Tanzania.

MATERIALS AND METHODS

Study area

The survey was conducted in the two sites of the Aga Khan University, namely Manyara and Two Rivers sites in Arusha region. The Manyara site of the Aga Khan University is located at the edge of Arusha, 13 km West of central Arusha, 3 km North from Arusha airport, and borders with the pastoral herding communities and smallholder farms to the Western and North sides (Map 1). The site has 3,200 acres that are ecologically important. It has seven preserved hills, randomly located in the area. There is also productive land which is currently used for farming activities, while the Western part of the site is degraded with gully erosions. For the last 10 years AKU has been operating a reforestation program to address overgrazed and deforested areas.



Map 1: AKU Arusha Manyara Site - dotted redline is the study area boundary

The Two Rivers site has about 500 acres and is located between two tributaries to the Usa River, 14 km due East of Arusha city centre (Map 2). Currently, about 80% of land is under cultivation and the remaining 20% consists

of scattered trees, shrubland, and open grass areas. The general characteristics of the area include high precipitation, fertile soil, and diverse vegetation.

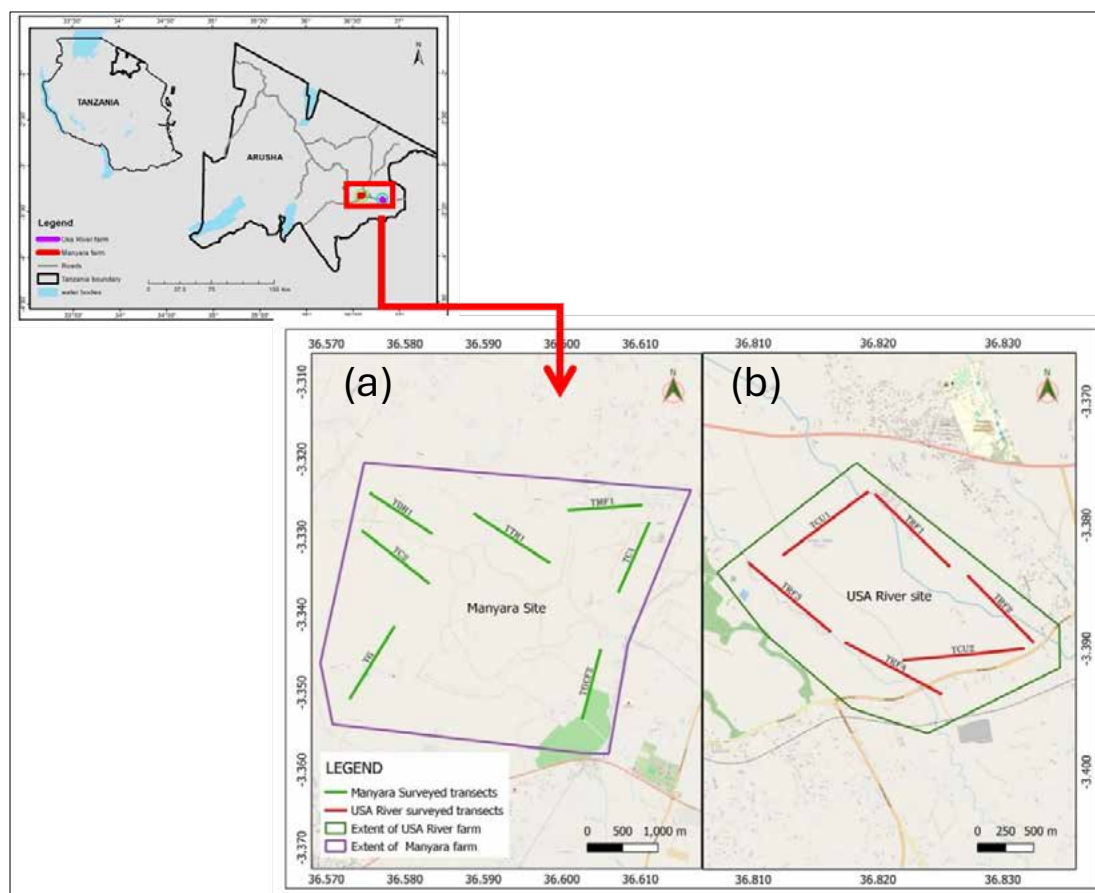


Map 19: AKU Arusha Two Rivers Site – left top-down frames are total cultivated and uncultivated areas respectively. Right top-down frames are grassland and shrubland.

Transects designs

The survey was conducted from December 2023 to January 2024, to assess insect diversity in various habitat types in AKU-ACER. A total of thirteen walking transects of 1 kilometre long was established (Map 3), with seven (7) transects established in Manyara site namely; Transect Mixed Forest (TMF), consisting of eucalyptus plantation and natural forests; Transect Graviillea and Coffee Forest (TGCF); Transect Cultivation Site one (CT1); Transect Cultivation Site two (TC2); Transect Tower

Hill (TTH); Transect Decision Hill (TDH); and Transect Grassland (TG) (Map 3a). Also, 6 transects were established in Two River site namely; Transect Riverine Forest 1 (TRF1); Transect Riverine Forest 2 (TRF2); Transect Riverine Forest 3 (TRF3); Transect Riverine Forest 4 (TRF4); Transect Cultivation 1 (TCU1); and Transect Cultivation 2 (TCU2) (Map 3b).



Map 20: Study site and Transect layout at Manyara and two River site-USA-River in Arusha City.

Data collection and analysis

Ground surveys were conducted to collect butterflies in the two sites studied. In all habitat types, transects of one kilometer long were established with three sampling plots where traps were placed to collect insects. Transects were purposively selected based on their characteristics. In Manyara site, insects were collected from the seven transects consisting of mixed natural forest and plantation, farmland, grassland and bush thickets habitats. In the Two Rivers site, butterflies were collected from the six transects consisting of farmland and riverine forest habitats. Butterflies were collected using two methodologies; sweep (hand) nets and baited butterfly traps. Sweep-nets were used to collect insects that were observed directly by sweeping all butterflies observed within a transect of one kilometer long. In addition, butterfly traps baited with fermented banana hanged on the branches of the trees were used

to collect butterflies. Butterflies were identified to species level with the aid of field guide books (Larsen T, 1996 and Kielland J, 1990).

The diversity and abundance of insects per habitat type in each habitat types were calculated. Shannon–Wiener index (H'), species richness (S) and relative abundance (A) were computed for each surveyed habitat type using the PAST 4.03 statistical software (Hammer *et al.*, 2001).

RESULTS.

Butterfly species composition in the different surveyed plots in Manyara site

In total 161 individuals representing 5 families and 55 species of butterflies were collected in the study area. In Manyara site, 66 individuals of butterflies were collected from the surveyed plots in Manyara site representing four families,

nine subfamilies, fourteen genera and twenty five species of butterflies. When the seven plots were compared Transect Tower Hill 1 (TTH1), had the highest number of individuals (n=23), followed by Transect Mixed Forest 1 (TMF1), n=21 while the Transect Grevillea and Coffee Forest 1(TGCF1) and Transect cultivation 2 (TC2) had the least number of individuals observed n=4 for both (Fig. 1).

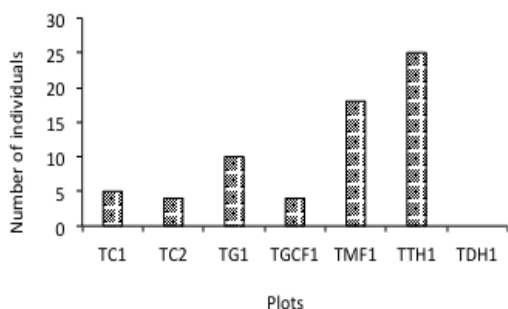


Figure 1: Number of individuals observed in the surveyed plots in Manyara site

Four families of butterflies were observed; Nymphalidae, Pieridae, Hesperidae and Papilionidae. Family Nymphalidae had the highest number of individuals observed (n=33), (Fig. 2) and was the only family that was observed in most of the plots surveyed (Table 1), followed by family Pieridae n=30 while Hesperidae (n=2) and Papilionidae (n=1) was least observed families and was only observed in Transect Mixed Forest 1 and Transect Tower Hill 1 respectively (Table 2).

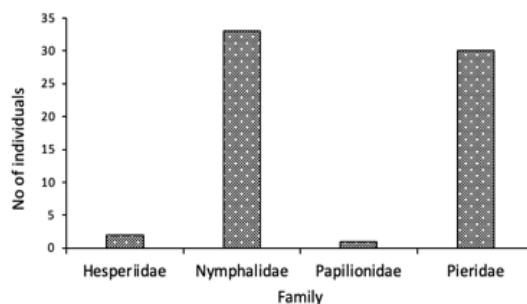


Figure 2: Number of families observed in Manyara site

Table 1: Distribution of butterfly families in different surveyed plots in Manyara site.

Family	TC1	TC2	TG1	TGCF1	TMF1	TTH1	TDH1	Grand Total
Hesperidae					2			2
Nymphalidae	5		2	4	9	13		33
Papilionidae						1		1
Pieridae		4	8		7	11		30
Grand Total	5	4	10	4	18	25		66

A total of 25 species of butterfly were collected from Manyara site (Appendices 1), and according to IUCN categories all species collected are of least concern (LC). *Neocoenya gregorii*, *Danaus chrysippus*, *Charaxes candiope* and *Colotis aurora* were the most observed species in the surveyed plots. While most of the species were observed in only one or two plots, *Danaus chrysippus* was observed in four plots suggesting its preference for the habitats in the respective plots.

The higher species richness was observed in TTH1 and TMF1. Species diversity was higher

in TMF1 ($H' = 2.322$) and TTH1 ($H' = 2.13$) and was lower in TC2 ($H' = 0$) (Fig. 3).

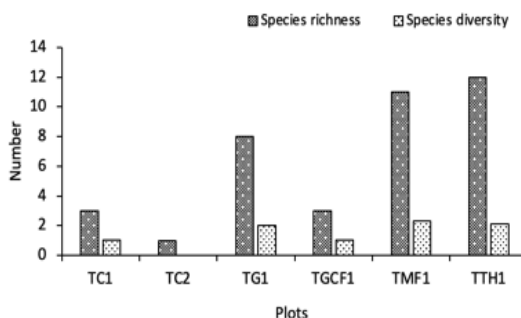


Figure 3. Species richness and diversity in the surveyed plots in Manyara site

Butterfly Species richness, diversity and abundance in two rivers site.

In total 95 individuals of butterflies were collected from the six plots surveyed representing four families, ten subfamilies, twenty six genera and forty six species of insects (Appendices 2). The results shows that there was a higher number of individuals observed in Transect Riverine Forest 2 (TRF2), n=36 and fewer were observed in Transect Cultivation (TCU2), n=2 (Fig. 4).

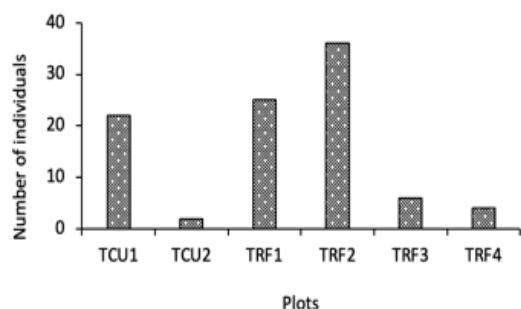


Figure 4: Number of individuals observed in the surveyed plots in Two rivers site

A total of four families of butterflies were also observed in two rivers site, namely Lycaenidae, Nymphalidae, Papilionidae and Pieridae. When the four families are compared, Family Nymphalidae had the highest number of individuals observed (n=57) followed by Pieridae (n=27), (Fig. 5) Papilionidae was the least observed family (n=3). Family Nymphalidae was observed in all plots surveyed while Papilionidae and Lycaenidae were only observed in two surveyed plots. (Table 2).

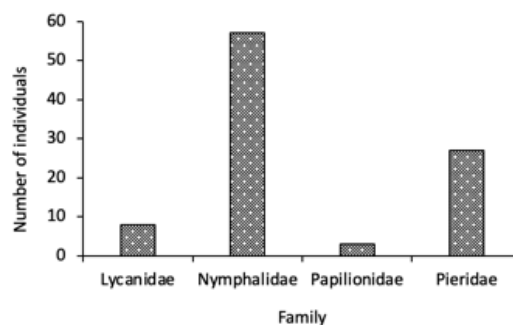


Figure 5: Number of families observed in two rivers site

Table 2: Distribution of butterfly families in different surveyed plots in two rivers site.

Family	TCU1	TCU2	TRF1	TRF2	TRF3	TRF4	Total count
Lycaenidae	6			2			8
Nymphalidae	11	1	19	18	4	4	57
Papilionidae			1	2			3
Pieridae	5	1	5	14	2		27
Total count	22	2	25	36	6	4	95

Our result further shows that a total of forty six species of butterflies were collected in two rivers site. While most of the species were less observed, *Junonia natalica* was the most observed species of butterfly. The highest species richness was recorded in the Transect Riverine Forest 2 and was lowest in Transect Cultivation 2 (Fig. 6). Species diversity was also higher in Transect Riverine Forest 2 ($H' = 2.816$) and was lower in Transect Cultivation 2 ($H' = 0.6931$).

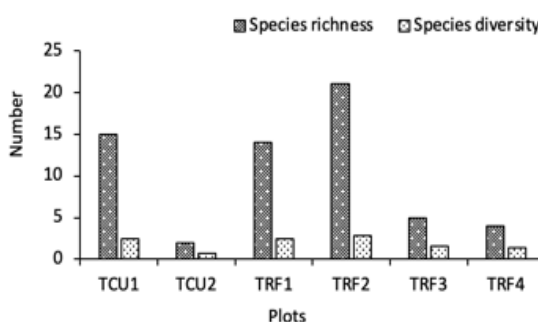


Figure 6. Species richness and diversity in the surveyed plots in two rivers site

DISCUSSION

Butterfly diversity patterns across habitats

The results demonstrate clear differences in butterfly diversity and abundance among habitat types, strongly influenced by habitat structure, vegetation complexity, disturbance levels and environmental conditions (Dawar *et al.*, 2024; Maskell, 2022; Lien & Yuan, 2003; Chaianunporn & Chaianunporn, 2019). Forested and riverine habitats consistently supported higher species richness and diversity compared to cultivated and grassland areas. Our results also reveal these differences across different plots surveyed with specific characterized habitats. This pattern is consistent with previous studies showing that heterogeneous habitats with diverse plant communities provide a wider range of nectar sources, larval host plants, and suitable microclimatic conditions for butterflies (Sagwe, *et al.*, 2015).

In the Manyara site, higher butterfly abundance and diversity were recorded in Transect Tower Hill (TTH) and Transect Mixed Forest (TMF) while less was observed in Transect Grevillea and Coffee Forest (TGCF) and Transect cultivation 2 (TC2). This is attributed to the nature of the habitat of these plots that determine the resource availability necessary to support butterfly abundance and diversity. Transect Tower Hill 1 (TTH1) was dominated by bush thickets and Transect Mixed Forest 1 (TMF1) was also dominated by mixed vascular plants including natural plants and plantations that provided resources to butterflies hence supporting their diversity. These habitats with mixed vegetation, bush thickets, and relatively low disturbance were likely to offer continuous floral resources and shelter, supporting both generalist and specialist species.

By contrast, Transect Grevillea and Coffee Forest (TGCF) were mainly dominated by a few species of grevillea and coffee exhibited lower diversity, reflecting a reduction in the

heterogeneity of the habitat, and limited availability of butterfly resources. Monoculture plantations and intensively managed farmland are known to simplify habitat structure, thereby constraining butterfly assemblages. The more the plant diversity the greater the butterfly diversity as vascular plant provide nectar and pollen, cover of larval food plants, and nesting sites contributed to the high species richness and diversity in the forest (Cassandra *et al.*, 2023).

Grassland habitats recorded low butterfly abundance, possibly due to seasonal dryness in the grassland that restricted the availability of food such as nectar in flowers that adult butterfly depend on for feeding. Moreover butterflies have physiological tolerance limits to drought leading to absence in areas with high temperatures, hence emphasizing landscape management for preventing extinctions (Oliver 2015). Likewise, Niharika *et al.*, 2024 identified temperature and vegetation cover as key factors influencing species richness, while humidity levels correlated with the presence of specific moisture-dependent species.

In the Two Rivers site, riverine forest transects, particularly Transect Riverine Forest 2, supported the highest butterfly richness and diversity. The presence of permanent water sources, high humidity, and dense vegetation likely created favourable microhabitats for butterflies, including moisture-dependent species. Riverine forests often function as biodiversity refugia within modified landscapes, buffering species against climatic extremes and human disturbance. Also increased vascular plant diversity, abundant insect-pollinated plants providing nectar and pollen, cover of larval food plants, and nesting sites contributed to the high richness and diversity of species in the forest (Cassandra *et al.*, 2023). Conversely, cultivated areas at the Two Rivers site supported very few individuals and species, highlighting the negative effects of agricultural disturbance on butterfly communities. The removal of native vegetation, frequent cultivation, and limited floral diversity likely reduced habitat

suitability for both larval and adult stages. Moreover, unavailability of abundant insect-pollinated plants providing nectar and pollen, cover of larval food plants, and nesting sites in the farmland due to disturbance and elimination of plants for cultivation contributed.

Papilionidae was the least observed family and was only observed in riverine forest and was not observed in the farmland (Figure 5). The farmland area was dry and hence unsuitable for Papilionids that mostly prefers moist habitats (Larsen T, 1996).

Family-level patterns

Family Nymphalidae was mostly observed in all plots surveyed except Transect Cultivation 2 (TC2) and Transect Decision Hill (TDH1). Papilionidae and Hesperidae was the least observed family and was only collected in Manyara site. This can be explained by the fact that most of Hesperids are small in size and are hence poorly detected. They are considered as an elusive group which is why they are being frequently omitted from tropical butterfly surveys (Spitzer *et al* 1993, Sparrow *et al* 1994, Lien & Yuan 2003) and need major effort for sampling and identification unlike other butterflies (Brown Jr. & Freitas 2000a, Caldas & Robbins 2003, Dennis *et al* 2006, Basset *et al* 2011). However studies have shown that Hesperiid assemblages may well reflect relationships with environmental factors such as temperature, soil mosaic, or habitat type,. In addition, the richness of the entire family or even of some of its subfamilies can be accurately used as surrogate for the species richness of all butterflies (Carneiro, 2014). Moreover, Papilionids prefers moist habitats (Larsen T, 1996) while all plots surveyed were characterized by dry conditions that were unsuitable for the Papilionidae family. Butterflies were not collected in Transect Decision Hill 1 because of lesser sampling efforts applied in this habitat due to the prevailing conditions including rain.

CONCLUSION AND RECOMMENDATIONS

This study provides baseline information on butterfly diversity and distribution within the Aga Khan University's Arusha Climate and Environmental Research Centre. Richness and diversity of butterfly species were strongly influenced by habitat type, with riverine and mixed forest habitats supporting the most diverse assemblages, while cultivated areas recorded the lowest diversity. Habitat heterogeneity, vegetation complexity, and moisture availability emerged as key drivers of butterfly diversity.

The findings emphasize the conservation value of riverine forests and structurally complex habitats within peri-urban landscapes. Protecting and restoring such habitats, alongside promoting biodiversity-friendly land-use practices, will be critical for sustaining butterfly populations and broader insect biodiversity in the study area. These results can inform future biodiversity monitoring, land-use planning, and conservation initiatives at AKU-ACER and similar landscapes in Tanzania.

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Appendix 1: Checklist of butterflies collected in the surveyed plots in the Manyara site

Order: Lepidoptera			IUCN status: Least concern (LC)								
Family	Subfamily	Genus	Scientific name	TC1	TC2	TG1	TGCF1	TMF1	TTH1	TDH1	Total count
Pieridae	Pierinae	Belenois	Belenois calyposo minor			2					2
Pieridae	Pierinae	Belenois	Belenois creona					2	2		4
Pieridae	Pierinae	Belenois	Belenois gidica			1					1
Pieridae	Coliadinae	Coliadini	Catopsilia florella		4						4
Nymphalidae	Charaxinae	Charaxes	Charaxes baumanni granti					2			2
Nymphalidae	Charaxinae	Charaxes	Charaxes brutus	1				1	1		3
Nymphalidae	Charaxinae	Charaxes	Charaxes candiope	2				3	1		6
Nymphalidae	Charaxinae	Charaxes	Charaxes zoolina					2			2
Pieridae	Pierinae	Colotis	Colotis amatus amatus						1		1
Pieridae	Pierinae	Colotis	Colotis aurigeneus			1		1			2
Pieridae	Pierinae	Colotis	Colotis aurora			2		2	1		5
Pieridae	Pierinae	Colotis	Colotis euipe			1					1
Pieridae	Pierinae	Colotis	Colotis phisadia amatus						1		1
Nymphalidae	Danainae	Danaus	Danaus chrysippus	2			1	1	2		6
Hesperiidae	Pyrginae	Eretis	Eretis leugens					1			1
Pieridae	Coliadinae	Eronia	Eronia cleodora			1			3		4
Pieridae	Coliadinae	Eurema	Eurema desjardinsii						1		1
Nymphalidae	Nymphalinae	Eurytela	Eurytela dryope				1				1
Nymphalidae	Nymphalinae	Junonia	Junonia hierta			1					1
Nymphalidae	Nymphalinae	Junonia	Junonia natalica		0	1					1
Nymphalidae	Nymphalinae	Junonia	Junonia oenone				2				2
Nymphalidae	Satirinae	Neocoenyra	Neocoenyra gregorii						9		9
Papilionidae	Papilioninae.	Papilio	Papilio demodocus						1		1
Pieridae	Pierinae	Pierini	Pontia helice					2	2		4
Hesperiidae	Hesperiinae	Baorini	Zenonia zeno					1			1
Total count				5	4	10	4	18	25	0	66

Appendix 2: Checklist of butterflies collected in the surveyed plots in the Two Rivers site

Order: Lepidoptera		IUCN status: Least concern (LC)								
Family	Subfamily	Genus	Scientific name	TCU1	TCU2	TRF1	TRF2	TRF3	TRF4	Total
Nymphalidae	Acraeinae	Acraea	Acraea asebria			3				3
Nymphalidae	Acraeinae	Acraea	Acraea eponina eponina	1						1
Nymphalidae	Acraeinae	Acraea	Acraea poggei			1				1
Pieridae	Pierinae	Belenois	Belenois calyposo minor				2			2
Pieridae	Pierinae	Belenois	Belenois creona				1			1
Nymphalidae	Satyrinae	Bicyclus	Bicyclus dentatus					1		1
Nymphalidae	Satyrinae	Bicyclus	Bicyclus safitza			1		2		3
Nymphalidae	Nymphalinae	Bicyclus	Byblia ilithyia		1					1
Pieridae	Coliadae	Coliadini	Catopsilia florella		1		2			3
Nymphalidae	Charaxinae	Charaxes	Charaxes baumanni granti						1	1
Nymphalidae	Charaxinae	Charaxes	Charaxes candiope						1	1
Lycaenidae	polyommatae	Chilades	Chilades kedonga	1			1			2
Lycaenidae	polyommatae	Chilades	Chilades trochylus	1						1
Pieridae	Pierinae	Colotis	Colotis amatus amatus				1			1
Pieridae	Pierinae	Colotis	Colotis aurora	1			1			2
Pieridae	Pierinae	Colotis	Colotis euppe				2	1		3
Pieridae	Pierinae	Colotis	Colotis regina			2				2
Lycaenidae	Polyommatae	Cupidopsis	Cupidopsis iobates	1			1			2
Nymphalidae	Danainae	Danaus	Danaus chrysippus				1			1
Pieridae	Coliadae	Eronia	Eronia cleodora				1			1
Lycaenidae	Polyommatae	Euchrysops	Euchrysops nandensis	1						1
Pieridae	Coliadae	Eurema	Eurema desjardinsii	1		1		1		3
Pieridae	Coliadae	Eurema	Eurema hapale	1						1
Nymphalidae	Satyrinae	Gnophodes	Gnophodes betsimena			3	1			4

Family	Subfamily	Genus	Scientific name	TCU1	TCU2	TRF1	TRF2	TRF3	TRF4	Total
Papilionidae	Papilioninae.	Graphium	Graphium leonidas				1			1
Papilionidae	Papilioninae.	Graphium	Graphium philonoe			1				1
Nymphalidae	Nymphalinae	Hypolimnas	Hypolimnas misippus				2			2
Nymphalidae	Nymphalinae	Junonia	Junonia hierta			1	1			2
Nymphalidae	Nymphalinae	Junonia	Junonia natalica	6			5		1	12
Nymphalidae	Nymphalinae	Junonia	Junonia oenone	2						2
Nymphalidae	Nymphalinae	Junonia	Junonia orithya				1			1
Nymphalidae	Nymphalinae	Junonia	Junonia sophia	1			6			7
Nymphalidae	Nymphalinae	Junonia	Junonia terea elgiva			2	1			3
Lycaenidae	Polyommatainae	Lampides	Lampides boeticus	1						1
Pieridae	Pierinae	Leptosia	Leptosia alcesta	2			3			5
Nymphalidae	Satyrinae	Melanitis	Melantis leda			4		1		5
Pieridae	Pierinae	Nepheronia	Nepheronia argia			1				1
Pieridae	Pierinae	Nepheronia	Nepheronia thalassina			1				1
Nymphalidae	Biblidinae	Neptidopsis	Neptidopsis ophione			1				1
Nymphalidae	Nymphalinae	Neptis	Neptis penningtoni	1						1
Nymphalidae	Nymphalinae	Neptis	Neptis saclava marpessa			3				3
Papilionidae	Papilioninae.	Papilio	Papilio nireus				1			1
Pieridae	Pierinae	Pierini	Pontia helice				1			1
Nymphalidae	Nymphalinae	Salamis	Salamis anacardii						1	1
Lycaenidae	Polyommatainae	Zizeeria	Zizeeria knysna	1						1
			Grand Total	22	2	25	36	6	4	95

ASSESSMENT OF THE OPPORTUNITIES AND CHALLENGES FACING BEEKEEPING ACTIVITIES AT RAU FOREST RESERVE, MOSHI

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ABSTRACT

Beekeeping is an important source of income and supports many rural economic activities, especially agriculture. The activity promotes conservation of habitat in several key areas and yet remains underperformed in many regions. This study examined the socioeconomic and environmental factors influencing beekeeping activities at Rau Forest Reserve in Moshi Urban. Data were collected from Mabogini and Mandaka villages, which were randomly selected from 25 villages recognized to border the Rau forest alongside the major economic activities conducted. A total of 93 respondents, representing 5% of registered households in each village, were randomly sampled and interviewed. Data were analyzed by using Statistical Package for the Social Sciences (SPSS) version 25. Chi-square test at 0.05 significance level was used to determine the relationship between variables. Poor quality of packaging materials, lack of technical skills and pests were the major challenges hindering local communities in beekeeping activities. The result also identified opportunities for local communities to participate in the activity, including the presence of a river in an area, availability of flowering plants as well as the presence of a Forest Reserve. Conclusively, the majority of local communities produced honey for home use and not for the market. The study recommends the provision of technical skills to local beekeepers, improving access to modern equipment and packaging materials to sustain beekeeping activities at Rau Forest Reserve, Urban area.

Keywords: Beekeeping, Challenges, Forest reserve, Opportunities, Urban

INTRODUCTION

Beekeeping is an art and science of managing colonies of bees for the production of honey and other hive products (URT, 2020). It is one of the agricultural sub-sectors that most suits both rural and non-rural people. It is simple and relatively cheap to start, as it requires a very low level of inputs such as labor, capital and knowledge (Bett, 2021). Beekeeping is practiced worldwide and applied by different countries (Bradbeer, 2023). For example, in Germany, it has been practiced for quite some years with the major aim of promoting recreational activities, pollination and obtaining the bee products such as honey, bee wax, propolis for food, medicine and income (Münstedt *et al.* 2020). Globally,

beekeeping plays a major role in improving the living standard of both small-scale farmers rural and urban people (Nkumilwa, 2020). Mostly beekeeping activities provide job opportunities to most of indigenous people and act as a source of food and income. It is a source of recreation, ecotourism and foreign exchange earnings (Beyene, 2022). The most common known bee product, which has been used for several purposes, is honey. Honey, in most cases, is used as a traditional medicine as well as a source of additional nutrients to the body once taken as food (Otero & Bernolo, 2020). Other beekeeping products include, beeswax, propolis and the bees provide pollination services (Otero & Bernolo, 2020).

In Africa, Beekeeping has always been considered as an activity which promotes conservation of forests as well as biodiversity (Bradbear, 2023). Several countries, including Tanzania, are known to conduct beekeeping activities with the major aim of promoting conservation and alleviating poverty (URT, 2020). It is the source of food and provides raw materials for industries, i.e. beeswax. Furthermore, the sector offers job opportunities to a large number of people in Africa, especially those living adjacent to the forest and woodland areas (Richardson, 2023). Despite the suitability of the sector there has been occurrence of several factors which hinder the development of beekeeping activities in Africa (Paji, 2019).

Recently, the use of local knowledge in carrying out beekeeping activity is the major constrain which faces the beekeepers in Africa (Bradbear, 2023). It contributes to the production of low quality honey bee products (Teferi, 2020). For example, the use of fire and smoke to rid the bees from their nest has led to harvesting of the honey with parts of the combs in it; ash and brood (Teferi, 2020). The honey, which is not properly harvested, will obviously lack market, it will have nowhere to be sold, and as an option, it may be treated as waste, leading to great loss and disappointment to beekeepers (Echodu, 2023). This kind of practice not only affects the quality of the honey but also goes beyond affecting the surrounding environment in terms of pollination services and may also destroy the entire colony of the bees and ignite the wildfire (Hilmi, 2024).

Ethiopia is the leading country for honey bee production in Africa and one of the ten largest honey producing countries in the world (Beyene, 2022). It ranks ninth in honey and third in beeswax production worldwide (Terefi, 2021). The country is rich in flowering plants diversity and possesses favorable climatic conditions that provide an ideal environment for beekeeping activities (Terefi, 2021). Beekeeping practices contribute direct and indirect in raising the economy of Ethiopia. Direct through collection of bee product

and indirect through conservation of natural resources as well as increase in crop production through pollination (Beyene, 2022).

Tanzania is the second largest honey producer country in Africa after Ethiopia (Nicholaus & Vanhaverbeke, 2018). More than 50% of land in Tanzania is covered by forest and multi-vegetation which provide a suitable environment for carrying out beekeeping activities (Nicholaus & Vanhaverbeke, 2018). This is to say there are about 33.5 million hectares of forest throughout the country in which the beekeeping activities can be carried out (Tubuta *et al.*, 2022). Being the fact that Tanzania owns about 9.2 million honeybee colonies, making a potential production of about 138,000 tons of honey and 9,200 tons of beeswax per year (URT, 1998). It is estimated that the sector generates about US\$ 1.7 million each year from sales of honey and beeswax and employ about 2 million rural people thus making the beekeeping sector proven to be one of the sector which improves the livelihood of most of the rural communities, especially those adjacent forest and woodlands in terms of income as well as employment (Mwakatobe and Mligwa, 2014).

Beekeeping has been practiced in Tanzania for many generations thus considered to serve as an alternative activity to environmentally destructive activities such as poaching, charcoal burning, as well as slash (Tutuba *et al.* 2022). With regard to Tutuba *et al.* (2022), the process has been carried out and managed without a policy since 1949, after it was officially documented as a department under the Ministry of Agriculture by the British colonials. In the year 1998, the government of Tanzania formulated a National Beekeeping Policy (URT, 1998). The overall goal of the policy was to facilitate conservation and management of the natural resources for the benefit of present and future generations (Mwakatobe & Mligwa, 2016).

Currently, beekeeping in Tanzania plays a major role in socio-economic development

and environmental conservation (Cahmi *et al.*, 2022). The practice has been serving as the source of food in particular honey, pollen and brood, a source of raw materials for various industries, including beeswax for candles and lubricants, a source of medicine, eg., honey, propolis, royal jelly and a source of income for beekeepers (URT, 1998). Apart from being a source of food, medicine and income, beekeeping practice plays a vital role in poverty reduction and improves conservation of forests (Hausser, 2020b).

However, despite the favorable and suitable environments present in Tanzania for carrying out beekeeping, the sector has not yet been exploited to its significance level (Martin, 2011). Several studies have examined the factors hindering the development of beekeeping in Tanzania, however, limited documentation on the opportunities and challenges faced by beekeepers, specifically at Rau forest Reserve in Moshi. This study assessed the challenges and opportunities faced by beekeepers at Rau forest reserve, with the main focus on social-economic and environmental factors. The findings of this study are very important to beekeepers at Rau Forest and the Kilimanjaro as the region as it will help the authority to know the socioeconomic, political, as well as ecological and environmental challenges facing the beekeepers, thus coming up with the improved practical knowledge which will initiate community participation in beekeeping activities.

MATERIAL AND METHODS.

Description of the study area

The study was conducted at Rau Forest Reserve, found in Moshi (Fig 1), Kilimanjaro region. The forest covers an area of 570 hectares (JJaenssion, 2019) and is one of the urban forests in Tanzania located 3km South East of Moshi Town between Latitude 3° 23' S and Longitude 37° 22' E. Tracing back the history of Rau forest reserve, the Reserve was established by the Government of Tanzania in the year 1957. It is renowned to be the home of *Oxystigma msoo*, the indigenous plant species, and is only found at Rau. The forest hosts the biggest tree in East Africa, *Milicia excelsa*, with a diameter of about 3 meters, which is rarely found in other places. The forest is faced with some threats such as illegal logging, wood fuel collection, and human encroachment for agriculture, invasive species and incidents of wildfires. Its boundaries are: to the north is bordered by Msaranga village; to the southern part is bordered by Kaloleni village, Mabogini village and rice farms. To its East it borders with Mwananguluwe tributary, which is a natural boundary; the western part is bordered by Mjimpya and Njoro villages. Several economic activities are carried out within the surrounding villages, including farming, which is carried out in a large percent as well as beekeeping activities, which decline over time.

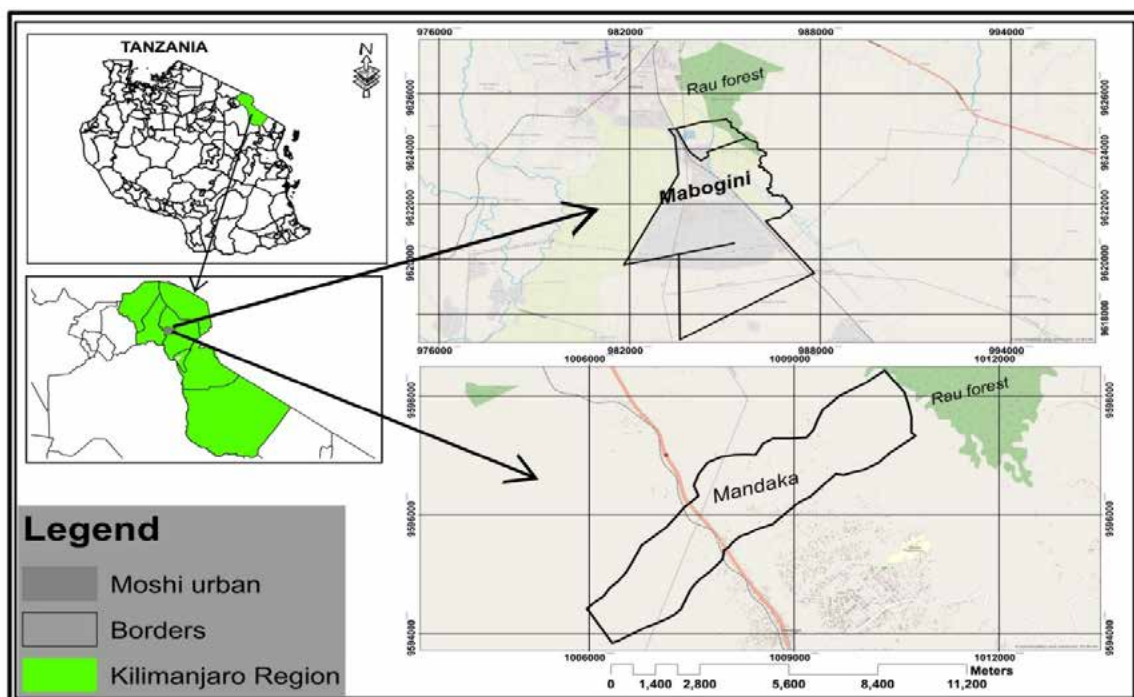


Figure 1: A map of the study area including the two villages (Mandaka and Mabogini) adjacent to Rau forest.

Sample size.

Sample size refers to the number of sampled elements from which the information will be obtained. Five percent (5%) of the total population is sufficient to represent the whole population (Msuya & Kideghesho, 2009). Two villages, Mandaka and Mabogini, were purposively selected because they are the closest to the forest. From these villages 5% of the total registered households were randomly selected to obtain the required information making a total of 93 respondents as described below.

- i. Mabogini sample size = $5/100 \times 1,060 = 53$ individuals.
- ii. Mandaka sample size = $5/100 \times 800 = 40$ individuals.

Making a total of 93 respondents which were involved in the study

Sampling technique.

All villages recognized to border the Rau forest were written on a piece of paper and

simple randomly technique was used to pick only two pieces where two villages (Mabogini and Mandaka) were selected. Purposive sampling techniques is defined as the process which involve picking units which are most knowledgeable about the subject matter of your study (Phrasisombath, 2009). The technique was applied to sample key informants who are more knowledgeable on beekeeping activities practiced at Rau forest Reserve. This included beekeeping officer, forest officer and the village leaders. Snowball technique was applied to sample households of the beekeepers from the two villages who consistently participated in this study.

Data collection methods.

Semi-structured interview was conducted to the key informant within the study area. This then played a big role in obtaining the information regarding the opportunities and challenges facing beekeeping activities at Rau forest. Questionnaire was purposely used to help gathering primary data from the local communities of the two selected villages

adjacent the Rau forest Reserve. Questionnaires (closed ended and open-ended questions) were self-administered and the questions were formulated basing on research questions. The questionnaire consisted of two parts (part A and part B). Part A was purposely for gathering demographic information of the respondents while part B consisted of the questions which helped in collecting information which answers the objectives of the research. .

Data analysis.

Content analysis was used to compute data generated from interviews while data collected from questionnaires were coded and entered using SPSS version 20 software. The Ms excel spread sheet was used to draw graphs and pie chart using the data which were coded and entered in SPSS. Data in SPSS were finally analyzed using descriptive analysis. Chi-square test was used to test whether the answers were significantly different among different variables at 5% level of significance. The analyzed data were presented in descriptive statistics, graphs and tables for easy interpretation.

RESULTS

The finding of this study covers profile of the respondents, potential social economic opportunities and environmental opportunities for beekeeping activities at Rau Forest Reserve in Moshi as well as social economic and environmental challenges facing beekeeping activities at Rau Forest Reserve. A total of 93 respondents were engaged to obtain the required information, and they were selected from the two villages using the snowball sampling technique. (Mandaka and Mabogini). The finding revealed that most of individuals by 92.5% who are engaged in beekeeping activities carry the activity traditionally. They use the local beehives, and the way of maintaining apiaries are done locally.

Profile of the respondents

This part presents the background information of the respondents who participated in the

study. The demographic characteristics of the respondents are characterized by gender, age, education level and duration of stay of the respective respondent.

Table 1: Profile of the respondents

Parameter Characteristics		Frequency	Percent (%)
Gender	Male	65	69.9
	Female	28	30.1
	Total	93	100.0
Age	18yrs-35yrs	9	9.7
	36yrs-53yrs	62	66.7
	above 53 years	22	23.7
	Total	93	100.0

Source: primary data 2020

Respondents by gender

From Table 1, it is observed that the majority of the respondents were male (69.9%) compared to the female (30.1%). This implies that most of the individuals who are interested in beekeeping activities are males.

Respondents by age

From Table 1 above, the majority of the respondents (66.7%) were in the age group 36yrs - 53yrs, (23.7%) had an age of above 53yrs, and the rest (9.7%) were in the age group 18-35yrs. This implies that the individuals who understand the beekeeping activities are the matured youths rather than older and juveniles.

Market challenges

Reported market challenges are inadequate markets for bee products (37.6%), high price of the modern beehives (91.4%), inadequate skills for beekeeping activities (95.7%), shortage of packaging materials (86%), lack of support from the local government (83.9%) and lack of beekeeping training (95.7%) Fig. 2.

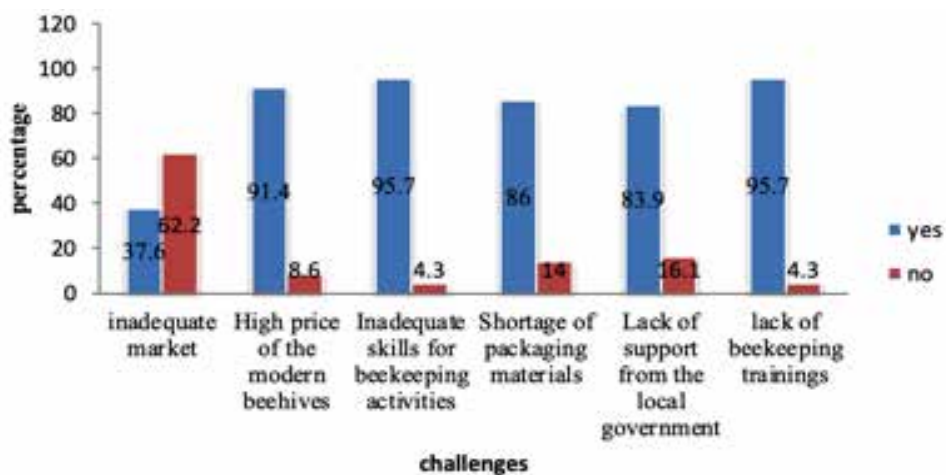


Figure 2: The social economic challenges facing beekeeping activities at Rau forest reserve. *Source: primary data 2020.*

Figure 3 represents the social economic opportunities enhancing beekeeping activities at Rau forest reserve in Moshi. The opportunities include Social economic value of the honey (94.6%) and extension service to support beekeeping activities (16.1%).

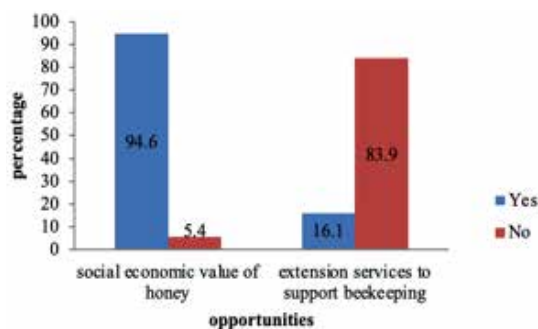


Figure 3: Socioeconomic opportunities: *Source: Primary data 2020.*

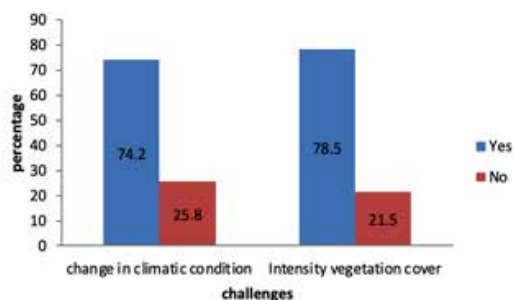


Figure 4: Environmental constraints for beekeeping practices in Rau Forest: *Source: primary data 2020*

Environmental opportunities

Notably, over half of the respondents (52.7%) identified the availability of diverse flowering plant species as a major advantage, as these provide essential nectar and pollen resources required for sustaining healthy bee colonies and improving honey yields. In addition, a larger proportion of respondents (71%) highlighted the presence of a river within the area as a critical opportunity, given that reliable water sources are vital for bee survival, hive temperature regulation, and overall colony productivity (Figure 5).

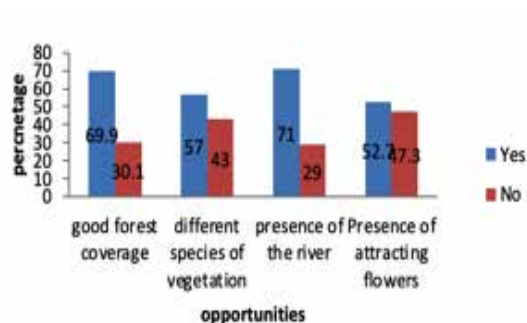


Figure 5: Environmental opportunities for sustainable beekeeping in Rau Forest: *Source: primary data 2020.*

DISCUSSION

This study was set out to examine the opportunities and challenges influencing beekeeping activities in Rau Rorest reserve in Moshi. Findings reveal that beekeepers face a range of constraints, including lack of beekeeping training, inadequate technical skills, scarcity of packaging materials, high costs of the modern equipment and tools, change in climatic conditions and insufficient support from the local government. These challenges collectively hinder productivity and limit the full potential of the sector. Despite of these constraints, the area presents several promising opportunities that can support and sustain beekeeping activities. . These opportunities fall into both social economic and environmental categories which enhance the beekeeping activities. These opportunities fall into both socio-economic and environmental categories. On the one hand, the socio-economic value of bee products provides income-generating potential and livelihood diversification for local communities. On the other hand, favorable environmental conditions, such as the presence of forest ecosystems and a reliable river, offer essential resources for bee survival and productivity. Together, these opportunities provide a strong foundation upon which beekeeping can be strengthened, even in the face of existing challenges. The study shows that most of the respondents were male by 69.9% of the total respondents. This indicates that there is low involvement of women in beekeeping activities, hence little participation and understanding of the beekeeping activities among women. This is probably because beekeeping in East Africa is a male-dominated enterprise

Challenges Facing Beekeeping Activities in Rau Forest Reserve, Moshi

The findings of this study reveal that lack of beekeeping training and inadequate skills for beekeeping activities were the most challenges pinpointed by large percent of respondents. Most of the respondents claimed that they are

not skilled enough to carry out the beekeeping activities in such a way that they fail to understand the exactly time for harvesting the bee products thus ending up to pre harvesting as well as post harvesting of the bee products.

“Keeping of bees is one of my favorite activity which contribute in my daily income generation. But at some point I find myself incurring lost due to the fact that I fail to understand the exactly time for harvesting the bee products, ending up at pre-harvesting and sometimes post harvesting”.(Respondent number 16. Male).

But also, the training provided to them is not sufficient to make them competent enough to carry out the beekeeping activities. They are provided with the education from the non-qualified individuals who are not knowledgeable enough in beekeeping activities as well as the local skilled trainers. The training they receive is often delivered by individuals who lack adequate expertise in beekeeping, including both unqualified facilitators and insufficiently skilled local trainers.

It is infrequently offered in the area and, when available, is not delivered comprehensively enough to equip individuals with the skills required to carry out the beekeeping activities effectively. These findings are consistent with John (2014), which also revealed that the major constraints facing beekeeping activities are inadequate skills as well as poor training provided in carrying out the beekeeping activities.

High price of the modern beehives

Findings reveal that most of the individuals in the area complained about the price of modern beehives. Most of them cannot afford to buy modern beehives, thus forcing them to rely on traditional hives. Most of them wished to be provided with the modern equipment and tools which would help them improve the beekeeping activities rather than buying them, of which most of them cannot afford the

price. Using the local beehives becomes a big challenge for them to harvest the bee products as it was intended. This is just because during the rainy season it becomes difficult for one to keep water from entering the beehives, thus ending up lowering the production of honey as well as the absconding of the bees .

“I cannot afford buying modern beehives because of the price to which they are sold. They are sold at a high price that most of us in our area cannot afford buying them thus end up using the local beehives which then retard the productivity”. (Respondent number 73 Male)

The results are compatible with those of Munuo (2015) which focused on assessing the challenges facing beekeepers in commercializing their products. The study suggests introduction of the modern beehives with affordable price . This will motivate beekeeping activities. Same results are observed to contradict the finding of , Ingram & Hitimana (2017) who argue that as long as an individual can engage in beekeeping, the use of modern beehives is unnecessary.. The introduction of the modern beehive is just the optional means for carrying out the activity and is not a must that an individual should have the modern tool to carry out the beekeeping activities.

Shortage of packaging materials

The materials which are used for packing bee products are still of low quality. Most of the respondents were found using re-used “Konyagi” bottles for packing bee products . Beekeepers tend to wash Konyagi bottles, dry them off and there after packing the products ready to sell. This then decreases the quality of packed products. There should be introduction of the specialized packing materials to improve quality and branding of bee products .

Lack of support from the local government

The local government demonstrates minimal to no involvement in beekeeping activities, largely due to limited awareness and understanding of

the sector. of the activity. The local government authority provides little support to encourage individuals to engage in beekeeping activities. As a result, those who choose to participate often struggle to access technical guidance since extension officers are rarely available and difficult to reach. The local government should serve as a link between the local community and extension officers. Strengthening this connection would likely motivate more individuals to participate in beekeeping and improve the overall effectiveness of the activity.

“The local leaders do not show any support in enhancing the practice of beekeeping activities. Once you are engaged in the beekeeping activities, you receive nothing from the local leaders in terms of support and advice”.

Similar results have been reported by (Tutuba & Vanhaverbeke (2018), who reported that the local leaders have a great chance of developing the beekeeping activities in an area. But once the leaders are not interested in the activity, there is a great possibility of the activity remaining stagnant, showing no positivity of influencing other individuals to participate in the activity.

Inadequate market for bee product

The findings further show marketing of the bee products in the two villages is poorly organized in such a way that most of harvested honey locally consumed within the villages as food and medicine. In the past, honey production was relatively high; however, it has steadily declined over time, and the study now indicates that only small quantities are currently being produced. This is another reason why produced honey ends up being consumed in the level of household.

Change in climatic conditions.

Climate change have been found out to contribute in poor performance in production. These changes has impacted resources,

including the drying of water resources that significantly impacts beekeeping as water as a crucial component in honey production, feeding brood food and hatching bees. Similar results were reported by Gameda (2014) who also claims climate change contributes much in hindering the growth of beekeeping activities. Other challenges which were pointed out include the absence of conducive area for carrying out beekeeping activities.

“I was disappointed with the beekeeping activities after the absconding of the bee from the beehives due to presence of predators”.

Additionally, the study identified pests, diseases, and predators as key challenges affecting beekeeping activities, as they contribute to the absconding of bee colonies.

“I couldn’t continue with the beekeeping activities due to obstructions from the neighbors as they considered the bees as threat and enemies to them.”

Opportunities that enhance beekeeping activities at Rau Forest Reserve in Moshi.

Social economic value of the bee products

Honey is a source of source of carbohydrates and hence, contributes significantly to human nutrition (Martin Hilmi, 2011). However, the study reveals that respondents do not value honey as medicines as well as dietary need.

“First of all, the honey is very sweet I like it, but also for the purpose of curing diseases I have been enhanced to Engage myself in beekeeping activities”.

These findings are consistent with those of (Zone, 2018) who also reveal that the socio economic value of the bee products has been the major reason for individuals to participate in beekeeping activities. Furthermore, this finding is aligned with that of (Vural & Karaman 2015) conducted in Turkey concerning the socioeconomic value of beekeeping activities. The study also identifies the dietary contribution of the bee products in enhancing

a large number of individuals engage in beekeeping activities. The study shows most of the villagers are encouraged to participate in beekeeping activities due to the presence of natural forest adjacent to their areas. Trees are basically important in beekeeping activities as they significantly provide shelter for beehives.

Presence of the river

The study reveals that the area is blessed with several streams of water as well as the river which originate from Rau forest hence the name Rau River.

CONCLUSION

Generally basing on the findings from this study, the majority of the respondents (66.7%) fall within the productive ages (36-53 years). It was realized that the beekeeping activities are still faced with challenges regardless of the countless opportunities present for carrying out beekeeping activity in an area. The major challenges which were observed include a lack of training for the residents, inadequate skills for carrying out the beekeeping activities, as well as the high price of the modern equipment and tools. Apart from that, most of the individuals claimed that the absence of the specialized area obstructs most of them from carrying out the beekeeping activities. Beekeeping activity in an area has not been developed to a satisfactory level due to the fact that most of the beekeepers do attain informal skills from their ancestors. The practice is among others considered to be an additional source of income. Most of them are not exposed to the advanced beekeeping practices, employing traditional hives, hence faced with a lot of challenges which retard their production. Furthermore the area is blessed with potentials for beekeeping practices however, the observation indicates a decreasing number of beekeepers in the area. Conclusively, the beekeeping activities in an area are operated traditionally as a result of low production of bee products, hence a decline in the beekeeping activities.

RECOMMENDATION

As the concluding remark, there should be an introduction of the affordable and appropriate beekeeping technology in the area. This will easily be achieved through the provision of beekeeping training and encouraging the community as a whole. Furthermore, to enhance the beekeeping activities, associations should be encouraged to perform the beekeeping activities rather than individual performance.

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SPATIAL MODELLING OF BEEKEEPING SUITABILITY UNDER CLIMATIC SEASONALITY AND LAND-USE ACCESSIBILITY IN THE GREATER MAHALE ECOSYSTEM, TANZANIA

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ABSTRACT

Identifying where environmental conditions and human pressures interact to influence apiculture potential is crucial for optimising hive placement, enhancing productivity, and informing landscape-level planning in biodiversity-rich areas. In the Greater Mahale Ecosystem (western Tanzania), we modelled beekeeping suitability using MaxEnt (version 3.4.4) with presence-only beehive records, divided into 70% training and 30% testing datasets, and supported by 10,000 background points. Predictor variables were standardised and checked for multicollinearity (Pearson's $r < 0.7$), resulting in six independent variables that represent important climatic, topographic, and human-driven factors: temperature seasonality (BIO4), annual precipitation (BIO12), precipitation seasonality (BIO15), elevation, distance to roads, and land use/land cover. The model performed well, with a training AUC of 0.86 and a test AUC of 0.86 (SD = 0.024), indicating an excellent ability to distinguish between suitable and unsuitable conditions. Temperature seasonality was the most critical predictor (40.3% contribution; 41.6% permutation importance), followed by elevation (21.7%) and distance to roads (11.6%). This highlights the combined influence of climatic stability, mid-altitude environments, and accessibility in selecting areas for apiculture investment. Response curves indicated higher suitability at elevations ranging from 1,000 to 1,600 m and in moderately accessible areas (~2–5 km from roads). Using the “balance” threshold (logistic = 0.092), 63% of the landscape was classified as suitable with low omission (1%), mainly in mid-altitude Miombo woodlands, riparian corridors, and semi-managed village landscapes. In contrast, steep highlands, arid plains, and degraded agricultural zones were generally less suitable for hive placement. These results offer a practical decision-support tool for zoning apiculture expansion and targeting management efforts amid increasing climatic and land-use pressures.

Keywords: Apiculture, Beekeeping, Environmental Drivers, Honeybees, Western Tanzania,

INTRODUCTION

Honeybees (*Apis mellifera*) are amongst the most essential pollinators worldwide (Hung *et al.*, 2018), playing a vital role in supporting agricultural productivity and sustaining natural ecosystem balance (Reilly *et al.*, 2024). Their contribution to biodiversity conservation and crop pollination services is well-documented (Thakur, 2012), with over 75% of the world's

top food crops relying, at least in part, on animal pollination, particularly insect pollinator species (Klein *et al.*, 2007; Potts *et al.*, 2010). In sub-Saharan Africa, particularly in rural communities, beekeeping offers a low-cost and sustainable livelihood strategy that simultaneously promotes forest stewardship and supports biodiversity conservation (Carroll & Kinsella, 2013). In Tanzania, particularly across the diverse Miombo woodlands

of western Tanzania, beekeeping offers ecological and socio-economic benefits (Kata, 2019; Ntalwila *et al.*, 2017) and supports the conservation of non-timber forest products (Munishi *et al.*, 2010).

Despite its promise potential, the sustainability of beekeeping in these areas is increasingly threatened by climate and environmental changes (Syampungani *et al.*, 2009). Variations in rainfall, fluctuations in temperature, and prolonged dry periods can reduce flower availability and nesting sites, thereby affecting honeybee foraging success and colony health (Rader *et al.*, 2012). Simultaneously, land conversion, deforestation (Lobora *et al.*, 2017), wildfires (Naftal *et al.*, 2022; Tarimo *et al.*, 2015), and overgrazing have altered natural habitats and disrupted the distribution of floral resources, causing ecological mismatches between bees and their foraging environments (Archer *et al.*, 2014; Munyuli, 2011). These pressures undermine colony health, honey production, and ecosystem resilience, thereby underscoring the need for targeted, evidence-based strategies to identify optimal sites for beekeeping and to mitigate emerging environmental threats. This is crucial to maintain a balance between the sustainable utilisation of ecosystem services and the conservation of habitats and their associated species (Sari, 2020).

While many studies in Tanzania have examined the socio-economic and market aspects of beekeeping, few have investigated how spatial and bioclimatic factors interact to determine habitat suitability for apiculture. Consequently, the vulnerability of beekeeping systems to climate variability and land-use change remains poorly understood, undermining the long-term productivity and sustainability of the sector (Newman *et al.*, 2021). Most existing analyses relay on descriptive or community-based data (Newman *et al.*, 2021), leaving a knowledge gap about the spatial dynamics of environmental suitability for beehive placement. To fill this gap, spatially explicit modelling approaches, such as the MaxEnt algorithm, offer robust

analytical tools for predicting species or habitat suitability based on presence-only occurrence data and environmental variables (Elith *et al.*, 2011; Phillips *et al.*, 2006).

Researchers have successfully applied MaxEnt to model the distributions of diverse taxa in relation to climatic and landscape variables. For example, Elith *et al.* (2011) provided a detailed explanation of the statistical underpinnings of MaxEnt and demonstrated how the algorithm performs in various ecological scenarios. While many studies have focused on vertebrates (Araújo & Guisan, 2006; Ndimuligo *et al.*, 2022) or plant species, the application of MaxEnt to pollinators, particularly for assessing beehive or honeybee habitat suitability, is growing. Klein *et al.* (2007) and Garibaldi *et al.* (2013) emphasised the critical role of suitable habitat plays in mediating responses to land use change and climatic shifts, discussing how landscape configuration influences foraging, reproduction, and colony health. Although these studies did not directly use MaxEnt for spatial prediction, they laid the foundation for incorporating detailed environmental and land cover predictors into distribution models (Garibaldi *et al.*, 2013; Klein *et al.*, 2007).

In the context of honeybee habitat modeling, studies have explored how climatic variables (e.g., temperature, precipitation) and land use patterns (e.g., forest cover, agricultural expansion) affect the availability of forage and nesting sites (Roubik, 2012). As such, the MaxEnt tool can be extended to assess beehive suitability by using occurrence points (or validated localisations of natural nests or traditional hives) and a suite of climatic and land use covariates. Combining these models with geospatial datasets on climate, elevation, land use, and human activities enables a better understanding of how ecological gradients influence honeybee distribution and apicultural potential across landscapes.

The Greater Mahale Ecosystem (GME), situated in western Tanzania along the eastern shores of Lake Tanganyika, offers a unique ecological

setting for studying these dynamics. It features diverse vegetation types, including Miombo woodlands, riparian forests, and semi-managed village lands, which provide essential forage and nesting resources for honeybees. However, increasing human pressures (Maijo *et al.*, 2020) and climate variability have exacerbated resource competition, heightened threats, and contributed to habitat loss and fragmentation, thereby making spatially informed beekeeping planning more urgent (Munishi & Kilungu, 2020; Naftal *et al.*, 2022). Understanding how environmental variables and human activities affect beekeeping potential is crucial for safeguarding livelihoods and ecosystem services in this ecologically sensitive region.

This study addresses the existing knowledge gap by conducting an integrative spatial assessment of the bioclimatic and environmental factors that determine beekeeping suitability and related environmental pressures within western Tanzania's GME. Using georeferenced beehive presence data alongside key climatic, topographic, and anthropogenic variables, the study applied a MaxEnt modelling framework to identify, map, and evaluate spatial patterns of apicultural potential and vulnerability. Specifically, the study aims to (i) determine the most influential bioclimatic and environmental variables affecting the spatial suitability of beekeeping habitats, (ii) produce a predictive habitat suitability map showing optimal and marginal zones for beehives, and (iii) analyse the combined effects of climatic variability and human-driven landscape changes on apicultural potential.

The core assumption of this research was that environmental stability, vegetation structure, and landscape accessibility together determine the spatial suitability for beekeeping. It was hypothesised that mid-altitude Miombo and riparian landscapes provide the most favorable ecological conditions for sustainable apiculture. The study improves predictive ability and spatial detail in apicultural suitability analysis by combining ecological modelling with participatory field data. The findings aim to

offer practical insights for evidence-based land-use planning, biodiversity conservation, and sustainable livelihood development by highlighting possible opportunistic zones for beekeeping establishment. Ultimately, this integrated approach contributes to the broader scientific discussion on pollinator ecology and promotes spatially informed apiculture strategies that can be utilised in the face of climatic and human pressures across western Tanzania's ecologically sensitive landscapes.

MATERIALS AND METHODS

Study Area Description

This study was conducted in the Greater Mahale Ecosystem (GME), situated in western Tanzania, along the eastern shores of Lake Tanganyika (Fig. 1). The ecosystem spans approximately 18,000 km² and encompasses Uvinza and Tanganyika Districts within the Kigoma and Katavi Regions, respectively. The GME serves as a vital conservation area, including the Mahale Mountains National Park, Tongwe East and West Forest Reserves, surrounding village lands, and multiple wildlife corridors that facilitate animal movement between protected zones. This distinctive ecological setup supports diverse biodiversity and extensive community-based land uses, making the GME a dynamic socioecological system of regional and national significance.

Vegetation within the GME is dominated by Miombo woodlands, primarily consisting of *Brachystegia*–*Julbernardia* formations, mixed with riverine forests, grasslands, and seasonal wetlands (Frost, 1996; White, 1983). These vegetation types provide vital forage and nesting resources for pollinators, including wild and managed honeybees (*Apis mellifera*), supporting a diverse range of floral and faunal species. The elevation of the area, ranging from approximately 770 m above sea level along Lake Tanganyika's shoreline to over 2,400 m in the Mahale Mountains (Carvalho *et al.*, 2022), creates pronounced microclimatic gradients that influence vegetation distribution, flowering

times, and pollinator activity (Dicks *et al.*, 2012). The GME has a tropical sub-humid climate, characterised by a single rainfall peak, with a wet season from November to April and a dry season from May to October (Collins & McGrew, 1988; Piel *et al.*, 2017). Annual rainfall averages range between 800 and 1,200 mm, and temperatures range from about 20°C in upland areas to 30°C in the lowlands (Ogawa *et*

al., 2006; Yoshikawa *et al.*, 2008). This climate variation creates diverse ecological niches that support different types of vegetation and boost honey production. Despite its ecological significance, the GME faces increasing human pressures that harm key habitats and threaten biodiversity (Maijo *et al.*, 2020; Naftal *et al.*, 2022; Tarimo *et al.*, 2015).

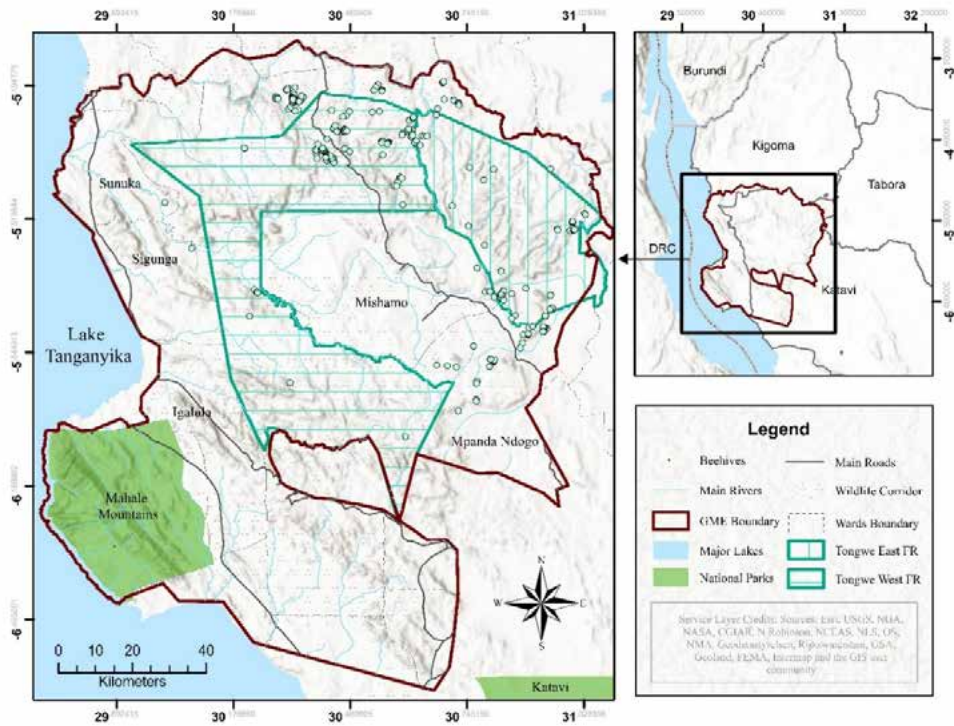


Figure 1: Study area location

Data Collection

Beehive Occurrence Data

Presence-only data on beehive locations was collected from community-managed apiaries across the GME. Data were collected using the Survey123 mobile application enhanced with built-in GPS capabilities and a customised digital survey form. Beehive locations were recorded as part of a broader chimpanzee survey that systematically documented chimpanzee nests, food tree species, anthropogenic threats,

and other ecological variables. Survey coverage followed a standardised design across 28 pre-established chimpanzee survey sites, with 16 randomly generated transects of 1 km length implemented within each site.

During field surveys, each beehive encountered was visually inspected to verify the presence of an active colony, and only occupied beehives were recorded as valid occurrence records. A total of 264 verified beehive locations were georeferenced. Since environmental predictors

were standardised to roughly 1 km spatial resolution, duplicate occurrences within the same raster grid cell were removed to maintain consistency with predictor resolution and reduce pseudo-replication caused by clustered observations. After retaining a single representative point per grid cell, 150 unique records of beehive presence remained for the final MaxEnt modelling and evaluation.

Environmental and Bioclimatic Variables

To characterise the environmental and climatic variables influencing beehive distribution across the GME, a total of 19 bioclimatic variables (BIO1–BIO19) were obtained from the WorldClim Version 2.1 database at a spatial resolution of 30 arc-seconds (~1 km) (Fick & Hijmans, 2017). These variables describe ecologically meaningful aspects of temperature and precipitation regimes, including seasonal trends, annual means, and extreme climatic conditions known to affect honeybee foraging behavior, colony development, and hive survival (Le Conte & Navajas, 2008; Rader *et al.*, 2012). In addition to climatic factors, several environmental variables were integrated to capture the combined effects of topography, land cover, and anthropogenic influences. Topographic variables (elevation, slope, and aspect) were derived from the Shuttle Radar Topography Mission (SRTM) Digital Elevation Model (DEM) at 30 m resolution (Farr *et al.*, 2007). Land use and land cover (LULC) data were sourced from the European Space Agency's Climate Change Initiative (ESA CCI), which provides globally consistent, annual land cover classifications (CCI, 2018).

Proximity variables such as distance to roads and rivers were calculated in ArcGIS 10.8 using data from OpenStreetMap (Haklay & Weber, 2008) and HydroSHEDS (Linke *et al.*, 2019), respectively. These layers were included to represent the accessibility and landscape connectivity that influence hive placement and foraging activity (Newman *et al.*, 2021). Disturbance-related variables were also incorporated to account for anthropogenic and

ecological stressors: wildfire frequency was derived from the MODIS Burned Area Product (Giglio *et al.*, 2018), while cattle density, used as a proxy for grazing pressure, was calculated using the kernel density function tool and GPS coordinates of all cattle signs that were encountered during the survey.

Data Preprocessing

All spatial datasets were harmonised to ensure consistency in spatial resolution and coordinate reference system before analysis. Each environmental and bioclimatic layer was resampled to a uniform grid size of 30 arc-seconds (~1 km) and projected to the World Geodetic System 1984 (WGS 84), Universal Transverse Mercator (UTM) Zone 36S coordinate reference system. The processed predictor variables were then clipped to the extent of the GME and converted into an ASCII raster format compatible with the MaxEnt modelling environment (Elith *et al.*, 2011; Phillips *et al.*, 2006). Multicollinearity was assessed using Pearson's correlation analysis on all continuous predictors to identify potential redundancy among variables. Variables with strong pairwise correlations ($r \geq 0.7$) were flagged, and one from each correlated pair was removed from further analysis (Dormann *et al.*, 2013; Feng *et al.*, 2019). This approach ensured that only independent and ecologically relevant predictors retained in the final model, thereby enhancing interpretability and minimising overfitting (Merow *et al.*, 2013). After the correlation screening process, a subset of six uncorrelated and ecologically relevant variables was retained for the final MaxEnt modelling. These selected predictors effectively represented the primary environmental variables and anthropogenic pressures influencing beehive distribution within the study landscape.

Modelling Approach

Habitat suitability for beehives within the GME was modelled using MaxEnt version 3.4.4 (Elith *et al.*, 2011; Phillips *et al.*, 2006). MaxEnt is a

machine-learning method designed explicitly for presence-only data and is widely recognised for its robustness, predictive accuracy, and ability to capture complex, nonlinear ecological relationships (Elith *et al.*, 2011; Merow *et al.*, 2013). The modelling framework incorporated both climatic and landscape-level predictors, refined to reflect the ecological requirements of honeybees and insights from prior modelling iterations. The analysis used 150 verified beehive presence records, with 70% ($n = 105$) randomly chosen for model training and 30% ($n = 45$) reserved for independent testing. To characterise the environmental background, 10,000 pseudo-absence points were randomly sampled throughout the study area. To improve model generalisability and reduce overfitting, a regularisation multiplier of 1.5 was applied, and feature classes were limited to linear, quadratic, product, and hinge types, an approach demonstrated to balance model complexity and interpretability (Warren & Seifert, 2011). A total of six ecologically relevant and statistically independent environmental variables were included: temperature seasonality (BIO4), annual precipitation (BIO12), precipitation seasonality (BIO15), elevation, distance to roads, and land use/land cover (LULC). After Pearson's correlation analysis ($r < 0.7$), these variables were selected to minimise redundancy and ensure independent representation of key ecological variables. Variables such as slope, aspect, and distance to rivers were excluded due to low contribution and high correlation during preliminary runs.

Model performance was evaluated using the Area Under the Receiver Operating Characteristic Curve (AUC), jackknife tests of variable importance, and omission rate analysis to assess discrimination ability and model reliability (Fielding & Bell, 1997). Response curves were generated to illustrate the relationships between environmental predictors and habitat suitability. At the same time, the importance of permutation was calculated to determine the relative contribution of each

variable under randomised conditions. The model output was generated in logistic format, with suitability values from 0 (unsuitable) to 1 (highly suitable). The continuous results were reclassified into four categories to enhance clarity of model outputs and facilitate interpretation: unsuitable (0–0.2), low (0.2–0.4), moderate (0.4–0.6), and high (0.6–1.0), following the classification criteria of Ansari and Ghoddousi (2018). This setup helped to map out the best zones for beekeeping and identify areas facing environmental constraints across the GME.

RESULTS

Model Performance

The MaxEnt model showed a strong ability to predict suitable habitats for beehives across the Greater Mahale Ecosystem. The training AUC was 0.86, and the test AUC was 0.86, with a standard deviation of 0.024, indicating excellent performance and applicability. The regularised training gain was 0.72, and the unregularised test gain was 0.93, indicating the model's effectiveness in distinguishing between suitable and unsuitable locations.

A range of omission thresholds was evaluated to guide interpretation. The minimum training presence threshold (logistic = 0.026) resulted in 0% omission for both training and test data, classifying 80.4% of the study area as suitable. The “balance” threshold (logistic = 0.092), which optimises training omission, predicted Area, and threshold value, achieved just 1% training omission and classified 63% of the Area as suitable. The equal training sensitivity and specificity threshold (logistic = 0.453) represented a more conservative estimate, classifying 20.3% of the area as suitable but with a higher test omission rate of 22.2%. These metrics affirm the model's reliability in identifying potential zones for beekeeping.

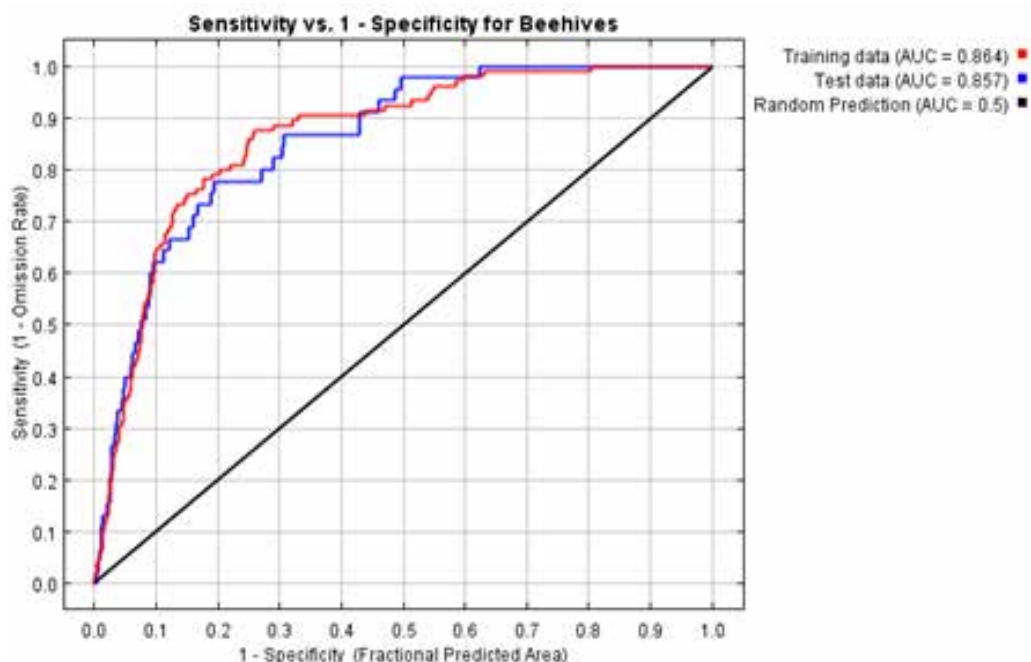


Figure 2. Maxent output showing model accuracy for current conditions.

Variables Importance and Contribution

Among the six environmental predictors used, temperature seasonality (bio04) was the most influential variable, contributing 40.3% to the final model and showing the highest permutation importance (41.6%). This underscores the critical role of stable seasonal temperatures in regulating flowering cycles and foraging efficiency for honeybee populations. Elevation was the second most crucial factor, with a 21.7% contribution and 15.3% permutation importance, reflecting its influence on microclimatic conditions, vegetation types, and nectar-producing flora. Distance to roads followed, contributing 11.6% and showing a 15.0% permutation importance, possibly indicating the influence of edge habitats, road-accessible floral corridors, and reduced disturbance in moderately remote zones. Land use/land cover contributed 12.4%, with a lower permutation importance of 1.8%, suggesting its role in defining general habitat

suitability, particularly for semi-natural landscapes such as agroforestry and miombo woodland mosaics. Annual precipitation (bio12) contributed 10.0% to the model, with a 5.7% permutation importance, confirming the importance of moisture availability in sustaining flowering and nectar flows. Lastly, precipitation seasonality (bio15) contributed 3.8%, but with a disproportionately high permutation importance of 20.5%, suggesting strong interactions with other climatic drivers.

A jackknife test of variable importance confirmed that bio04 had the highest training gain when used in isolation and caused the most significant drop in gain when excluded, emphasising its standalone predictive power. Elevation and distance to roads also contributed substantially, both independently and in interaction with other variables.

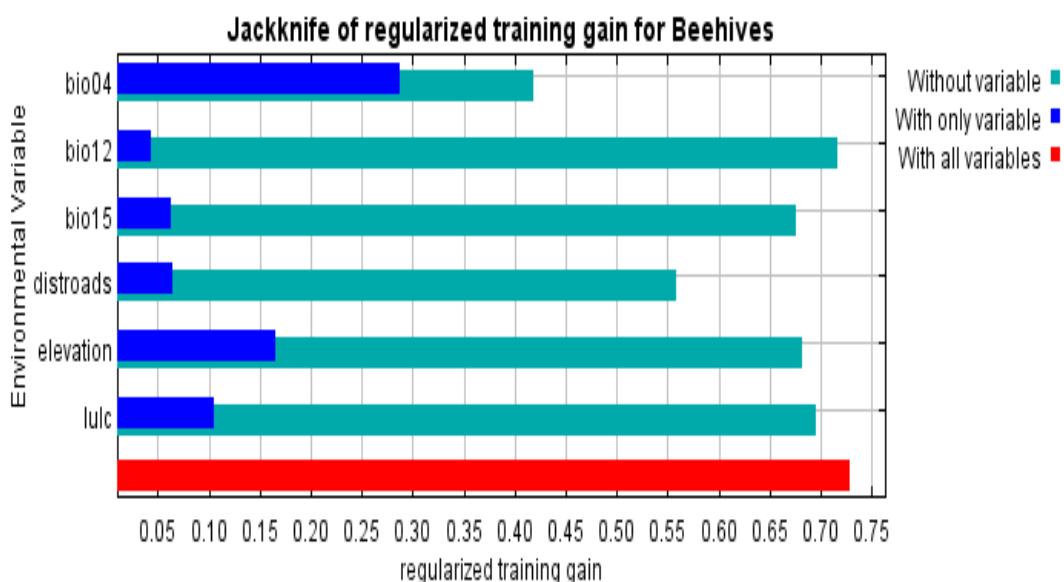


Figure 3. Jackknife estimates of variable importance in predicting the current potential suitable habitat distribution for Honeybees in GME.

Response Patterns and Habitat Suitability

Response curves indicated that suitability was highest in areas with moderate temperature seasonality, elevations between 1,000 and 1,600 m, and proximity to roads within a moderate buffer (~2–5 km). Suitability dropped sharply in regions with low precipitation, high elevation (>2,000 m), or extreme temperature

variability, likely due to ecological stress and limited floral resources. The final habitat suitability map highlighted concentrated zones of high suitability along riparian corridors, mid-altitude miombo woodlands, and semi-managed village lands. In contrast, lower suitability was predicted in steep montane slopes, degraded agricultural zones, and arid plains.

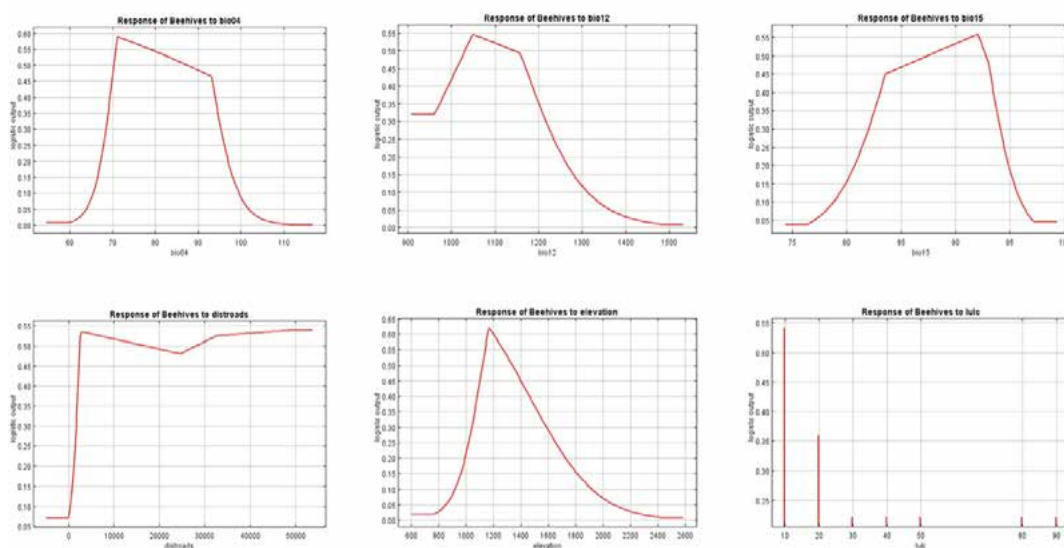


Figure 4. Maxent outputs of the most critical variables in predicting the current potential suitable habitat distribution for Honeybees in GME.

Current Predicted Potential Suitable Habitat Distribution

The final habitat suitability map shows varied patterns of beekeeping potential across the GME. High suitability areas are primarily located in mid-elevation zones, ranging from 1,100 to 1,600 meters, where moderate temperature fluctuations and steady rainfall combine to create ideal conditions for beehive placement. These areas include parts of Miombo woodlands, riparian corridors, and semi-managed village landscapes, which offer

diverse forage and natural nesting sites. Lower suitability was observed in steep highland slopes, arid lowland plains, and intensively cultivated agricultural fields, where ecological stressors or limited floral diversity reduce habitat quality. Regions labelled as barren or heavily disturbed also showed low logistic suitability scores. Overall, about 63% of the study area was classified as moderately to highly suitable under the “balance” threshold, offering practical guidance for prioritising areas for beekeeping, apiculture investment, and integrated land management strategies.

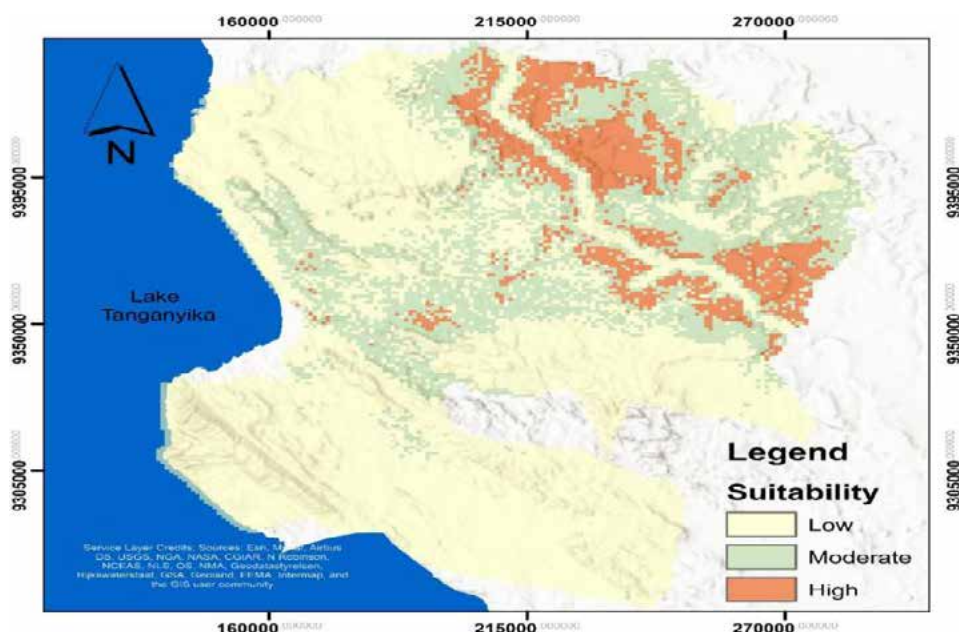


Figure 5: Current predicted potential suitable areas for Beekeeping activities GME

DISCUSSION

The MaxEnt model has provided a spatially explicit assessment of apicultural suitability across the Greater Mahale Ecosystem, indicating that beekeeping potential is jointly determined by climatic stability, topography, and access constraints. The model achieved intense discrimination ($AUC = 0.857$), which is generally interpreted as functional performance for presence-only niche models when the objective is landscape screening and prioritization rather than direct inference on abundance (Fielding & Bell, 1997; Phillips *et al.*, 2006). This level of performance also

suggests that the selected predictor set captured key environmental variables relevant to where beehives can be placed and maintained, while remaining consistent with core assumptions and interpretation practices of MaxEnt outputs (Elith *et al.*, 2011; Merow *et al.*, 2013).

Temperature seasonality emerged as the dominant predictor (40.3% contribution; 41.6% permutation importance), highlighting intra-annual thermal variability as a primary factor for determining apicultural suitability. The MaxEnt model's identification of temperature seasonality as the dominant predictor aligns with previous studies that

have highlighted the sensitivity of pollinators to climatic fluctuations. Phillips *et al.* (2006) demonstrated that presence-only models, such as MaxEnt, are especially effective in detecting key environmental variables, even when spatial data are limited, which is reflected in the high contribution of temperature seasonality in this study. In our case, the high permutation importance (41.6%) further emphasises that the model's predictive capability is highly dependent on the accurate representation of these variable. This suggests that even small changes in temperature stability over the year could lead to significant shifts in habitat suitability for bees.

Moreover, the dominance of temperature seasonality in the model may imply that seasonal extremes, both in terms of high variability during transitional periods and extreme differences between seasonal minima and maxima, act as stressors that limit bee distribution. Bees have specific physiological and behavioral constraints; for example, they require moderate temperature ranges to maintain hive activities and brood rearing (Elith *et al.*, 2011). High temperature fluctuations can lead to energetic inefficiencies and disrupt foraging cycles, which may ultimately affect the population of both wild and managed bees in GME.

Thermal variability influences colony performance and health by shaping flight activity windows and activity time, brood development rates, and flowering/nectar phenology, and it may interact with stress and disease processes that reduce productivity and survival in honeybee colonies (Le Conte & Navajas, 2008). The strong role of seasonality is also consistent with the value of standardised, spatially continuous climate surfaces for suitability modelling in data-limited landscapes (Fick & Hijmans, 2017). Collectively, these findings imply that zones characterised by comparatively stable seasonal temperature patterns are more likely to support sustained foraging and colony persistence, and therefore represent priority areas for apiculture

establishment, expansion, and extension support. This finding is consistent with other studies that have highlighted the influence of temperature on honeybee colony activities and their interactions with forage (Mengistu *et al.*, 2025).

Elevation was the second most influential predictor (21.7% contribution), with suitability concentrated in mid-altitude landscapes. Elevation is a well-established environmental variable that influences several microclimatic factors and ecosystem characteristics. In the GME, mid-altitude ranges generally provide moderate temperatures, balanced moisture levels, and diverse flowering plant communities, collectively creating favorable conditions for bee foraging and colony maintenance. These conditions may contrast with those at lower or higher elevations, where extreme temperatures or reduced vegetation diversity could limit apicultural potential (Araújo & Guisan, 2006).

The role of elevation as a critical predictor aligns with earlier studies that have documented its influence on species distribution. In many cases, mid-altitude areas are associated with a combination of favourable microclimates and heterogeneous landscapes that provide both shelter and abundant forage, a combination that is particularly beneficial for bee populations (Elith *et al.*, 2011; Phillips *et al.*, 2006). The moderate thermal conditions at mid-altitudes could mitigate the adverse effects of extreme temperatures observed at lower and higher elevations. This climatic moderation is crucial for maintaining colony health, as bees rely on stable temperature ranges for brood development and foraging activities (Klein *et al.*, 2007).

Moreover, the concentration of suitability in mid-altitude landscapes may also be influenced by land cover and human land use patterns. In many regions, mid-elevations are less intensively altered by agricultural expansion and urban development when compared to lowlands. The resulting mosaic of natural vegetation and semi-natural habitats can

sustain a rich diversity of nectar and pollen sources that are crucial for bees. This is particularly important in the GME, where land-use pressures are significant. Thus, elevation acts indirectly by mediating the availability and quality of resources needed for apiculture (Araújo & Guisan, 2006).

It is also possible that mid-altitude areas serve as a climatic refugee in the face of increasing temperature variability. These zones often experience less pronounced extremes than lower and higher regions, thereby providing stable environmental conditions amidst broader climatic fluctuations. The mid-altitude focus could therefore reflect an ecological balance, where the physiological tolerances of bees are optimally met while the landscape supports sufficient floral resources and nesting sites (Elith *et al.*, 2011).

Elevation frequently behaves as a composite proxy for multiple coupled gradients, including temperature, moisture balance, vegetation structure, and soil properties, so its prominence is expected where direct predictors of floral phenology and nesting substrate availability are not explicitly mapped (Elith *et al.*, 2011; Merow *et al.*, 2013). In the GME context, mid-elevation zones have been identified as ecologically plausible “sweet spots,” likely retaining Miombo woodland structure and diverse forage resources while avoiding the cooler, steeper highlands and the hotter, drier lowlands. This elevation signal, therefore, enhances the ecological interpretability of the area and its suitability, thereby supporting management recommendations that prioritise mid-altitude woodland and riparian mosaics for sustainable beehive placement, as well as highlighting its conservation implications.

Anthropogenic gradients added a precise socioecological dimension to the model. Distance to roads contributed 11.6% (15.0% permutation importance), indicating that realised suitability reflects both logistical feasibility for installation, routine management, and product transport, as well as the biophysical

quality of the habitat. This pattern is typical in presence-only models of managed activities because human decision-making constrains the use of suitable sites (Merow *et al.*, 2013). At the same time, proximity to roads can covary with disturbance and habitat conversion, meaning that accessible areas may also carry a higher risk of resource degradation. The influence of land use/land cover further supports this coupled human–environment interpretation, as less-converted habitats are more likely to maintain vegetation structure, forage continuity, and colony retention. Consistent with this logic, the model indicates that suitability is not purely ecological; the relative integrity of the habitat and the practicality of access jointly condition where beekeeping is most viable (Elith *et al.*, 2011).

Spatially, the predicted suitability surface was heterogeneous, with higher suitability clustering in Miombo blocks, riparian corridors, and semi-managed village lands. Under the balanced threshold scenario, approximately 63% of the study area was categorised as moderately to highly suitable, indicating a substantial opportunity for targeted apiculture expansion in the GME, provided that environmental pressures are addressed. Landscape heterogeneity has been shown to influence pollinator success. In the context of beekeeping, it positively ensures that honeybees maintain access to foraging resources necessary for colony health and productivity (Cavigliasso *et al.*, 2022; Parreño *et al.*, 2024).

For interpretability, the four-class probability scheme (0–0.2 unsuitable; 0.2–0.4 low; 0.4–0.6 moderate; 0.6–1.0 high) is widely applied in MaxEnt-based suitability mapping, providing a practical framework for zoning and extension messaging (Ansari & Ghoddousi, 2018). Operationally, the “moderate-to-high” zones identified by the model represent candidate areas in the GME that can be utilised for beekeeping, including the development of hive-siting guidelines, community training, and monitoring, thereby providing an overall socioecological benefit for the communities in

the area. In contrast, low-suitability zones (e.g., heavily cultivated areas and steep highlands) can be treated as lower-priority areas, providing an opportunity to improve habitat conservation by requiring restoration and risk mitigation. The combined effect of using environmental, human-driven, and ecological characteristics in the model has successfully highlighted areas for potential beekeeping in the GME, thereby enabling strategic planning for utilising the area for sustainable conservation. This highlights the importance of using multiple criteria in modelling ecological scenarios to aid conservation planning as well as be able to predict future scenarios in the face of climate change (Naqibzadeh *et al.*, 2021)

Several limitations warrant attention to guide future refinement. Presence-only modelling is sensitive to sampling bias, and participatory mapping can cluster records near settlements and roads, potentially inflating the apparent importance of access variables (Merow *et al.*, 2013; Phillips *et al.*, 2006). Incorporating explicit bias correction (e.g., bias files or target-group backgrounds), additional spatial thinning, and independent validation surveys would strengthen inference and transferability. Predictor collinearity can also influence model stability and interpretation, making transparent correlation screening and sensitivity checks essential (Dormann *et al.*, 2013).

Finally, model complexity and parameterisation affect the risk of overfitting and spatial generalisation, so reporting and justifying key settings (feature classes, regularisation) aligns with best practices for MaxEnt applications (Warren & Seifert, 2011) and broader reproducibility guidance for niche modelling studies (Feng *et al.*, 2019). Despite these constraints, the model provides a defensible decision-support layer, guiding potential beekeeping investments toward zones where climatic suitability, habitat condition, and practical access converge, while highlighting areas where land conversion, such as intensive agricultural areas, may undermine apicultural potential over time.

CONCLUSION AND RECOMMENDATIONS

The MaxEnt modelling results provide a clear, spatially explicit foundation for guiding sustainable beekeeping in the Greater Mahale Ecosystem. Model performance was strong (training AUC = 0.86; test AUC = 0.86; SD = 0.024), and the operational “balance” threshold (logistic = 0.092) classified approximately 63% of the landscape as suitable while maintaining a low omission rate (1%). These outputs therefore offer a practical basis for zoning, targeting investments, and supporting evidence-based hive placement for beekeeping purposes. Suitability was influenced most strongly by temperature seasonality, followed by elevation and distance to roads, indicating that climatic stability, topographic variation, and manageable access jointly shape where beekeeping is most viable. In the GME, high-suitability areas were concentrated in mid-elevation Miombo woodlands, riparian corridors, and semi-managed village landscapes. In contrast, steep highlands, arid lowlands, and intensively cultivated or disturbed areas were consistently less suitable for beekeeping establishment.

Based on these findings, implementation should prioritise hive placement within the mapped high-suitability zones, particularly mid-elevation Miombo–riparian mosaics, and apply the “balance” threshold classification to reduce both over- and under-allocation of sites during planning. Apiaries should be located in moderately accessible areas (approximately 2–5 km from roads) to maintain management feasibility while minimising exposure to disturbance. The suitability maps should also be integrated into community forest management and village land-use planning processes to safeguard priority beekeeping habitats and minimise land-use conflicts. Finally, extension services should adopt the four-class suitability framework (unsuitable to high) to support clear communication, phased implementation, and consistent decision-making among stakeholders.

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SUB-THEME: CLIMATE CHANGE ADAPTATION AND RESILIENCE

THE IMPLICATIONS OF CLIMATE CHANGE ON WILDLIFE RESOURCES MANAGEMENT IN TANZANIA'S SOUTHERN PROTECTED AREAS

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ABSTRACT

Protected areas (PAs) are critical for biodiversity conservation, ecological connectivity, and the provision of nature-based tourism benefits to local communities. However, climate change increasingly threatens their ecological integrity, particularly in wildlife-rich landscapes in the Tanzania's Southern PAs. Despite their significant impact, there is limited information on how climate change and variability affect the management of natural resources within these PAs. This paper investigates the impacts of climate change on wildlife resources management within Tanzania's Southern PAs, focusing on the Ruaha–Rungwa landscape. The study employed mixed-methods approach involving collection of data from household surveys, interviews, focus group discussions, climatic trends and remote sensing for measuring land use/land cover (LULC) changes. A random sampling technique was employed to conduct household surveys, with sample size of 234 households. A purposive sampling technique was adopted to select 110 key informants and 44 focus group participants. The XLSTAT and ERDAS tools were employed to analyse meteorological and remote sensing data, respectively, to assess the spatial and temporal impacts of climate change on land use and land cover (LULC) in the landscape over 30 years from 1987 - 2017. The results indicated variations in temperature and rainfall patterns over the period 30 years, leading to recurrent droughts, extended dry spells, and reduced water availability for wildlife. These climatic stresses, compounded by anthropogenic activities such as upstream water abstraction, deforestation in catchment areas, and livestock encroachment, have disrupted landscape connectivity, degraded habitats, and intensified human–wildlife conflict. Wildlife populations face additional pressures from habitat transformation, invasive species, and increased disease outbreaks. The paper concludes that mitigating climate change impacts on wildlife resources management in southern PAs requires strengthening water resource management, enhancing habitat restoration, restoring ecological corridors, enhancing adaptive capacity in PAs governance and aligning conservation planning with climate resilience strategies. These measures are essential for safeguarding wildlife populations and sustaining both ecological integrity and community livelihoods in the face of a changing climate.

Keywords: Climate change, Ruaha–Rungwa landscape, Southern protected areas, wildlife resources, wildlife conservation,

INTRODUCTION

Wildlife resources are vital for maintaining healthy, balanced ecosystems and supporting human survival and development (Díaz *et al.*, 2019). In Africa, biodiversity is a cornerstone

of natural capital, supporting local economies, tourism and cultural practices (Hoffman and Vogel, 2008). The presence of diverse species and extensive ecosystems not only contributes to environmental health but also offers significant opportunities for sustainable

livelihoods, particularly in rural areas (Kilungu *et al.*, 2016). In Tanzania, wildlife resources, especially those found in protected areas like the Ruaha–Rungwa landscape, are key drivers of conservation efforts and the national economy (Chilagane *et al.*, 2020; Hariohay, 2019). These areas not only support endangered species but also attract tourists, which generates revenue and creates jobs for local communities (Komba *et al.*, 2021; Mariki, 2015). The unique flora and fauna of Africa and Tanzania hold immense ecological, cultural, and economic value, reinforcing the need for effective wildlife management and conservation strategies (FAO, 2012; IPCC, 2022).

Despite the importance of wildlife resources, climate change poses significant challenges to their survival (IPCC, 2022). Climate impacts such as rising temperatures, altered precipitation patterns and extreme weather events like droughts and wildfires, are already disrupting wildlife habitats and migration patterns (IPCC, 2014; 2022). Habitat loss, fragmentation and shifts in species distributions further exacerbate these challenges, placing additional strain on wildlife management systems (Prins *et al.*, 2025). Moreover, the combined effects of climate change and other stressors, such as anthropogenic activities threaten biodiversity and the ecosystem services that depend on it (Brondizio *et al.*, 2021). Consequently, wildlife management strategies are increasingly shifting toward ecosystem-based approaches that can adapt to the uncertainty and variability induced by climate change (Caro and Davenport, 2016; Kangalawe, 2016; Kilungu *et al.*, 2016). Tanzania, a globally renowned for its rich endowment of natural resources and extensive network of protected areas, has increasingly faced the adverse impacts of climate change on wildlife conservation and natural resource management (Kangalawe, 2016; Kideghesho *et al.*, 2013; Mushi and Kangalawe, 2020). Though several studies have examined the impacts of climate change on wildlife resource management in protected areas across Africa, including Tanzania, few have specifically focused on the southern regions such as

the Ruaha–Rungwa landscape. Despite the significant ecological and socio-economic implications, there is limited empirical evidence on how climate change affects the management of natural resources within these protected areas. The growing concern over climate-induced changes in rainfall patterns, temperature regimes and vegetation cover has heightened the need for context-specific research on wildlife management under changing climatic conditions. Consequently, it is crucial to understand the impacts of climate change on management of wildlife resources in this landscape for developing evidence-based strategies that enhance resilience, sustain biodiversity, and ensure effective ecosystem management. This paper investigates the implications of climate change on wildlife resources management in the Tanzania’s Southern protected areas particularly on the Ruaha–Rungwa landscape.

This paper has both empirical and practical significance in the context of wildlife resource management under changing climatic conditions in Tanzania’s Southern protected areas. Empirically, it provides evidence on the trends and patterns of climate change over the past three decades (1987–2017) and their impacts on ecological and socio-ecological systems in the Ruaha–Rungwa ecosystem. Specifically, the findings highlight significant change in land use and land cover types, leading to the destruction and fragmentation of critical wildlife habitats. The study further documents the reduction in river water levels, resulting in inadequate water availability for both wildlife and ecosystems, and the invasion and domination of invasive plant species that threaten native biodiversity. In addition, it captures the eruption of wildlife diseases and a growing incidence of human–wildlife conflicts, all of which undermine conservation objectives in the landscape. Practically, the study offers insights that can guide adaptive management strategies, inform policy formulation, and strengthen conservation interventions. Through integrating these findings, conservation stakeholders can enhance the resilience of

wildlife populations, maintain ecological functionality, and promote sustainable ecosystem management in the face of ongoing and projected climate challenges.

Conceptual Framework

The conceptual framework (Figure 1) for this paper shows that climate and non-climate stressors interact with immediate drivers (population growth, weak enforcement, poor infrastructure) to cause ecological change. This ecological change leads to multiple impacts on wildlife resources, which can be mitigated through targeted adaptation interventions, ultimately ensuring improved ecological integrity, enhanced community resilience, and sustainable tourism development.

Climate change, manifested through rising temperatures, prolonged dry spells, floods, and recurrent droughts, has become a major driver of ecological change and wildlife vulnerability (FAO, 2012; IPCC, 2014; IPCC, 2022). These climatic stressors alter water availability, disrupt habitat integrity, and intensify extreme events that threaten the survival of key species. For instance, prolonged dry spells and declining river levels in Southern Tanzania, particularly within the Ruaha–Rungwa landscape, have caused widespread water scarcity, destruction of habitats, increased human–wildlife conflicts, and loss of connectivity between protected areas (Kaswamila *et al.*, 2010; Mayengo *et al.*, 2017). Non-climatic stressors further compound these risks. Human-induced habitat modifications such as deforestation, agricultural expansion, poaching, and unsustainable resource use have been reported to degrade nearly half of primate habitats globally (IUCN, 2008; Mukeka *et al.*, 2018; Kupika *et al.*, 2019). These drivers, coupled with weak policy

enforcement, inadequate infrastructure, and limited adaptation strategies, leave ecosystems increasingly vulnerable to climate-induced stressors (Shemsanga *et al.*, 2010; Lohay *et al.*, 2020).

The combined effects of climate and non-climate stressors on wildlife resources include inadequate water availability due to reduced river levels, destruction and fragmentation of habitats and the invasion and domination of non-native species in rangelands. These stressors also contribute to outbreaks of diseases leading to wildlife mortality, escalate human–wildlife conflicts and disrupt connectivity between protected areas, particularly in regions like the Ruaha–Rungwa corridor (Zeller *et al.*, 2012; Simpkins *et al.*, 2018). These impacts not only threaten ecological integrity but also undermine community livelihoods and tourism. Droughts reduce agricultural productivity and food security, while wildlife distribution shifts and habitat loss lower the quality of tourism experiences, resulting in reduced revenues and shortened tourist seasons (Mushi & Kangalawe, 2020; UNWTO, 2015). Inadequate alignment between climate adaptation strategies and conservation/tourism plans further exacerbates these challenges (USAID, 2015; Kupika *et al.*, 2018).

To address these issues, adaptation interventions are necessary. These include water resource management, habitat restoration and connectivity, control of invasive species and wildlife diseases, community-based adaptation strategies, and investment in green and grey infrastructure. Adoption of climate-smart conservation plans will reduce vulnerability, safeguard ecological integrity, and promote sustainable wildlife-based tourism (Lohay *et al.*, 2020; Jones *et al.*, 2012).

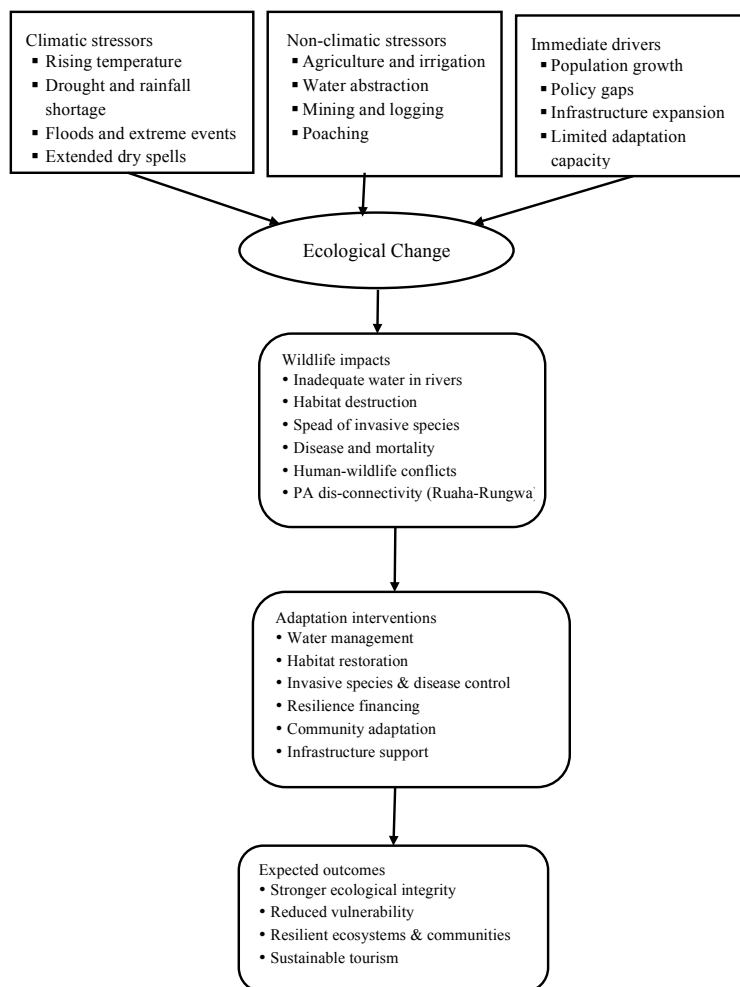


Figure 21: Conceptual Framework Indication the Impact of Climate Change

MATERIALS AND METHODS

This study was conducted in Ruaha–Rungwa ecosystem, comprised of two protected areas namely, Ruaha National Park (RUNAPA) and Rungwa Game Reserves (RGR) (Fig. 2). RUNAPA is located between 7°00' S – 8°00' S and 33000' E – 36000' E and RGR is located between 6°40'0" S – 7°03'0" S and 34000'0" E – 35000'0" E (Figure 1). RUNAPA is the second largest National Park in the country, lies to the west of the southern highlands of Tanzania and covers an area of 20,226 km² (RUNAPA, 2019). RGR is located in two regions of Singida and Mbeya, covers about 17,320 km² (URT, 2011; WD, 2015) and is a product of unification of Rungwa, Kizigo, and Muhesi Game Reserves. The study area was

purposely selected due to its being the largest wildlife area in Tanzania that offers various tourism activities (WB, 2015), its diversity of wildlife (USAID, 2015; UNWTO, 2015), and its growing number of ecological monitoring, enforcement and infrastructure development program (Coppolillo, 2004).

This study employed a mixed research design involving a combination of quantitative and qualitative methods and approaches. Quantitative methods involved conducting household surveys and spatial analysis techniques to assess the impact of climate change on protected ecosystems. Qualitative design utilized exploratory methods (interviews, focus group discussions and field observation) to study human beliefs, opinions, and attitude

regarding the implications of climate change on wildlife resources. The application of mixed (quantitative and qualitative) research methods is justified by its superiority in ensuring data

quality and reliability as advocated by various researchers (Bergman, 2008; Maxwell, 2010; Morgan, 2017; Creswell, 2018).

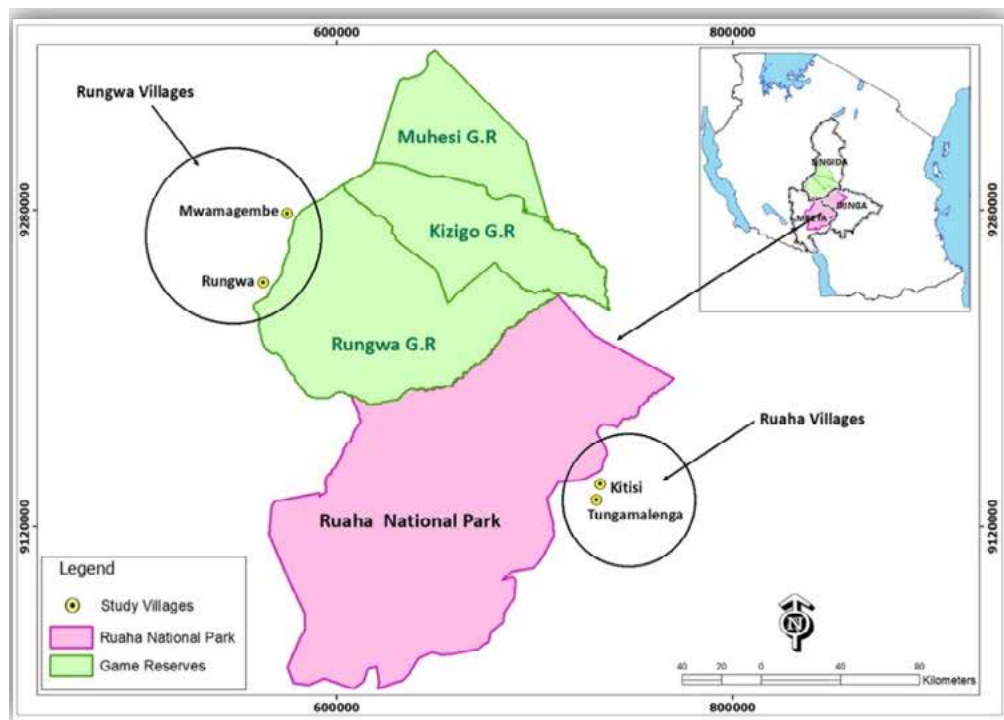


Figure 22: Map showing the location of Ruaha National Park and Rungwa Game Reserve
Source: GIS Lab – Ruaha National Park, (2019).

The study population was comprised of households in four villages in the study area surrounding Ruaha National Park and Rungwa G.R. namely Tungamalenga, Kitisi, Rungwa and Mwamagembe. According to the statistics given by the village executive officers in the study villages, there were a total of 2340 households in the four villages from which 10% were selected for inclusion in this paper. Other target populations included PAs staffs, Government Officers from the office of District Executive Directors in the two districts, Private sectors representatives, Civil Society Organizations (CSOs) representatives, including Community Based Organization (CBOs), village executive officers, village chairpersons, ward executive officers and representatives of significant groups such as influential people, elders, head of pastoralists / farmers, traditional leaders and women representatives in study villages.

The paper aimed at selecting a minimum of 10% of total households in each sampled villages as recommended by various scholars (Nardi, 2003; Lyimo, 2005; Singh & Masuku, 2011; Kothari, 2014). Thus, 234 heads of households, 110 key informants and 44 participants of FGDs were selected to represent the study population. The focus groups consisted of influential people, elders, pastoralists, farmers, traditional leaders, women, village game scouts, and natural resource committee representatives. Purposive sampling method was used to select four villages namely Tungamalenga, Kitisi, Rungwa and Mwamagembe: two villages adjacent to Ruaha National Park, and two adjacent to Rungwa Game Reserves. Selection of study villages considered accessibility to the park in terms of distance, magnitude of interdependence between local community and protected areas, natural resources availability in the protected areas, and nature of human - protected areas

interactions (e.g. river catchments, tourism, encroachments, wildfires, poaching). Simple random sampling was employed to select a sample size of 234 households for administering questionnaires. The use of random sampling technique ensured equal chance of inclusion of every head of household in the study (Long, 2007). Heads of households were sampled from village household registers in each study village. Further, purposive sampling technique was employed to select 110 key informants for in-depth interviews and 44 participants of FGDs in the study area. Selection of the participants considered age, sex, marital status, occupation and other socio-demographic variables of the respondents.

Primary data were collected by conducting 234 household surveys using questionnaire consisting of open and close-ended questions. The questionnaires comprised of questions that assessed the respondents' perceptions on climate trends and patterns, associated consequences to wildlife resources in the study area. Key informant interviews were conducted to 110 key informants, guided by a checklist of questions in the Interview guide aimed at obtaining detailed information regarding the impacts of climate change on management of wildlife resources in the study area. The key informants comprised of 16 PAs staffs (4 ecologists, 2 veterinary officers, 4 protection officers, 2 GIS officer and 4 tourism officers from both RUNAPA and Rungwa Game Reserves), 24 Government Officers from office of executive directors in the two districts, 27 Private sectors representatives (tour operators, airline operators, hunting companies, tour guides, lodges and camp managers), 13 CSOs representatives including CBOs, 4 village executives officers, 4 Village chair persons, 2 ward executive officers and 20 representatives of important groups in study villagers. One (1) focus group discussion was conducted in each study village, and was guided by a checklist of questions in the FGD guide for the purpose of obtaining essential information for the analysis of climatic trends impacts on wildlife management. The relevant participants

were objectively selected based on their roles/positions in the community, knowledge, experience and capacity to provide the required information in this paper.

The secondary data including climatic data (temperature and precipitation) for the study area over the past 30 years (1987-2017) were collected from Tanzania Meteorological Agency (TMA). These data were used to assess the trends and patterns of climate variability and changes. The data obtained generated information on patterns and trends of climate variability and change, and examine its impacts on wildlife ecology and livelihoods of adjacent communities. Further, remote sensing data were collected from satellite imagery of years 1987, 1997, 2007, and 2016 acquired from Earth Explorer of the United States Geological Survey (USGS) through <https://earthexplorer.usgs.gov/>. The extracted satellite images were used to determine land use/land cover (LULC) changes over 30 over years from 1987 to 2016. Direct field observation involved transects walk approach for ground-truthing in order to validate land use/cover findings generated through remote sensing. One transects walk of about one kilometre was conducted in each village, and in selected segments within the focused protected areas in order to determine the trend of land-use/cover changes and its linkage to climate change impacts on natural resources management.

This study collected quantitative and qualitative data. Quantitative data from household surveys were processed and analysed using the Statistical Package for Social Science (SPSS) software and Microsoft Excel. Meteorological data from TMA was subjected to XLSTAT statistical analysis tool to generate descriptive information related to climate trends and patterns. Satellite imagery was pre-processed in ERDAS Imagine 2022 by mosaicking multiple paths and rows, followed by clipping to the study area. The images underwent DOS-based atmospheric correction to retrieve land surface reflectance and were re-projected to WGS 84 (UTM Zone 36S, EPSG:32636) for accurate

spatial alignment. Finally, semi-supervised classification in ERDAS was applied to identify and map land use and land cover types (Tempa & Aryal, 2022). The qualitative data were analysed using content analysis, a systematic approach to interpreting textual information. The process started with familiarizing with the data through thorough reading of the transcripts from interviews and FGDs. Then, coding was applied, where specific segments of the text were labelled with codes that represented key ideas. These codes were then grouped into broader themes that captured the essence of the data. Content analysis helped to uncover underlying meanings, relationships, and insights that are not immediately apparent, providing a comprehensive understanding of the qualitative data. The findings from both quantitative and qualitative data were analysed using descriptive and inferential statistics

and presented through figures, tables and explanatory text.

RESULTS AND DISCUSSION

Trends and Patterns of Climate Change Over the Past 30 Years

Trends of Annual Rainfall from the 1980s-2018

This paper found that there was an insignificant difference in annual rainfall over 30 years from 1989 to 2018 across the study area. However, according to the analysis of meteorological data, there was extreme rainfall during 2009, and from 2015-2018 (Figure 3). Trends of extreme annual rainfall in the Rungwa Game Reserve were observed from 2009 (LM, $F_{(1, 28)} = 3.516$, $P = 0.06$).

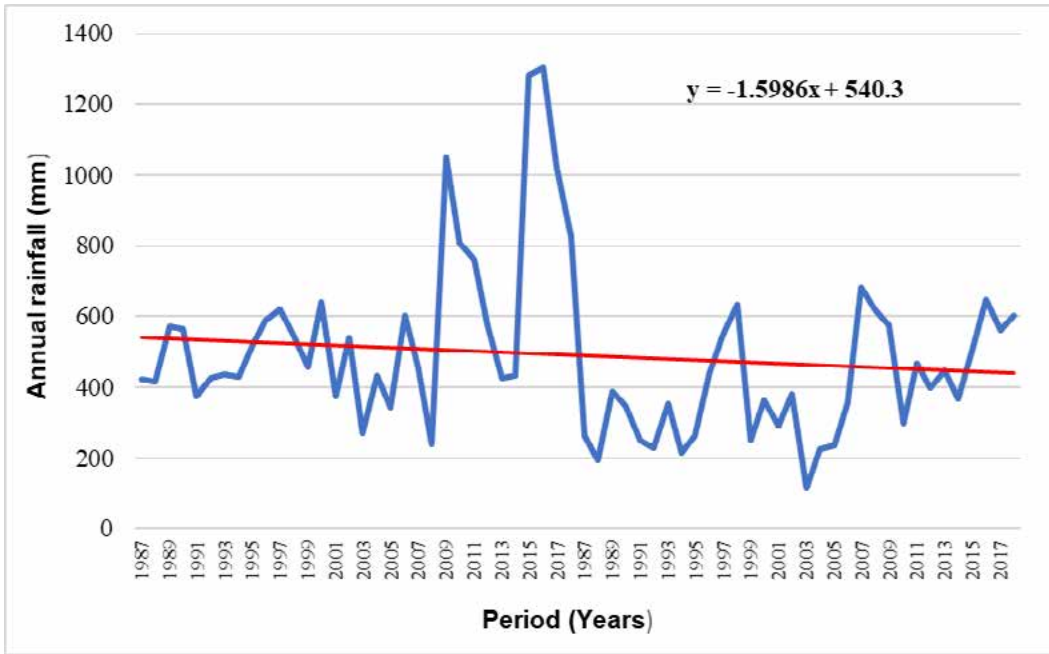


Figure 23: Observed annual rainfall trends in Rungwa Game Reserve, 1980s-2018.

Source: Computed from TMA data, (2021).

Similarly, analysis of meteorological data indicated an insignificant difference in annual rainfall over 30 years from 1989 to 2018 in the Ruaha National Park (LM, $F_{(1, 28)} = 1.148$, $P = 0.293$). According to the results, Ruaha

National Park experienced a similar pattern to the Rungwa Game Reserve. As with the Rungwa Game Reserve, Ruaha National Park also experienced extreme high rainfall during 2009, and 2015-2018 (Figure 4).

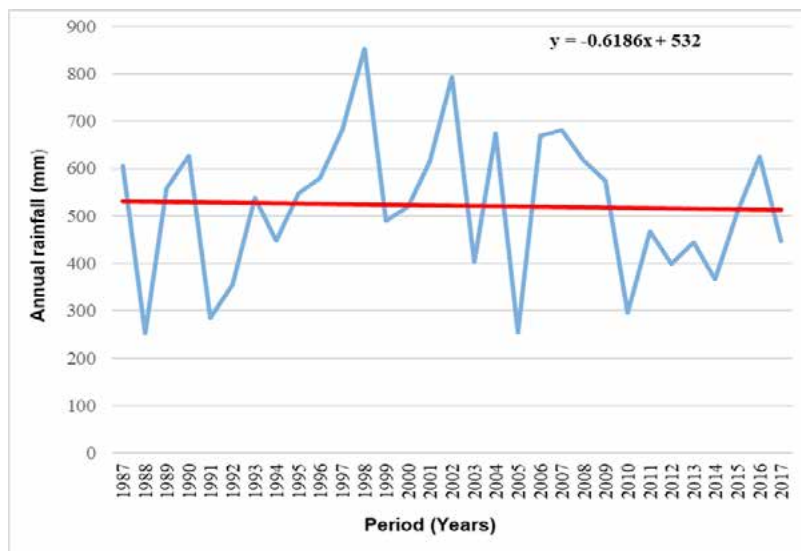


Figure 24: Observed annual rainfall trends in Ruaha National Park from the 1980s-2018.

Source: Field Data Collection, (2021).

Trends of annual temperature from the 1980s-2018

This paper observed that the overall annual temperature trend over the past 30 years from 1989 to 2018 increased but was not statistically significant (LM, $F_{(1, 28)} = 1.881$, $P = 0.181$) in Rungwa Game Reserve (Figure 5). The analysis results indicated that there was also

an insignificant increase of annual temperature from 1989 to 1998 (LM, $F_{(1, 8)} = 0.686$, $P = 0.401$) and significant decrease of annual temperature from 1999 to 2008 (LM, $F_{(1, 8)} = 6.692$, $P = 0.031$). It was also observed that there was an insignificant increase in annual temperature (LM, $F_{(1, 8)} = 2.286$, $P = 0.169$) from 2009 to 2018 (Figure 5).

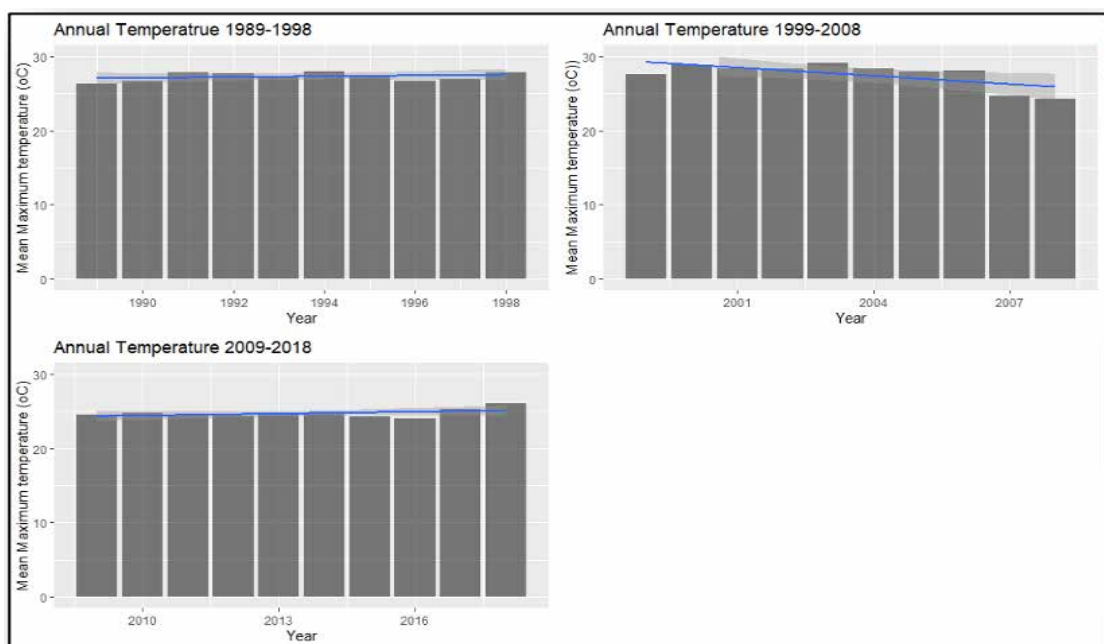


Figure 25: Annual temperature trends in Rungwa Game Reserve from the 1980s-2018.

Source: Field Data Collection, (2021).

Unlike Rungwa Game Reserve, the overall annual temperature trend in Ruaha National Park over the past 30 years from 1989 to 2018 indicated decreasing trend but was not statistically significant (LM, $F_{(1, 28)} = 2.634$, $P = 0.109$). However, based on the decadal patterns, the analysis indicated an insignificant increase of annual temperature from 1989 to 1998 (LM, $F_{(1, 8)} = 0.664$, $P = 0.408$) and

a significant decrease in annual temperature from 1999 to 2008 (LM, $F_{(1, 8)} = 8.86$, $P = 0.02$). Comparatively, there was a strong significant increase in annual temperature from 2009 to 2018 (LM, $F_{(1, 8)} = 34.64$, $P = 0.0004$). Generally, this implies that over the past two decades temperatures have been significantly variable in the Ruaha National Park (Figure 6).

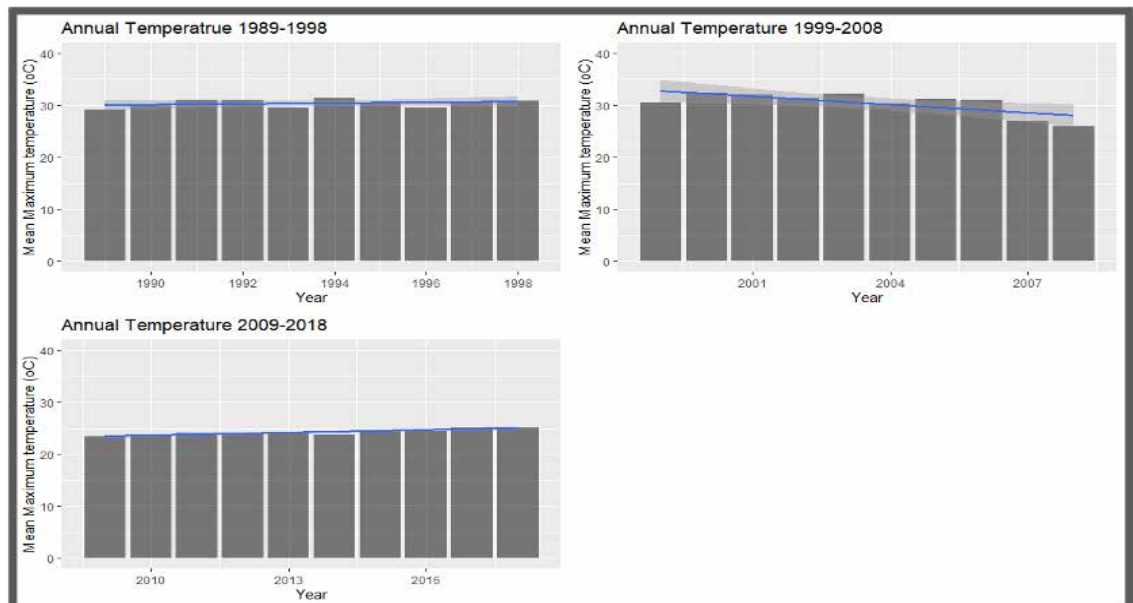


Figure 26: Annual temperature trends in Ruaha National Park from the 1980s-2018.

Source: Field Data Collection, (2021).

Impacts of the Changing Climate on Wildlife Resources Management

Inadequate water due to a decrease in water levels in rivers

This study revealed that climate change and variability have significantly affected water resources essential for sustaining wildlife in the study area. Key informants reported a noticeable decline in river water levels within Ruaha National Park, particularly during the dry season. They further emphasized that several water sources completely dried up during this period, a situation corroborated by multi-temporal satellite images (Figure 7), which illustrate a pronounced reduction in water flow in the Great Ruaha River during the dry season.

Figure 7 illustrates the perceived decline in river water levels attributed to climate change across the study villages. The findings show that all surveyed villages reported experiencing a severe reduction in river water levels, largely associated with climate-related factors such as delayed rainfall and prolonged dry spells. In the Ruaha-Rungwa landscape, the decline and drying of rivers were primarily linked to drought conditions resulting from delayed rains. Statistical analysis further indicated a significant variation in respondents' explanations for the observed decline in river water levels ($X^2 = 303.9$, $df = 5$, $p < 0.01$). Overall, approximately 95% of respondents identified decreased rainfall as the leading cause of reduced river flow.

It was noted that in the wildlife management perspective, the decline in river water levels poses serious threats to ecosystem stability and species survival. Reduced water availability limits access to essential resources for both wildlife and vegetation, leading to increased competition, habitat degradation, and in some cases, species migration or mortality. Drying rivers also affect the ecological balance by disrupting breeding grounds, food availability, and movement corridors, which

in turn threatens biodiversity conservation efforts. Moreover, water scarcity heightens human–wildlife conflicts as both communities and animals compete for limited resources, complicating conservation strategies. Effective wildlife management in such contexts requires integrated water resource management, climate change adaptation measures, and strengthened conservation practices to ensure the sustainability of ecosystems and wildlife populations.



Figure 27: Multi-temporal satellite image showing decrease in water in the Great Ruaha River, left image (October 2021), right image (March 2022). **Source:** Google Earth (2021 and 2022)

Land Use/Cover Types over the past 30 years (1987 - 2017)

Landsat images of the Ruaha–Rungwa landscape for 1987, 1997, 2007, and 2017 were analyzed to detect and quantify changes in land use/cover classes. Six main classes were identified: bare soil, bushland, forest, grassland, water, and wetland (Table 1). Of these, bushland, forest, and grassland dominated the landscape, together accounting for more than 80% of the total area across the 30-year period. In 1987, bushland (36%),

forest (35%), and grassland (26%) were the most extensive land cover types. By 1997, both forest and bushland declined by 3% and 5% respectively, while grassland expanded by 8%. In 2007, bushland continued to decline sharply to 22% (a 9% loss compared to 1997), whereas forest and grassland increased to 38% and 36% respectively. By 2017, grassland remained dominant (36%), followed by bushland (33%) and forest (28%). Other classes such as bare soil, water, and wetland consistently covered less than 1%

Table 1: Area (ha) and percentage of area covered by LULC classes in Ruaha-Rungwa Landscape

Classes	1987		1997		2007		2017	
	Ha	%	Ha	%	Ha	%	Ha	%
Bare Soil	20,330	0.5	20,202	0.4	36,418	0.8	26,414	0.6
Bushland	1,652,536	36.6	1,432,159	31.6	1,013,286	22.5	1,524,995	33.8
Forest	1,614,951	35.8	1,455,963	32.3	1,625,056	38.2	1,261,954	28.0
Grassland	1,166,803	26.1	1,562,166	34.6	1,642,053	36.4	1,661,839	36.1
Water	21,100	0.5	16,096	0.4	20,066	0.4	15,084	0.3
Wetland	25,186	0.6	24,299	0.5	64,035	1.6	9,629	0.2
Total	4,510,906	100	4,510,896	100	4,510,915	100	4,510,915	100

Source: Field Data Collection, (2021).

The classification of NDVI values further revealed severe vegetation disturbances over the three decades. In 1987, most NDVI values (0.3–0.4) reflected shrubland, bushland, and woodland. By 2017, these values had dropped to 0.1–0.2, indicating a shift toward meadow and grassland, largely due to deforestation for fuelwood, timber, and medicinal plants. Analysis of the rate of change (Figure 7) revealed substantial net losses and gains among different land cover classes. Between 2007 and 2017, wetlands decreased by 20%, forests by 3%, water by 2%, and bare soil

by 2%. Conversely, bushland and grassland recorded modest gains of 4% and 0.1%. Wetland and water loss were largely attributed to river drying, leading to increased bare land. Between 1997 and 2017, wetlands (9%), water (0.6%), and forest (1%) declined, while bare soil (3%), bushland (0.6%), and grassland (0.6%) increased. Over the full 30-year period (1987–2017), wetlands and water experienced net losses of 9% and 3% respectively, while forest declined by 2% and bushland by 0.8%, contributing to a 3% gain in grassland and 2% in bare soil.

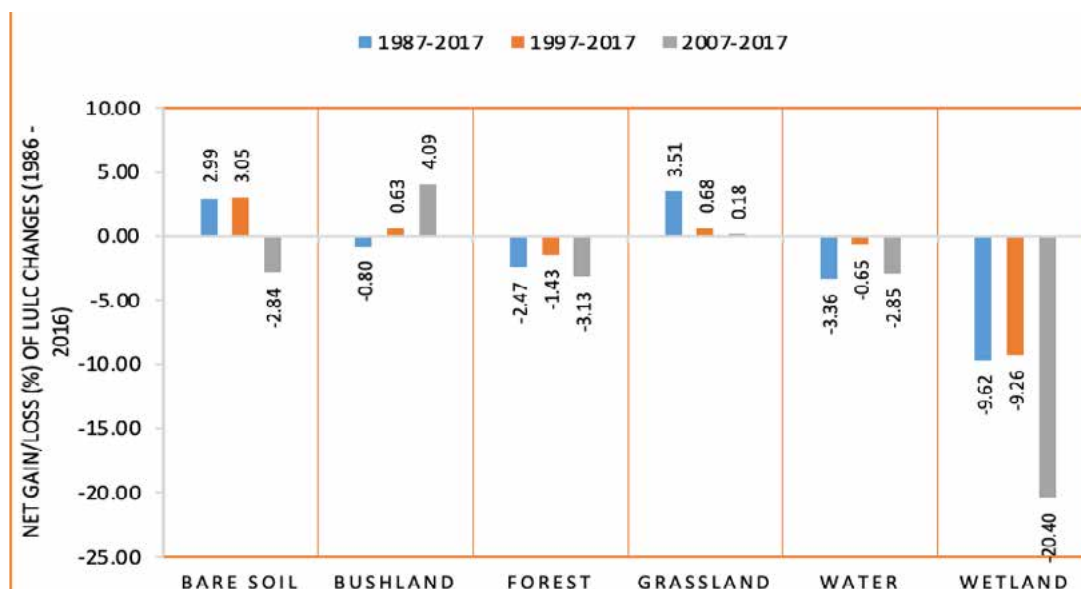


Figure 28:Rate of change of land use/cover types over the past 30 years from 1987 to 2017 in the Ruaha – Rungwa landscape.

Source: Field Data Collection, (2021).

Overall, the Ruaha–Rungwa landscape experienced a significant reduction in wetlands, water bodies, bushland, and forest cover, accompanied by expansion of grassland and bare soil. These transformations are linked to both climatic pressures, such as prolonged droughts, and anthropogenic drivers including deforestation, illegal timber extraction, charcoal production, and river channel diversion. Underlying these pressures are broader socio-economic challenges such as poverty, population growth, shifts in livelihood strategies, and weak enforcement of resource use policies. Importantly, these land use/cover changes carry direct implications for wildlife management. While core protected areas such as Ruaha National Park and Rungwa Game Reserve have remained relatively intact, the landscape edges have undergone intense transformation driven by human encroachment. Expanding agricultural activities, livestock grazing, and resource extraction around the peripheries have fuelled escalating human–wildlife conflicts, including crop destruction, livestock predation, and even loss of human life. These dynamics highlight the urgent need to link land use change drivers with strategies for sustainable wildlife resource management in Tanzania’s southern protected areas.

Destruction of habitat for wildlife

The analysis of land use/cover over the past 30 years revealed significant shifts in vegetation types within the Ruaha–Rungwa landscape. Specifically, there was a marked net loss of wetlands, water bodies, and forest cover, accompanied by a net gain in grassland, bushland, and bare soil. Such transformations indicate increasing ecosystem degradation and fragmentation. Ecosystems are highly vulnerable to climate change impacts, including rising temperatures and altered rainfall patterns, which exacerbate habitat loss. The study further observed that intact and healthier ecosystems demonstrate greater resilience and adaptive capacity to climate change compared to degraded systems, underscoring the importance of maintaining ecosystem integrity for wildlife survival.

It was reported by the respondents that destruction of habitat poses critical implications for wildlife resources management. The loss of wetlands, forests, and water bodies diminishes the availability of essential resources such as food, water, and shelter, leading to declines in wildlife populations and threatening the survival of many species. This loss also results in reduced biodiversity, with specialist and endangered species being particularly at risk of extinction. Habitat fragmentation further compounds the problem by disrupting migration routes and reducing genetic diversity, which weakens the long-term resilience of wildlife populations. In addition, habitat degradation increases the frequency of human–wildlife conflicts, as animals are forced to move closer to human settlements in search of food and water. This creates challenges for wildlife managers, who must devise and implement conflict mitigation strategies while also balancing the needs of local communities. The degradation of ecosystems also undermines the provision of vital ecosystem services, such as water regulation and carbon sequestration, which support both conservation goals and community livelihoods. Consequently, wildlife resources management faces rising costs related to law enforcement, monitoring, habitat restoration and conflict resolution.

Invasion and domination of Invasive species

The study revealed that climate change is influencing rangeland composition, largely through the spread of invasive species. Findings from key informant interviews (53%) (n=16) indicated that within the studied protected areas (PAs), there is an increasing trend of invasive species dominance, driven partly by climatic factors. This shift poses a conservation threat by displacing native plant species and altering ecosystem balance. For instance, one key informant highlighted several fast-spreading invasive species, including Kongwa weed (*Astripomoea hyoscyamoides*), milkweed (*Asclepias*), ironweed (*Vernonia gigantea*), giant weed (*Heracleum mantegazzianum*), and *Solanum incanum*. These invasive plants

reduce the availability of forage for herbivores, which in turn undermines wildlife-based tourism activities in the area. Additionally, respondents noted that variations in rainfall and temperature patterns have facilitated the spread of certain invasive species, particularly in Ruaha National Park, where the giant weed was identified as especially responsive to increased rainfall patterns.

It was noted that the increasing dominance of invasive species in rangelands has several critical implications in the wildlife management perspective. First, invasive plants outcompete native vegetation, reducing the quality and quantity of forage available for herbivores. This decline in food resources can negatively affect wildlife populations, leading to reduced carrying capacity of the ecosystem and altering species distribution. Second, the loss of native plant diversity undermines ecological balance, making habitats less resilient to climate variability and disturbances such as droughts or fires. Third, the spread of invasive species diminishes the aesthetic and ecological value of protected areas, which directly impacts wildlife-based tourism a key source of revenue for conservation. Furthermore, the management of invasive species requires significant resources, including manpower, financial investment, and technical expertise, which can strain already limited conservation budgets. If left unchecked, invasive species may trigger long-term ecological degradation that complicates conservation goals, disrupts predator-prey dynamics, and reduces the effectiveness of protected areas as wildlife refuges.

Eruption of diseases and death of wildlife

This study findings obtained through interviews with key informants from PA staff (67%) (n=9) indicated that, climate change impacts over the past 30 years within the study area was mentioned to cause death of some wild fauna especially during the periods of intensive decline and drying of water resources. Aquatic species were mentioned to be potential victims when there is decline of water quantity and quality. For example, one of the ecologists was of the view that hippopotamus and fish species were the mostly affected by the impacts emanating from changes in climatic conditions in the protected areas.

The study recorded **63** wildlife deaths in Ruaha National Park during 2019/2020 attributed to an anthrax outbreak (Table 2), with most cases occurring along the Great Ruaha River during the peak dry months of August and September. Anthrax emergence in the area is linked to prolonged drought, dry spells following heavy rains, and flooding events, which can mobilize dormant spores into water systems. Key informants noted that the severe drying of the Great Ruaha River in 2017 followed unusually high river flows in 2016, likely flushing spores into Jongomero and Nyamakifu dams and exposing hippos to infection. These findings align with Stears *et al.* (2019), who report that alternating cycles of flooding and extreme drying create environmental conditions conducive to anthrax outbreaks in Ruaha.

Table 2: Wildlife Mortality from 2017 - 2020

Year	Months	Location	Animal Species	Number of Mortality	Age of animals	Total
2017	February	Ibuguziwa	Hippos	7	Adult	72
	February	Lunda	Hippos	53	Adult	
	August	Jongomero	Giraffe	5	Adult	
	September	Jongomero	Hippos	1	Adult	
	September	Maganga Madungu	Hippos	5	Adult	
	September	Telekimboga	Hippos	1	Adult	

Year	Months	Location	Animal Species	Number of Mortality	Age of animals	Total
2018	September	Ibuguziwa	Hippos	7	Adult	7
2019 / 2020	February	Kariakoo area (Jongomero)	Giraffe	2	Adult	63
	February	Hussman bridge area	Hippos	2	Adult	
	August	Tourist Bandas	Hippos	16	Adult	
	September	Nyamakifu	Hippos	33	Adult, 1baby	
	September	Maganga Madungu	Hippos	5	Adult	
	September	Telekimboga	Hippos	1	Adult	
	September	Hussman bridge	Hippos	4	Adult	
Total						142

Source: Ruaha Veterinary Office (2021).

The eruption of diseases and the resulting death of wildlife due to climate change can have significant impacts on wildlife management. High mortality rates can lead to population declines, particularly among vulnerable species like aquatic fauna, which may disrupt ecosystem balance and food webs. Disease outbreaks can reduce genetic diversity and weaken the resilience of wildlife populations, making them more susceptible to future environmental stresses. Such impacts may also increase the pressure on wildlife managers to respond to sudden population losses and maintain ecological stability, complicating conservation planning and resource allocation.

Increasing human - wildlife conflicts

The results of this study showed that the effects of climate change are one of the factors causing changes in land use cover change by influencing the decline of water and forage in protected areas, particularly during dry seasons when the area is shown to experience severe human-wildlife conflicts. As a result, wildlife species has been forced to go outside PAs into village areas in search of their needs for instance food and water, consequently raiding crops, destroying infrastructures, attacking livestock and sometimes injuring and killing people. Land use change is among the root influencing the conflict between wildlife and people in Ruaha-Rungwa ecosystems. The findings indicate that climate change-driven

land use and resource changes are intensifying human-wildlife conflicts, which has significant implications for wildlife management. Wildlife moving outside protected areas in search of water and forage increases the risk of crop damage, livestock predation, and attacks on humans, thereby heightening tensions between local communities and conservation authorities. This situation challenges managers to balance species conservation with community safety, necessitating strategies that address habitat degradation, resource availability within protected areas, and mitigation of conflicts to maintain both ecological integrity and human livelihoods.

Protected Areas dis-connectivity within the Ruaha-Rungwa Landscape

Climatic impacts have been found to alter the connectivity of the Ruaha-Rungwa landscape, affecting the replenishment of ecological resources and overall ecosystem functioning. This study evaluated the connectivity between landscapes in the Southern Circuit to identify appropriate conservation strategies to address challenges, particularly regarding wildlife accessibility within the region. Findings from overlaid multi-layered maps (Figure 9) revealed that the ecological integrity and connectivity of PAs have been compromised by several challenges, including blocked dispersal areas and altered wildlife movement patterns. These blockages are largely driven by expanding

human activities along PA borders, such as cultivation, livestock keeping, settlements, and deforestation. Analysis of wildlife corridors in the Ruaha–Rungwa landscape showed that the area once had multiple corridors linking major protected areas in southwestern Tanzania. However, several key routes, particularly those

connecting the Ruaha–Rungwa landscape to northern regions like Ngorongoro Conservation Area and Swagaswaga Game Reserve, are now closed due to severe habitat encroachment and intensive crop cultivation starting from the boundary of Rungwa Game Reserve.

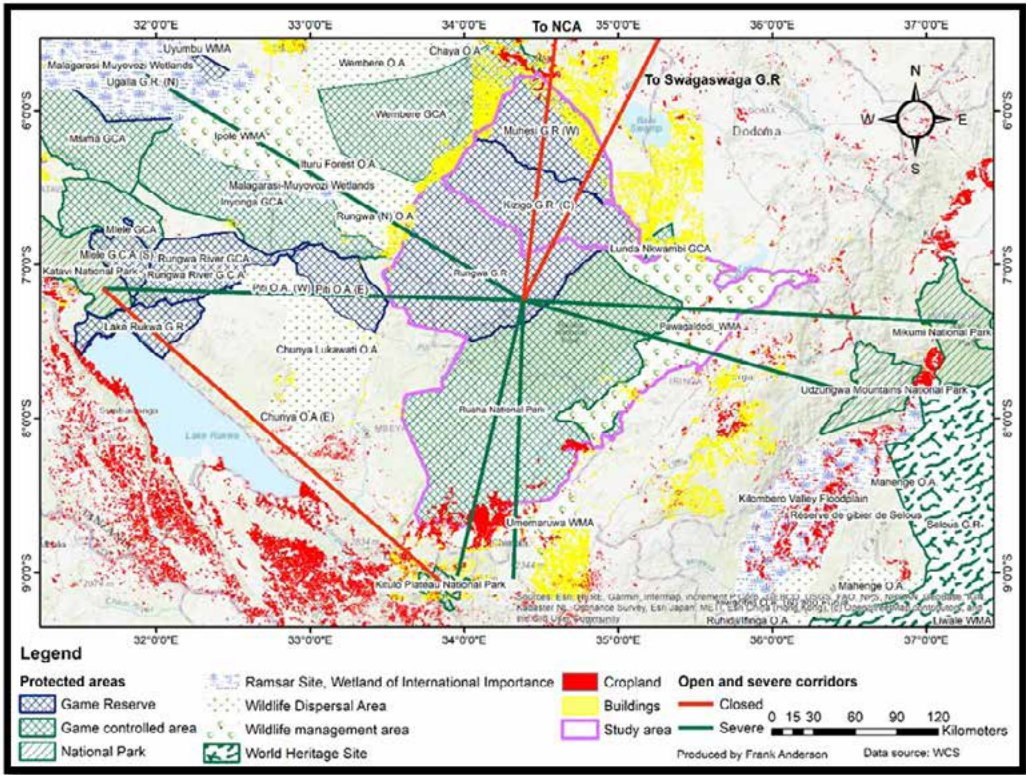


Figure 29: Open and closed wildlife routes in the Ruaha – Rungwa landscape
Source: Anderson (2023).

The disruption of ecological connectivity in the Ruaha-Rungwa landscape has significant implications for wildlife resource management. Blocked dispersal areas and closed wildlife corridors limit species’ ability to access critical habitats, water, and forage, increasing population fragmentation and vulnerability to environmental stressors. This challenges managers to maintain genetic diversity, ensure sustainable wildlife populations, and mitigate the impacts of human-wildlife interactions. The findings underscore the need for strategic planning to protect remaining corridors, restore degraded habitats, and manage land-use activities surrounding protected areas to sustain

ecosystem functioning and wildlife movement across the landscape.

The findings of this study indicate that climate change and variability over the past three decades have significantly influenced meteorological patterns, land use, and ecosystem dynamics within the Ruaha-Rungwa landscape, with profound implications for wildlife resources management. Analysis of annual rainfall trends from the 1980s to 2018 revealed a complex pattern where extreme rainfall events occurred in certain years (e.g., 2009, 2015–2018), contrasting with findings from Usangu Plains, which experienced a

long-term decline in rainfall (Ndomba, 2020). Despite occasional heavy rainfall, precipitation analyses indicated that Ruaha has received reduced rainfall in several years compared to 1987, reflecting local increases in rainfall extremes rather than consistent rainfall increases. Temperature trends corroborated previous findings from Iringa weather station (Pettengell, 2016) and projections by Tomalka *et al.* (2011) and the IPCC, indicating rising mean annual temperatures and a projected increase in very hot days, which further exacerbate climatic stress on ecosystems.

Over the past 30 years, land use and cover changes within the landscape have been driven by both climatic and anthropogenic factors. In 1987, the area was dominated by dense forests, miombo woodlands, and bushlands (Barnes, 1982; Coppolillo, 2004). However, settlement expansion, cultivation, livestock keeping, and deforestation have gradually transformed the landscape, increased grasslands while reducing forests and bushlands (Nahonyo, 2005). Wetlands and water resources have also been converted to bare soil, primarily due to extensive upstream water extraction for irrigation, which has reduced river flows, including in the Great Ruaha River (Chilagane *et al.*, 2020; Kashaigili *et al.*, 2007). Such changes have directly affected wildlife, leading to increased human-wildlife conflicts, crop raiding, livestock predation, and mortality of aquatic and terrestrial species (Hariohay, 2019; Komba *et al.*, 2021; Stommel *et al.*, 2016). Vegetative transformation, particularly increased grassland at the edges of protected areas, has been associated with anthropogenic pressures, altering wildlife habitats and resource availability (Estes *et al.*, 2012; Hariohay *et al.*, 2019).

Climatic changes have further contributed to habitat degradation through rising temperatures, higher evaporation, prolonged droughts, and increased frequency of forest fires and pest outbreaks (Kivuyo & Masao, 2020; Eliasch, 2008). These factors, combined with socio-economic drivers such as poverty,

population growth, and policy mismatches between conservation and livelihoods, have intensified land cover change, reducing suitable habitats for species adapted to specific vegetation types, such as small herbivores (Otieno, 2019; Kideghesho *et al.*, 2013; Caro & Davenport, 2016). Water scarcity due to declining river flows has also affected wildlife movement, access to forage, and overall health, as observed in other ecosystems like the Serengeti (Kilungu *et al.*, 2016; Kupika *et al.*, 2018). In addition, climate variability has facilitated the proliferation of invasive species, which compete with native flora and reduce forage availability for wild animals, further compromising ecosystem integrity (Hoffman & Vogel, 2008; Bhowmik, 2005; Hall *et al.*, 2020; FAO, 2012; Kideghesho *et al.*, 2013; Mainka & Howard, 2010).

The study also highlighted the role of climate change in triggering disease outbreaks among wildlife, with shortages of water and forage linked to increased incidence of diseases such as anthrax and vector-borne illnesses (Ngetich, 2019; Robinson *et al.*, 2005; Kutz *et al.*, 2009; Liao *et al.*, 2015; van Panhuis *et al.*, 2015; Onyango *et al.*, 2020; Faiman *et al.*, 2017). Land use changes further influence disease dynamics by altering host and vector interactions, population densities, and habitat conditions (Gottdenker *et al.*, 2014). Finally, the Ruaha-Rungwa landscape has experienced significant disruptions in ecological connectivity, with key wildlife corridors between major protected areas such as Ruaha, Tarangire, Nyerere, and Swagaswaga being blocked due to habitat encroachment and anthropogenic activities (Riggio & Caro, 2016; Lohay *et al.*, 2020; Jones *et al.*, 2012). These blockages have isolated populations of elephants, lions, sable, and other species, threatening genetic connectivity and long-term survival (Luikart *et al.*, 2010). Recent conservation measures, such as upgrading the Kilombero Valley to Game Reserve status, aim to enhance protection and restore ecological connectivity (MNRT, 2023). In summary, the findings demonstrate that climate variability, coupled with land use changes and human

pressures, has profoundly affected the Ruaha-Rungwa ecosystem, challenging wildlife resource management by reducing habitat quality, limiting water and forage availability, facilitating invasive species, increasing disease risk, and fragmenting wildlife populations.

CONCLUSION AND RECOMMENDATIONS

In conclusion, this study demonstrates that climate change over the past three decades (1987 - 2017) has significantly impacted the Ruaha-Rungwa landscape, with major implications for wildlife resources management. While core protected areas such as Ruaha National Park and Rungwa Game Reserve have remained relatively intact, the surrounding landscape edges have undergone intense transformation, affecting habitat quality and connectivity. Changes in rainfall patterns, rising temperatures, and recurrent droughts have further contributed to habitat degradation, reduced water availability, spread of invasive species, increased disease outbreaks, and fragmentation of wildlife populations. These climatic stressors, combined with the altered landscape edges, threaten the movement, survival, and long-term resilience of wildlife in the landscape.

This paper suggests the importance of developing strategies for the restoration of degraded habitats and the safeguarding of wildlife corridors in order to strengthen ecological connectivity and facilitate species movement across the Ruaha-Rungwa landscape. It further emphasizes the value of promoting sustainable water resource management, particularly through the conservation of rivers, wetlands, and riparian areas, to ensure reliable water availability for wildlife. In addition, the paper highlights the need to integrate climate adaptation strategies into conservation planning so as to better address the challenges posed by droughts and unpredictable rainfall patterns. It also points to the benefits of establishing monitoring and management programs aimed at controlling invasive species, conserving native vegetation,

and enhancing wildlife health surveillance systems for early detection and timely response to disease outbreaks. Finally, the paper draws attention to the importance of giving careful consideration to highly transformed landscape edges, with a view to reducing habitat degradation and sustaining the resilience of wildlife populations both within and beyond core protected areas.

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MAPPING STAKEHOLDERS AND ASSESSING COMMUNITY PERCEPTIONS FOR AI-DRIVEN CLIMATE RESILIENCE IN THE RUAHA-RUNGWA ECOSYSTEM, TANZANIA

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ABSTRACT

Rural communities bordering Tanzania's protected areas (PAs) face escalating climate resilience challenges, exacerbated by human-wildlife conflicts (HWC) and fragmented stakeholder efforts. This study conducted a comprehensive stakeholder and socio-economic mapping in the Ruaha-Rungwa Ecosystem, Iringa District, to identify key actors and assess community perceptions as a foundation for integrated, technology-supported solutions. Using a mixed-methods approach, we engaged 392 participants through questionnaires (91.8%), focus group discussions (4.8%), and key informant interviews (3.3%). Findings reveal critical stakeholder fragmentation, with farmers, entrepreneurs, pastoralists, and government agencies operating in silos. The study recorded a significant gender imbalance among respondents, with males comprising 63.8% of the sample. Agriculture is the primary livelihood (80.1%), with communities reporting severe climate impacts: 94% of farmers reported reduced food production due to droughts and erratic rainfall. Despite low smartphone ownership (28%), awareness of Artificial Intelligence (AI) was relatively high (72.4%), and 89.9% of respondents indicated that AI could aid climate adaptation, particularly through early warning systems and precision agriculture. The study concludes that, despite the presence of challenges, there is a strong community support for innovative solutions. We recommend targeted investments in digital infrastructure, the development of locally relevant AI application, and establishment of robust multi-stakeholder collaboration frameworks to enhance climate resilience and promote sustainable coexistence adjacent to protected areas.

Keywords: Adaptation,, Artificial Intelligence, Community Engagement, Climate Change, Protected Areas

INTRODUCTION

Tanzania's extensive network of protected areas (PAs), covering approximately 33% of the national territory, is a cornerstone of global biodiversity conservation (TAWIRI, 2020). However, these PAs interface directly with rural communities, creating a complex dynamic where conservation goals can clash with human livelihoods. Communities adjacent to PAs face a dual threat: significant human-wildlife conflicts (HWC) and the escalating impacts of climate change (Mwakalila *et al.*, 2023). Between 2020 and 2023, wildlife

interactions resulted in one human death every 52 hours and the destruction of 41,404 hectares of crops, despite government compensation of 4.6 billion Tanzanian shillings. Concurrently, climate-induced challenges, including erratic rainfall, droughts, and floods undermine agricultural productivity and threaten food security (URT, 2022).

Technological innovations, particularly Artificial Intelligence (AI), offer transformative opportunities to enhance climate resilience; however their adoption and application in rural Tanzanian contexts remains at an early stage.

AI-driven tools, such as early warning systems, climate forecasting models, and precision agriculture can help communities anticipate climate-related hazards and implement effective adaptive measures (Vaughan *et al.*, 2022). However, the successful deployment of such technologies hinges on a deep understanding of the local socio-economic landscape, existing stakeholder networks, and community perceptions (Garnett *et al.*, 2018).

This study, supported by the Tanzania Commission for Science and Technology (COSTECH) and NORAD under the NFAST climate research program, sought to address this gap. Conducted in the Ruaha-Rungwa Ecosystem, the research undertook a comprehensive stakeholder mapping and assessed community knowledge, attitudes, and perceptions (KAP) regarding climate change and the potential application of AI-driven solutions. The findings are intended to inform the co-design of contextually appropriate, technology-supported interventions that

bolster climate adaptation, mitigate HWC, and strengthen community resilience.

MATERILA AND METHODS

Genesis of Stakeholders Mapping and Study Area

Stakeholder mapping is the process of identifying, categorising, and prioritising stakeholders based on their influence, interests, and impact. A stakeholder is anyone who stands to gain or lose from the results of a given intervention or the actions of other social actors. Individuals, groups, or institutions are considered stakeholders if they affect or are affected by the way certain activities are implemented, or if they possess information, resources, experience, or other forms of power that can influence others. Stakeholder mapping helps businesses, governments, and community leaders develop targeted engagement strategies by determining who needs to be informed, consulted, or closely managed (Fig 1).

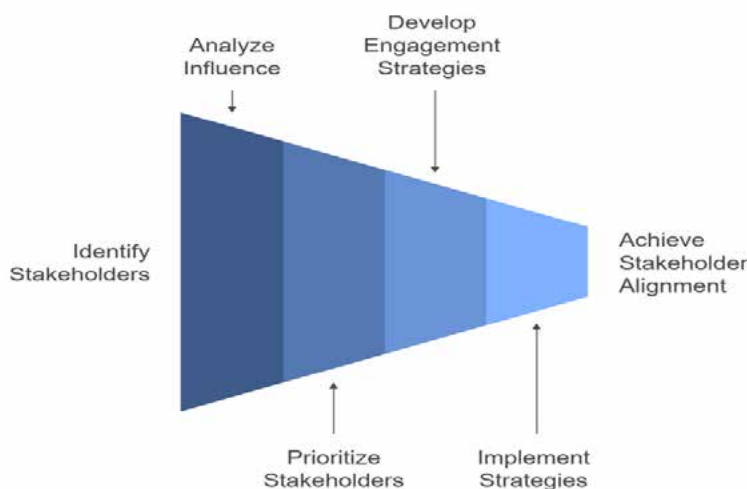


Figure 1: Stakeholders mapping model used

This research was conducted in the Ruaha-Rungwa Ecosystem, focusing on 12 villages across four wards (Idodi, Mahuninga, Mlenge, and Mlowa) in the Iringa District. The total population of the study area is 44,002, with a slightly higher proportion of women

(53%) than men (47%) (National Bureau of Statistics, 2022). The area is characterised by its proximity to major protected areas (PAs), a high dependence on rainfed agriculture, and is a known hotspot for human-wildlife conflict (HWC) (Kicheleri *et al.*, 2018).

Research Design and Data Collection

A sequential mixed-methods approach was employed, combining quantitative and qualitative techniques to triangulate data and provide a comprehensive understanding of the research problem (Creswell & Plano Clark, 2017). The study population comprised adults from villages bordering the Ruaha National Park and other PAs. A multistage sampling technique was applied. First, purposive sampling was used to select wards and villages based on their proximity to protected areas (PAs) and high incidence of Human-Wildlife Conflict (HWC). Subsequently, a combination of systematic random sampling (for questionnaires) and

purposive sampling (for FGDs and KIIs) was used to select individual participants, ensuring representation across gender, age, and livelihood groups. The proposed methodology consisted of six steps (Fig 2). It began with an initial classification of stakeholder's present (physically or symbolically) in the area, followed by identification and recognition of relationships among them and mapping of existing social networks. The process placed special emphasis on defining the roles and functions of different stakeholders in relation to the intervention. Matrices and visual representations were used to illustrate stakeholder relationships and interactions.

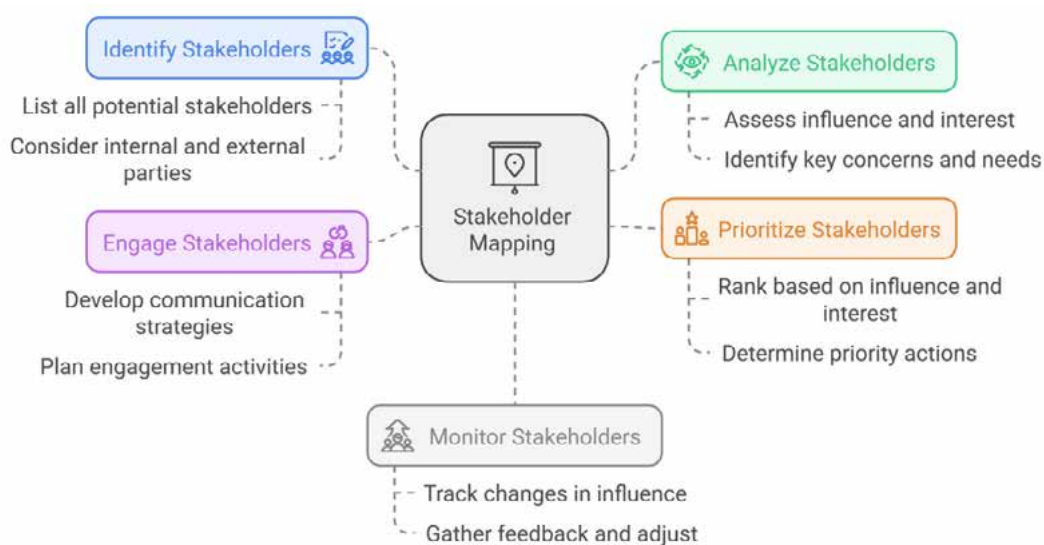


Figure 2: A multiple-step approach adopted for the stakeholders' physical and social mapping, Source: <https://www.6sigma.us/>

Figure. 3 presents the population distribution of primary data collected from 392 participants using three data collection instruments:

- Structured questionnaires (n = 360, 91.8%) administered to community members to collect quantitative data on demographics, livelihoods, climate impacts, and technology awareness;
- Focus group discussions (FGDs, n = 19 participants across four groups, 4.8%) conducted with separate groups of farmers, pastoralists, and entrepreneurs to gain in-depth insights into lived experiences and

collective coping mechanisms; and

- Key informant interviews (KIIs, n = 13, 3.3%) held with local government leaders, agricultural and livestock extension officers, village game officers, and representatives from accommodation facilities and religious institutions to understand institutional perspectives and existing coordination mechanisms.

All data collection tools were pre-tested and translated into Kiswahili, and administered by trained enumerators.

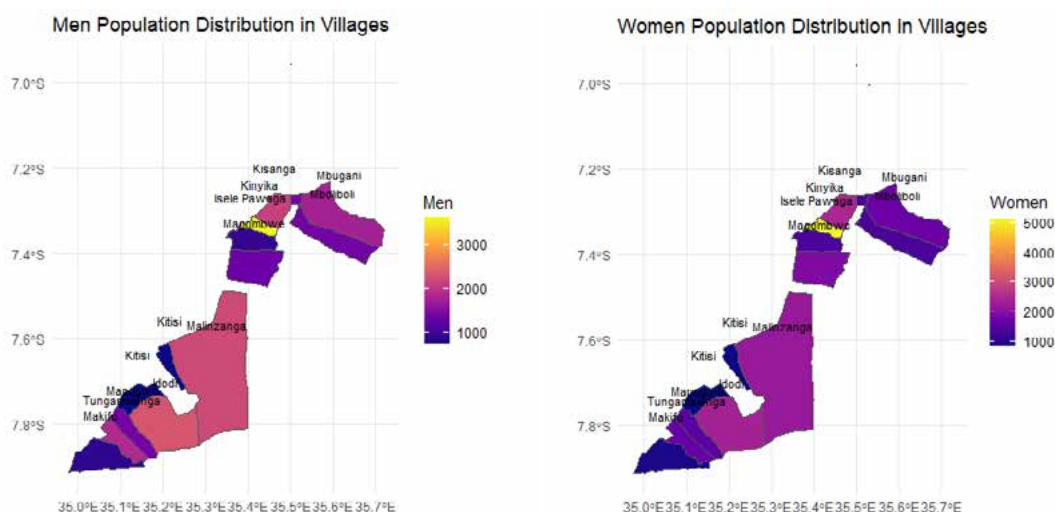


Figure 3: Response distribution by gender, sourced from Researcher’s construct, 2025

Data Analysis

Quantitative data from questionnaires were coded, cleaned, and analysed using IBM SPSS Software (Version 28). Descriptive statistics (frequencies, percentages, and means) were generated to summarise socio-economic characteristics and perceptions. Qualitative data from FGDs and KIIs were audio-recorded, transcribed, translated into English, and subjected to thematic analysis following the framework by Braun and Clarke (2006). Key themes included perceptions of climate change causes, impacts of HWC, existing adaptation strategies, and attitudes toward technology. Stakeholder mapping followed a stepwise process of identification, categorisation, and analysis of relationships and influence, presented in a matrix format.

Ethical Considerations

Ethical approval was obtained from relevant authorities, including the Tanzania Commission for Science and Technology (COSTECH), the Tanzania Wildlife Research Institute (TAWIRI), and the President's Office – Regional Administration and Local Government (PO-RALG). Informed consent was obtained from all participants prior to data

collection. The study ensured anonymity and confidentiality, with all data securely stored on password-protected servers in compliance with Tanzania's data protection guidelines.

RESULTS

Demographic and Socio-Economic

The sample comprised 63.8% male and 36.2% female respondents (Table 3.2). The majority of respondents were aged 36-60 years (46.9%), followed by those aged 18-35 years (40.9%). The majority of households (58.4%) were small, comprising of 1-4 members, in line with the national average. Educational attainment was predominantly at the primary level (69.6%), with only 5.6% of respondents having completed college or university education. Agriculture was the primary livelihood for 80.1% of respondents, with other activities being fishing, civil servants, tourism-related activities, and livestock keeping.

Land and Livelihoods

The majority of respondents (69.0%) owned land (Fig. 4a), primarily small plots ranging

from 1.1 to -5 acres (52.8%). Notably, 37% of landowners did not possess formal title deeds. Rice (43.4%) and maize were the dominant crops, often cultivated in mixed cropping

systems. A substantial majority of farmers (94%) reported that climate change had adversely affected their capacity to produce sufficient food (Fig. 4b).

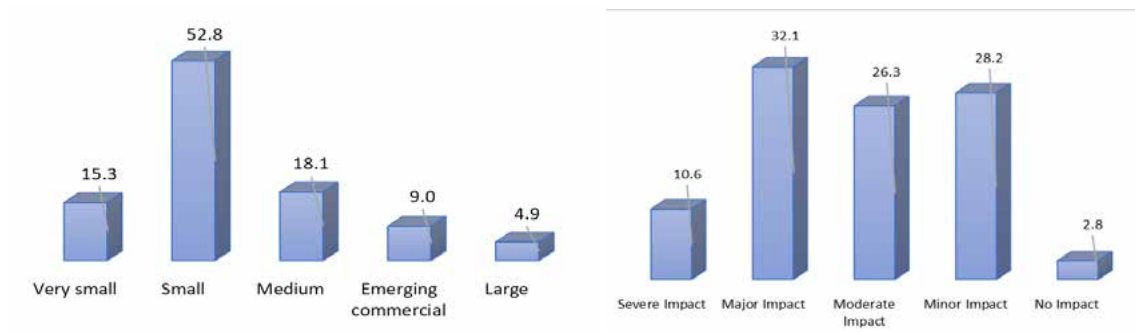


Figure 4: a. Landownership by size

b. Opinion on the effect of climate change

Proximity to Protected Areas and HWC

The majority of respondents resided at a considerable distance from PAs, with 35.7% residing more than 20 km away (Fig. 5). A similar pattern was observed for farm locations,

with 35.1% situated more than 20 km from the nearest PA. Despite these distances, human-wildlife conflict (HWC), particularly in the form of crop raiding, remained a pervasive issue.

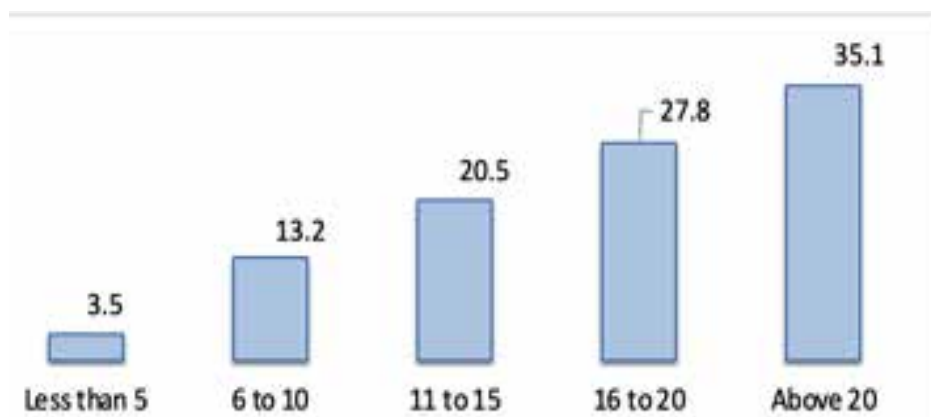
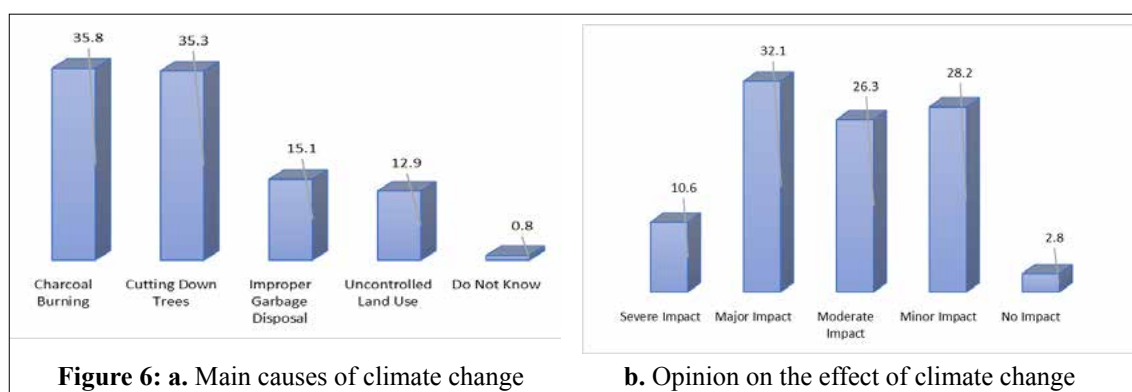


Figure 5: Distance from homes to the nearest protected area (in kilometres)

Knowledge and Perceptions of Climate Change

Respondents demonstrated a high level of awareness regarding climate change.. Figure 6a show that the main perceived drivers of climate change were local environmental degradation, specifically charcoal burning (36%) and tree

cutting (35%). The most observed climate impacts over the past decade (Fig. 6b) were droughts (73%), rising temperatures (63%), and water scarcity (61%). Notably, 75% of respondents considered that communities themselves bear the primary responsibility for addressing climate change, whereas only 23% attributed this role to the government.



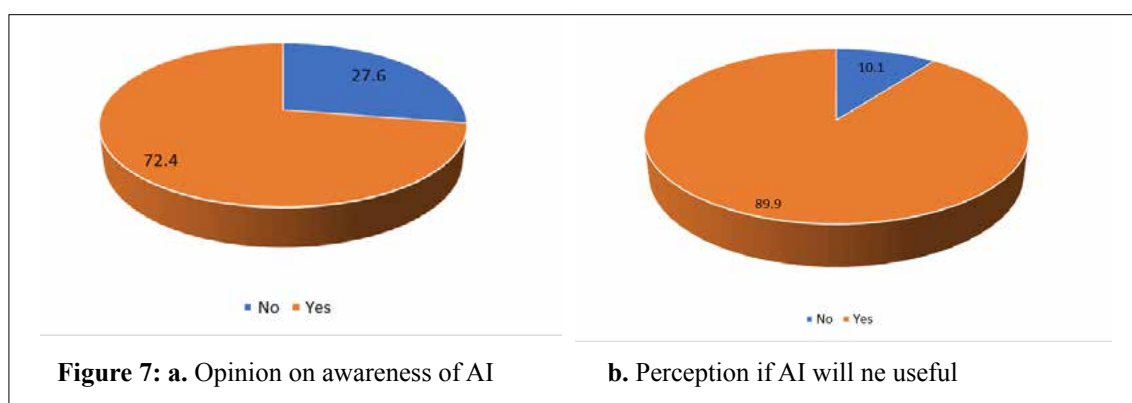
Stakeholder Mapping and Collaboration

The stakeholder mapping identified a diverse range of actors, including farmers, pastoralists, entrepreneurs, accommodation lodges, religious institutions, and various government agencies. A key finding was the lack of collaboration, with each group operating independently in silos.

A quotation from one of the respondents highlighted this fragmentation: *“I have been living in this area... but I had never seen or heard of a meeting that would bring together different stakeholders...”*

Awareness and Perceptions of AI

Figure 7 depicts results of awareness and perception of artificial intelligence (AI). Despite low smartphone ownership (28%), awareness of AI was relatively high (72.4%). The majority of respondents (89.9%) perceived that AI could help their community in adapting to climate change. Respondents also expressed strong willingness to adopt AI-driven solutions, particularly for early warning systems for extreme weather and wildlife movement, accurate weather forecasting, and the detection of disease in crops and livestock.



DISCUSSION

This study reveals that communities within the Ruaha-Rungwa Ecosystem are ensnared in a feedback loop of climate vulnerability and human-wildlife conflict, a pattern that is increasingly documented across protected area networks throughout Africa (Kiffner *et al.*,

2020). Our finding that 94% of farmers attribute reduced food production to climate shocks aligns with national climate vulnerability assessments (URT, 2022) but presents a more pronounced impacts than some regional studies (e.g., Moyo *et al.*, 2022, who reported 78% in Southern Africa), underscoring the acute sensitivity of this agro-ecological zone.

Contrasting Perceptions of Causation and Responsibility

A critical divergence emerges between local and global narratives on climate change. While the international discourse emphasises greenhouse gases, communities predominantly attribute climate change to local environmental degradation like charcoal burning and deforestation. This mirrors findings by Shemweta and Kideghesho (2023) in the Serengeti, suggesting a widespread perception rooted in immediate, visible environmental stressors. This local perception, however, presents an opportunity. It aligns conservation (reducing deforestation) directly with climate adaptation, a synergy often missed in top-down policy (Garnett *et al.*, 2018). Furthermore, the predominant attribution of responsibility for action to local communities (75%), rather than government (23%), contrasts sharply with the typical state-centric model of environmental governance. This indicates a strong sense of local agency but may also reflect a history of limited external support, potentially leading to maladaptive coping strategies if communities are left unsupported (Mutanga *et al.*, 2017).

Fragmentation vs. Integration in Stakeholder Networks

The observed fragmentation amongst stakeholder is a major barrier to resilience. In contrast to the integrated co-management models successfully implemented in some Namibian conservancies (Weaver & Skyer, 2020), the siloed operations of farmers, pastoralists, and government agencies in Ruaha-Rungwa lead to disjointed efforts. This fragmentation likely exacerbates HWC, as a lack of coordinated land-use planning and information sharing increases competition over scarce resources. This finding supports the argument by Kicheleri *et al.* (2018) that institutional weaknesses, not just ecological factors, are primary drivers of persistent HWC in Tanzania.

The Digital Divide and the Window of Opportunity for AI

The paradox of relatively high artificial intelligence (AI) awareness (72.4%) amidst low smartphone ownership (28%) highlights the complexity of the digital landscape. This contrasts with studies in more remote areas where both awareness and access are low (e.g., Zvoleva *et al.*, 2021). The strong optimism regarding AI's potential (89.9%) suggests that information is being disseminated through channels such as television and radio, creating a "readiness" that precedes infrastructure. This presents a critical window. However, the risk is that techno-optimism outpaces practical delivery, leading to a "pilot paradox" where solutions fail to scale or become relevant (Vaughan *et al.*, 2022). Therefore, proposed AI solutions must be low-bandwidth, accessible via basic mobile phones (e.g., USSD-based alerts, community radio-integrated forecasts), and directly target the community's identified priorities, including early warnings and crop disease detection.

CONCLUSION AND RECOMMENDATIONS

This study establishes a critical baseline for the integration of artificial intelligence (AI) and other technologies into climate resilience strategies for communities adjacent Tanzania's protected areas (PAs). The Ruaha-Rungwa Ecosystem presents both a clear need for and a receptiveness to innovative solutions. However, the successful implementation of such technologies depends on addressing fundamental challenges, including stakeholder collaboration, digital literacy, and infrastructure. Based on our findings, we recommend the following actions:

- i. Foster multi-stakeholder platforms: Establish formal coordination mechanisms to break down silos and enable collaborative planning and resource sharing among all identified stakeholders.

- ii. Pilot context-specific AI tools: Develop and deploy user-friendly, low-tech AI applications focused on high-priority needs, such as community-based early warning systems for weather and wildlife movement.
- iii. Invest in Foundational Infrastructure: Prioritise investments in rural digital connectivity and electricity to enable the functionality of digital tools.
- iv. Implement inclusive capacity building: Conduct targeted digital literacy and AI awareness programmes for all community segments, with special efforts to engage women and youth.
- v. Integrate Indigenous and Scientific Knowledge: Design AI solutions that incorporate local knowledge and perceptions to ensure relevance, accuracy, and community ownership.

By adopting this integrated approach, Tanzania can harness the potential of AI to build resilient rural communities, ensure sustainable conservation of its protected areas, and adapt more effectively to the challenges of climate change.

ACKNOWLEDGEMENTS

We acknowledge the Tanzania Commission for Science and Technology (COSTECH) and NORAD for funding this research through the NFAST program. We extend our gratitude to TAWIRI, PO-RALG, and the Iringa regional and district authorities for their facilitation. We extend our sincere gratitude to the ward and village leaders, enumerators, and all community members in the Ruaha-Rungwa Ecosystem who generously shared their time and knowledge.

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SUB-THEME: HABITAT CONSERVATION, VEGETATION ECOLOGY AND INVASIVE SPECIES

PERCEPTION AND WILLINGNESS TO CONSERVE ON-FARM MEDICINAL PLANTS: INSIGHTS FROM COMMUNITIES IN WEST USAMBARA MOUNTAINS, TANZANIA

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ABSTRACT

The study analyses community perception on the availability and use of medicinal plants in Eastern Arc Mountains (EAM) and Willingness to conserve (WTCons) medicinal plants. household questionnaire surveys, Focus group discussion (FGDs) and Key informant interviews (KII) were used to gather information during the study. Socio-economic and demographic results indicate male-headed, married dominance with primary education being the most common qualification of respondents. Also, study confirms that there is inter-generational transfer of medicinal plants (MPs) knowledge from one generation to another. Many youths reported trusting modern medicines compared to adult ones. Also, results reveal a significant influence of the socio-economic variables on the perception of MPs availability and use in the study area. Cross-tabulation results reveal that age of respondents and gender significantly influence WTCons medicinal plants. Respondents' willingness to conserve MPs, highlighted urbanization as a threat to medicinal plants availability, age and household duration of the respondents significantly influence willingness to conserve medicinal plants in the study area. Also, community' perception of MPs availability and use highlighted that, collection of medicinal plants from on-farm, to treat wounds, injuries and challenges in accessing or harvesting medicinal plants both significantly influence on the availability and use of medicinal plants in the study area. The study suggested that long term environmental concern and exposure, rather formal education, are strong conservation behavior motivators. The study recommended further research to explore the dimensions on cultural beliefs and ecological knowledge since variables such as trust in traditional healers (TMPs) and community beliefs showed practical relevance.

Keywords: Community Perception, Willingness to Conserve and Medicinal Plants.

INTRODUCTION

Medicinal plants (MPs) possess significant global value, serving as the foundation of traditional healing practices and playing vital roles in contemporary medicine, cultural identity, and economic growth (Roy *et al.*,

2025). Medicinal plants represent a multi-billion-dollar industry globally, bolstering national economies and providing support to millions of rural families reliant on their cultivation, processing, and commerce. Throughout Asia, Africa, South America, and the Middle East, societies have been depending

on herbal treatments for millennia (Gurib-Fakim, 2006). As of today, the World Health Organization estimates that nearly 80% of the worldwide population depends on traditional medicine for primary healthcare (Bodeker and Ong, 2005). Nations like China, India, Japan, and Korea have developed complex and well-organized healthcare systems based on their abundant plant resources (Lambert *et al.*, 1997). In recent years, increasing numbers of people have been choosing herbal medicines or products to improve their health conditions, either alone or in combination with others. Herbs are becoming popular again and there is a worldwide revival in their use. The reports from the World Health Organization (WHO), show that about 75% of people around the world use herbs for their basic health care (Si-Yuan *et al.*, 2014).

Chinese Medicine is regarded as a historic medical system and is widely used as a health care method by people around the globe (Zhong, 2014). Traditional Herbal Medicine (THP) is an important part of the traditional Chinese Medicine Culture. Due to its high medicinal potential, it should not only serve the Chinese people's medical use but also contribute to global medicine (Zhang *et al.*, 2021). India, boasting more than 7,000 medicinal plant varieties, underpins the Ayurvedic (Science of life), Siddha (Traditional medicine from the southern India), and Unani practices (Recognized, Traditional and holistic healthcare system), employing plants like turmeric (*Curcuma longa*), neem (*Azadirachta indica*), brahmi (*Bacopa monnieri*), and ashwagandha (*Withania somnifera*) for various health issues, including inflammation and chronic fatigue (Pandey *et al.*, 2013). India also exports Ayurvedic medicines, essential oils, spices, and natural cosmetics globally, generating employment opportunities for women, youth, and indigenous groups engaged in wild harvesting and small-scale agriculture. Nowadays, pharmaceutical chemistry and medicine dealers from animals and plants (pharmacognosy) are playing a very crucial role in treatment for diseases and their prevention.

Medicinal plants are being used about 80% of the world population mostly in the developing countries in the primary health care (Subhose *et al.*, 2005).

Similarly, Kampo (Cultivated traditional medicine) medicine in Japan and traditional medicine in Korea significantly depend on cultivated and wild plant species such as *Angelica acutiloba*, *Aconitum carmichaelii*, and *Panax ginseng*, which are directly incorporated into their national healthcare systems (Watanabe *et al.*, 2011). At the same time, South Korea's ginseng sector brings in significant income which is about \$ 1,140 million making it the largest distributor of the ginseng in the world (Baeg & So, 2013), while Japan's Kampo medicine industry provides support to numerous farmers and manufacturing firms. Kampo medicine is a mixture of various medicinal plants used for treatment of various diseases (Watanabe *et al.*, 2011).

Outside of Asia, medicinal plants are essential in the Amazon, where native tribes use thousands of species for treatment, and throughout Africa, where traditional healers still play a significant role in healthcare. Worldwide, conservation initiatives like seed banks, botanical gardens, cultivation programs, and Good Agricultural and Collection Practices (GACP) strive to protect these essential resources against dangers from overharvesting, urban development, and climate change (FAO, 2010).

In Africa, generally medicinal plants serve as key sources of livelihood, with local and indigenous families depending on the herbal market for their income. From a medical perspective, plants such as *Artemisia annua* (the source of artemisinin), neem (*Azadirachta indica*), ginseng (*Panax ginseng*), turmeric (*Curcuma longa*), ginger (*Zingiber officinale*), and (*Schisandra chinensis*) remain significantly influential in contemporary pharmaceutical research, with their scientifically validated antimicrobial, anti-inflammatory, antimalarial, and antioxidant qualities (Baars *et al.*, 2019).

In Tanzania, traditional healthcare still heavily relies on medicinal plants, especially in biodiversity-rich example in the areas adjacent the Eastern Arc Mountains (EAM) like in Iringa, Tanga, and Morogoro, where various ecological settings host numerous therapeutic species (Mussa, 2024). These plants are commonly utilized by traditional healers and local families to address malaria, digestive issues, maternal health challenges, infections, and long-term ailments (Nahashon, 2013).

Lushoto and Muheza districts in Tanga Region, boasts a rich diversity of medicinal plants, cherished for their therapeutic properties by the local communities. However, there is scientific data with regards to the community perception on the availability and use of medicinal plants and; the willingness to conserve on-farm medicinal plants in EAM are still scant. Hence this study aimed to analysed: (i) Community perception on the availability and use of medicinal plants; and (ii) Willingness to conserve (WTCons) medicinal plants. The study provides information that will support implementation of the targets related to investment and marketing systems of non-wood forest products in the National Forest Policy Implementation Strategy of 2021 – 2030.

MATERIAL AND METHODOLOGY

Description of the Study Area

This study was conducted in West Usambara Mountains, Tanga region where two Districts were purposively selected (Fig.1). Muheza and Lushoto districts were selected for the purpose of this study. In each district two villages were randomly selected These villages were at Muheza District (Kisiwani and Mashewa villages) and; Lushoto District (Gologolo and Hambalawei villages) were selected. These villages were selected due to recent increase extraction of medicinal plants in and around the reserves that have a direct and indirect impact on the conservation of its Biodiversity.

Muheza District is located on the northeastern coast of Tanzania with total area of 1,974 square kilometers. The district has two distinct agro-ecological zones, highlands and lowlands based on agro-ecological factors including altitude and temperature. The lowland area, which extends between 5°10 S latitude and 38°47 E longitude, lies at an altitude of approximately 200–900 m a.s.l and has a temperature range of 25°C–32°C (Malagira *et al.*, 2025). Muheza District experiences two rainfall seasons, one of which takes around two to four months depending on a specific year and local weather patterns, and an average annual rainfall of 1,100 to 1,400 mm (Mbedule & Mungwabi, 2023). However, according to the 2022 Tanzania National Census, the population of Muheza was 204,461 people. In addition, 19 Villages where the majority of villagers are subsistence-level farmers, border ANR found in Muheza District. Farming for both food and cash is an important occupation of the majority of people living adjacent to ANR (Rehema and Alexander, 2013). The vegetation cover in the district ranges from coastal lowland forests and thickets to East Usambara Mountain forests, though large areas of the coastal plains have been converted into sisal and coconut plantations.

Lushoto District is situated in the Northern part of Tanga Region with a total area of 4092 km² lies between latitude 4°25 and 4° 55'S, and longitude 30° 10 and 38°35E. Lushoto District lies at an altitude ranging from 480-2,271m a.s.l. The climatic condition of the area is characterized by cool weather throughout the year. The mean annual rainfall and temperature vary with altitude. The mean annual rainfall is 600 - 2000 mm per annum. The mean annual temperature varies with altitude ranging from 17 to 18°C on the plateau at 1500 - 1800 m a.s.l. (Lawal *et.al*, 2019; Chove & Athumani, 2023; Kimaro & Chidodo, 2021). According to the 2022 Tanzania National Census, the population of Lushoto District was 492,441 people which is 24.7% of the Tanga Region. The Lushoto district of Tanzania boasts a rich diversity of medicinal plants, cherished

for their therapeutic properties by the local communities (Mussa, 2024). However, the main economic activity of villagers bordering with Magamba Nature Forest Reserve found in Lushoto District is crop production mostly food

crops including maize, beans and potatoes. The vegetation cover in the district is characterized by montane forests, farmlands and mixed vegetation mosaics that support both intensive agriculture and biodiversity conservation.

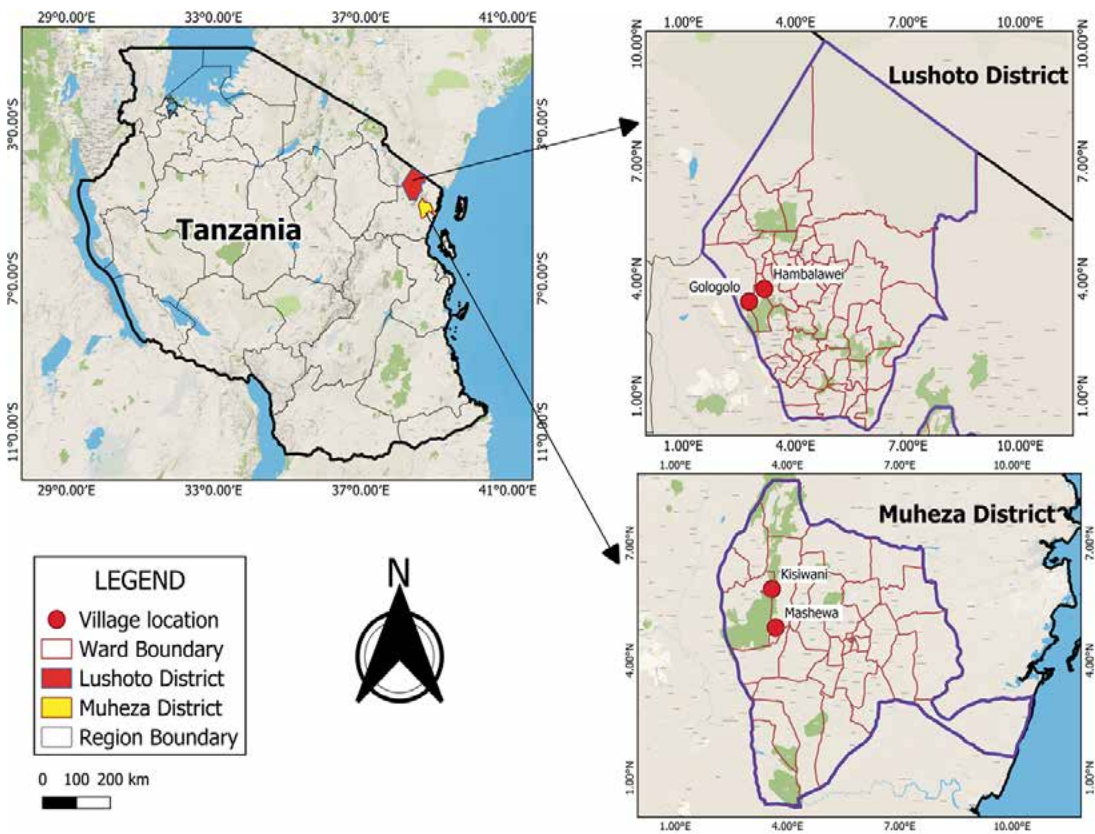


Figure 1: A map showing the study area (Source: Author GIS design, 2025)

Research Design and Sampling Strategy

This study employed a cross-sectional research approach that facilitates the simultaneous collection of data across different cases at once. A cross-sectional design is relevant when assessing the prevalence of disease or traits, attitudes and knowledge, in validation and in reliability studies (Kesmodel, 2018). In order to assess community willingness to conserve (WTCons) medicinal plant resources in the Eastern Arc Mountains (EAM), Tanzania, as well as the availability and use of medicinal plants, the study used a mixed-methods research approach. The methodology

combined quantitative household surveys with qualitative methods, such as focus groups, key informant interviews, and field observations, to gather both quantifiable trends and in-depth community knowledge. In addition to recording indigenous knowledge systems and perceived risks to medicinal plant resources, this method made it possible to evaluate the availability, use, and conservation attitudes of medicinal plants across various socioeconomic classes. In addition to providing a thorough foundation for guiding sustainable conservation and management methods in the EAM, the triangulation of several data sources improved the validity and robustness of the findings.

Data Collection Methods

Household questionnaire surveys

A household questionnaire surveys were used to gather quantitative data on the availability and use of medicinal plants as well as to gauge household-level willingness to conserve (WTCons) medicinal plant resources. Household heads or adult representatives were given structured questionnaires that addressed important topics such as the kinds of medicinal plants that are available, where they are collected, how often and why they are used, perceived trends in their availability, and current conservation measures. The questionnaires also included socioeconomic traits and perspectives on the preservation of medicinal plants, allowing for the examination of variables impacting WTCons. Standardized data collection across study sites was made possible by the use of household surveys, which reflected local knowledge and practices about the use and conservation of medicinal plants while also making comparability and statistical analysis of responses easier.

Focus group discussion (FGDs)

Focus group discussions (FGDs) were conducted to obtain in-depth qualitative insights into community knowledge, perceptions, and collective experiences regarding the availability, use, and conservation of medicinal plants in the Eastern Arc Mountains (EAM). The discussions involved purposively selected participants, including traditional healers, elders, women, and youth, to ensure representation of diverse knowledge systems and user groups. The focus group discussions were comprised of 10 people aged more than 30 years with knowledge and experience on the Improvement of livelihoods change issues caused by on-farm medicinal plants. Probe questions were used to guide collection of data from discussants who are knowledgeable about the status of on-farm medicinal plants and their contribution to livelihoods in villages studied (Balama *et al*, 2017). Guided by a

semi-structured checklist, the FGDs explored themes such as changes in medicinal plant availability, harvesting practices, cultural significance, perceived threats, and community willingness to conserve (WTCons) medicinal plant resources. The interactive nature of FGDs facilitated the exchange of ideas and consensus-building among participants, enriching the understanding of social norms and local perspectives that influence medicinal plant use and conservation in the EAM.

Key informant interviews (KII)

In order to gather expert and contextual information regarding the perceptions on availability, use, and conservation of medicinal plants in the Eastern Arc Mountains (EAM), key informant interviews (KIIs) were carried out. Traditional healers, forestry and conservation officers, village leaders, extension officers, and conservation practitioners with in-depth knowledge of medicinal plant resources and regional management techniques were among the purposefully chosen key informants. The KIIs investigated topics such trends in the availability of medicinal plants, harvesting and management techniques, conservation difficulties, institutional and policy frameworks, and elements influencing community willingness to conserve (WTCons) medicinal plants using a semi-structured interview guide. Key informant interviews strengthened the overall understanding of the study findings by supplementing household surveys and focus group discussions with authoritative viewpoints and community-level data validation.

Data Analysis

Descriptive and inferential statistics on the availability, use, and willingness to conserve (WTCons) medicinal plants in the Eastern Arc Mountains (EAM) were analysed. Household characteristics, medicinal plant use patterns, and perceived availability trends were summarized using descriptive statistics, such as frequencies, percentages,

means, and standard deviations. Relationships between socioeconomic characteristics and WTCons were investigated using inferential methods, such as logistic regression and chi-square testing. To find recurrent themes on indigenous knowledge, conservation issues, and community attitudes, qualitative data from focus group discussions and key informant interviews were transcribed, categorized, and subjected to thematic analysis. The strength and validity of the study’s conclusions were improved by the triangulation of quantitative and qualitative data

RESULTS AND DISCUSSION

Socio-Economic Characteristics of the Respondents

Table 1 shows the analytical results of the socio-economic and demographic results of the respondents in the study area. Results indicate male-headed, married dominance, with primary education being the most common qualification of respondents. Most of the respondents were farmers cultivating various crops such as maize, beans, banana, vegetables, spices, black pepper, as well as animal husbandry - common agricultural practices in rural Tanzania

(Kibonde, 2020; Mapunda and Mramba, 2025). Mean age of the respondents was adult age with respect to Tanzania age categories, while duration of household has a mean of above forty (40) years. This implies enough time for households to transfer MPs knowledge from one generation to another. Focus group discussions (FGDs) confirm that there is inter-generational transfer of MPs knowledge from one generation to another. Many youths reported trusting most modern medicines compared to adult ones, as it was found by Mapunda and Mramba, (2025), but differ from earlier findings by Otieno *et al.* (2007). Primary education of the respondents implies the educational category of people who engage in MPs harvesting, processing, use and commercialization. This complies with findings by Kibonde (2020). Primary source of income other than farming reveals that sales of medicinal plants have got impact to livelihood despite being at a low rate (6.3%). This reveals that MPs value chain is not well explored. Traditional medicines sales in urban areas have gained great momentum while its commercialization in rural area is weak (Mapunda and Mramba, 2025). According to FGD, there is no community initiative developed for MPs commercialization promotion, management and conservation.

Table 1: Socio-economic and demographic characteristics of the respondents (n=128)

Variable	Description	Frequency	Percent
Gender	Male	100	78.1
	Female	28	21.9
Position in the Household	Head	110	85.9
	Spouse	12	10.9
	Son	4	3.1
Marital Status	Married	108	85.8
	Single	12	10.4
	Widow	8	3.1
Educational level	None	6	4.7
	Primary School	104	81.3
	Secondary school	18	14.1
Occupation	Farmer	118	92.2
	Driver of motorbike	8	6.3

	Farmer and Entrepreneur	2	1.6
Primary source of income Crop cultivation	1	124	96.9
Primary source of income Sales of medicinal plants	1	8	6.3
Primary source of income motorbike driver(bodaboda)	1	6	4.7
Mean values			
	n	Mean	Std. Deviation
Age (Years)	128	54.06	14.253
Household duration (Years)	128	40.7	18.944

Communities' Perception of Medicinal Plants Availability and Use

Table 2 shows the results conducted by cross-tabulation of the probable effect of various socio-economic variables on the perception of the respondents regarding the availability and use of MPs in the study area. The results reveal that gender, age, household duration, crop cultivation and motorcycle transport income sources significantly influence community's perception of MPs availability and use in the study area. This implies that increasing these socio-economic variables necessarily enhances community perception towards availability and use of medicinal plants in the study area.

Education and marital status of the respondents did not significantly influence community perception of MPs availability and use. These results concur with findings by Rangel de Almeida *et al.* (2025), who observed that cultural and ecological familiarity were strong predictors of medicinal plants perception and knowledge rather than education. However, the findings differ from those reported by Kideghesho and Msuya (2010) who found that education significantly influences the perception and domestication of indigenous medicinal plants in Northern Tanzania. According to FGD, traditional medicine involvement does not consider demographic properties of the respondents, but rather the self-interest.

Table 2: Socio-economic and demographic factors on the respondents' perception of the availability and use of MPs (n=124)

Variable	Description	No	Yes	χ^2 value
Gender	Male	10	98	3.037*
	Female	0	28	
Position in the Household	Head	10	100	0.412
	Spouse	0	14	
	Son	0	4	
Marital Status	Married	10	98	0.366
	Single	0	12	
	Widow	0	8	
Age	Years	10	118	77.089***
Household duration	Years	10	118	65.056***

Variable	Description	No	Yes	χ^2 value
Educational level	None	0	6	0.783
	Primary School	8	96	
	Secondary school	2	16	
Occupation	Farmer	8	110	0.163
	Driver of motorbike	2	6	
	Farmer and Entrepreneur	0	2	
Income source	Crop cultivation	8	116	10.204***
	Sales of MPs	0	8	
	Motorcycle transport	2	4	

Community's perception towards Medicinal Plants availability and use

Model analysis results through Generalized Linear Model (GLM) for respondents' perceptions regarding MPs conservation, highlight that, collection of medicinal plants from on-farm, use of MPs to treat wounds, injuries and challenges in accessing or harvesting medicinal plants both significantly influence the availability and use of medicinal plants in the study area after Bootstrap (Table 4). Collection of medicinal plants from on-farm ($\beta = -1.426$, $p = 0.100$, $\chi^2 = 2.652$, $df = 1$), extraction of MPs roots treating ailments ($\beta = -1.603$, $p = 0.086$, $\chi^2 = 2.954$, $df = 1$) and application of MPs for wound and injuries treatments significantly influence negatively ($\beta = -1.124$, $\chi^2 = 5.374$, $p = 0.02$, $df = 1$) the community' perception while challenges in accessing or harvesting medicinal plants ($\beta = 0.917$, $\chi^2 = 3.812$, $p = 0.051$, $df = 1$) variables influence positively perception on the availability and use of medicinal plants in the Eastern Arc Mountains. These findings are consistent with ethnobotanical studies in Tanzania, such as Kayombo *et al.* (2013),

which show that medicinal plant harvesting, particularly through root extraction, reduces both resources and confidence in MPs as alternative medicines. Similar studies in South Africa reported that intensive use of medicinal plants for wound healing led to overutilization and reduced trust in MPs availability (Mabogo, 2012; Dold and Cocks, 2002). Our observation that challenges in harvesting or accessing MPs in the study align with evidence from Kenya where MPs scarcity raised awareness of conservation needs (Tabuti *et al.*, 2003). But these findings differ from those reported in Ethiopia and Northern Tanzania, where cultivation of MPs improves perception of sustainability and reliability (Lulekal, 2008; Otieno *et al.*, 2015). According to FGD, the study area has frequent incidence of malaria and injuries (since most respondents were farmers), so MPs are served for these ailments, of which treatment confidence perception is very high. The majority of the respondents depend entirely on MPs to treat ailments since they are conversant with quantity, quality and combination they understood from ancestors (intergenerational knowledge transfer).

Table 4: Determinants of the community’s perception towards MPs availability and use in the study area, n=128

Parameter	B	Std. Error	95% Wald Confidence Interval		Hypothesis Test			Collinearity Statistics	
			Lower	Upper	Wald Chi-Square	df	Sig.	Tolerance	VIF
(Intercept)	0.342	1.4807	-2.56	3.245	0.053	1	0.817		
Availability of MPs	-0.41	0.6133	-1.612	0.792	0.447	1	0.504	0.892	1.121
How obtained_obtained from private farms	-1.426	0.8754	-3.141	0.29	2.652	1	0.100	0.702	1.425
How obtained_Collected from the wild	0.943	0.5984	-0.23	2.116	2.483	1	0.115	0.746	1.341
Abundance season_Spring season	0.362	0.8295	-1.264	1.988	0.19	1	0.663	0.885	1.13
MPs availability of changes over the past few years	0.413	0.5765	-0.717	1.542	0.512	1	0.474	0.78	1.283
Challenges in accessing or harvesting medicinal plants	0.917	0.4697	-0.004	1.838	3.812	1	0.051*	0.923	1.084
Type of ailments_Malaria	1.064	0.967	-0.831	2.959	1.21	1	0.271	0.683	1.465
Type of ailments_Wounds and injuries	-1.124	0.4851	-2.075	-0.174	5.374	1	0.02**	0.705	1.419
Parts of plants mostly used_Roots	-1.603	0.9324	-3.43	0.225	2.954	1	0.086*	0.834	1.199
<i>Omnibus Test</i>									
Likelihood Ratio Chi-Square	df	Sig.							
28.001	9	0.001							
<i>Goodness of Fit</i>									
	Value	df	Value/df						
Pearson Chi-Square	84.696	19	4.458						

Note: ** p<0.05, * p<0.1 significant levels. Source: Author’s Survey, 2025.

Communities’ Willingness to Conserve Medicinal Plants

Cross tabulation was employed to examine the relationship between socio-economic variables with the WTCons of medicinal plants in the study area (Table 3). Results reveal that age of ($\chi^2 = 50.268, p=0.088$) and gender ($\chi^2 = 3.708, p=0.054$) of the respondents significantly influence WTCons medicinal plants. This implies that adult males are the group who are eager to conserve MPs in the study area. This finding aligns with those of Mapunda and Mramba (2025), who observed that adults tend to value traditional medicinal knowledge deeply and are more committed to its preservation. Also, Kibonde (2020) similarly reported

that elders in rural communities are often the custodians of indigenous traditional medicinal knowledge and are key players in conservation practices of medicinal plants. Additionally, Kideghesho and Msuya (2010) reported that gender and socio-economic factors strongly affect domestication and conservation of MPs in West Usambara Mountains. However, this differ with findings by Otieno *et al.*, (2007), who reported youth active engagement in some regions, particularly where traditional knowledge is integrated into formal education or initiatives. According to FGD, when youths cultivate their farms, they do not conserve MPs on-farm unless they understand their socio-economic importance.

Table 3: Effect of socio-economic variables on the WTCons medicinal plants

Variable	Description	Response (%)		χ^2 value
		No	Yes	
Gender	Male	12	88	3.708*
	Female	0	14	
Position in the Household	Head	6	49	1.083
	Spouse	0	7	
	Son	0	2	
Age		6	58	50.268*
Household duration		6	58	43.402
Marital Status	Married	6	48	1.226
	Single	0	6	
	Widow	0	4	
Sex	Male	6	47	1.374
	Female	0	11	
Educational level	None	0	3	2.232
	Primary School	4	48	
	Secondary school	2	7	
Occupation	Farmer	5	54	1.309
	Driver of a motorbike	1	3	
	Farmer and Entrepreneur	0	1	

* p<0.1 significant levels

Willingness-to- conserve (WTCons) medicinal plants in the study area

Generalized Linear Model (GLM) Probit analysis for respondents' willingness to MPs conservation, highlights that urbanization as a threat to medicinal plants availability, age and household duration of the respondents significantly influence willingness to conserve medicinal plants in the study area (Table 5). Age of the respondents ($\chi^2 = 3.659$, $p=0.056$) and household duration of the respondents ($\chi^2 = 2.308$, $p=0.094$) with urbanization as a threat to medicinal plants availability ($\chi^2 = 6.516$, $p=0.010$), influence positively WTCons

in the Eastern Arc Mountains. These findings are similar to those reported by Kayombo *et al.* (2013), that long-term residency supports deeper ecological stewardship and knowledge, which enhances WTCons. Similarly, earlier findings by Kideghesho and Msuya (2010) noted that older individuals were more likely to engage in MPs conservation due to knowledge retention and cultural attachment. According to FGD and KII, formerly before urbanization advancement, MPs were plentifully available and the accessibility was simple. Also, youths do not consider much intergenerational MPs knowledge transfer and conservation.

Table 5. Factors influencing community willingness-to-conserve (WTCons) medicinal plants in the study area; n=128

Parameter	B	Std. Error	95% Wald Confidence Interval		Hypothesis Test			95% Wald Confidence Interval for Exp(B)		Collinearity Statistics		
			Lower	Upper	Wald χ^2	df	Sig.	Exp (B)	Lower	Upper	Tolerance	VIF
(Intercept)	-2.227	1.4665	-5.101	0.647	2.307	1	0.129	0.108	0.006	1.91	0.868	1.152
Confidence in MPS - very confident	0.279	0.478	-0.658	1.215	0.34	1	0.56	1.321	0.518	3.372	0.918	1.089
Side effects - no side effects	0.604	0.5634	-0.5	1.708	1.15	1	0.284	1.83	0.606	5.52	0.621	1.61
Threat to the availability of MPs_Urbanization	1.105	0.433	0.257	1.954	6.516	1	0.010***	3.02	1.293	7.056	0.545	1.835
Age	-0.039	0.0204	-0.079	0.001	3.659	1	0.056*	0.962	0.924	1.001	0.504	1.984
Household duration	0.025	0.0167	-0.007	0.058	2.308	1	0.094*	1.026	0.993	1.06	0.837	1.194
Support encourages participation_Training	-0.038	0.4863	-0.991	0.916	0.006	1	0.938	0.963	0.371	2.498	0.92	1.087
Specific conservation method in place to protect medicinal plants	0.202	0.3924	-0.567	0.971	0.265	1	0.606	1.224	0.567	2.641	0.835	1.198

Dependent Variable: Would you willing to participate in community efforts to conserve medicinal plants?

Goodness of Fit

	<i>Value</i>	<i>df</i>	<i>Value/df</i>
Deviance	65.696	52	1.263
Pearson Chi-Square	101.084	52	1.944

Omnibus Test

Likelihood Ratio Chi-Square	df	Sig.
13.953	7	0.052

Dependent Variable: Would you willing to participate in community efforts to conserve medicinal plants? Note: *** p<0.01, * p<0.01 significant levels. Source: Author's Survey, 2025.

CONCLUSION AND RECOMMENDATIONS

This study highlights that socioeconomic variables such as position in the household, marital status, occupation and education influence community perception of MPs availability and use. Perception of community on the availability and use of medicinal plants is shaped by more practical realities, which are on-farm collection of MPs, root extraction of MPs, treatment of wounds and injuries, and challenges in obtaining and harvesting MPs, than demographic profile. Nevertheless, Willingness to conserve MPs is significantly influenced by age, urbanization, and household duration. Older residents of the study area show less inclination to participate in conservation efforts (as the sample had few young respondents). Based on the study findings, the following are recommended: Various interventions on medicinal plants conservation should target younger populations with culturally relevant education programs; community-based conservation should be strengthened in areas facing urbanization threats by promoting local stewardship programs in peri-urban zones; and there is a need for a further research to explore the dimensions of cultural beliefs and ecological knowledge since variables such as trust in traditional healers and community beliefs showed practical relevance.

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USING NORMALIZED DIFFERENCE VEGETATION INDEX AND LAND SURFACE TEMPERATURE ANALYSIS SUPPORT SUSTAINABLE MANAGEMENT OF RIVERINE FORESTS IN SUDAN?

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ABSTRACT

Natural riverine forests in Sudan support diverse plant and animal communities and provide multiple ecological and socio-economic services. Despite their importance, limited information exist regarding their composition and spatio-temporal dynamics under climatic variability and human disturbance. This study assessed land cover changes and population dynamics of *Acacia nilotica* in the Remaila Riverine Forest Reserve using Normalized Difference Vegetation Index (NDVI), Land Surface Temperature (LST) analysis, and ground inventory data. Results indicated a significant decline in vegetation cover and poor natural regeneration, characterized by limited juvenile trees and reduced tree density. Areas near settlements showed clear signs of illegal harvesting, overgrazing and soil disturbance. LST values were lower in dense forest areas and high elevations. The findings highlight the urgent need for conservation planning and restoration strategies to sustain riverine forest ecosystems while considering local community needs.

Keywords: Biodiversity; forest reserve; habitat conservation; NDVI; wetlands

INTRODUCTION

Remarkable landscapes found along riverbanks and floodplains hosting diverse plant species, bird, and aquatic species with multiple ecological and socio-economic functions and services (Gurashi *et al.*, 2024). They maintain biodiversity (Rajpar *et al.*, 2024), protect soils (Ibrahim *et al.*, 2018; Zhang *et al.*, 2023), regulate nutrients cycling (Biswas & Mallik, 2010; Hassaballah *et al.*, 2020), and enhance local community livelihoods (Ibrahim *et al.*, 2018). The common tree species usually observed in riverine forests worldwide include *Acacia nilotica* (Alzubair & Hamdan, 2020; Khalaf *et al.*, 2023), *Ceiba pentandra* (Miranda Egbe *et al.*, 2021), *Eucalyptus maidenii* (Kingazi *et al.*, 2020), *Khaya senegalensis* (Gaoue & Ticktin, 2008), and *Salix spp* (Weih

et al., 2021). These species are well-adapted to varied water, saturated soil, and prolonged immersion (Khalaf *et al.*, 2023; Pielech, 2021). Factually, the natural riverine forests in Sudan are distributed along the Blue and White Niles and their tributaries with various ecosystem services and forest products (Hagr *et al.*, 2025). They support the forest-based communities and other marginalized areas through the provision of food, fodder, medicinal materials, and timber (Ewunetu *et al.*, 2021; Hagr *et al.*, 2025). Besides that, riverine forests sequester carbon dioxide (Ibrahim *et al.*, 2018), offer habitats to wild animals (Hassaballah *et al.*, 2020), maintain the water sources (Rajpar *et al.*, 2024), and serve as recreation sites (Mohammed *et al.*, 2025). Despite all these functions and services provided by the riverine forests in Sudan, information is limited regarding their yield

regulation, composition, and sustainability. To bridge such gap, application of remote sensing and land cover change analysis supported by machine learning techniques is significant.

Although NDVI, LST, and NDWI are widely used techniques at regional and global scales to assess forest health and monitor its diversity (Belenok *et al.*, 2021; Mohiuddin & Mund, 2024; Yuan *et al.*, 2023), they have rarely been used in Sudan. Therefore, this study aimed to assess the spatio-temporal dynamics of riverine forests in Remaila Riverine Forest Reserve over the past twenty years using NDVI, LST, and field inventory data.. Specifically, the research will explore the dynamics of land use and land cover (LULC) changes using NDVI, LST, and forest inventory data, which are frequently recommended for such studies (Dammag *et al.*, 2024; Espindola *et al.*, 2025). This will highlight hotspots of forest degradation in relation to human activities and climate change to provide a basis for further sustainable management of these last remaining fragile forests.

MATERIALS AND METHODS

Study area

The study was performed in RFR that located near Khashm Elgeirba dam between latitudes $14^{\circ} 50' 30''$ N to $14^{\circ} 55' 00''$ N and longitudes $35^{\circ} 53' 00''$ E to $35^{\circ} 56' 00''$ E with a total area of 1,157 acres (Fig. 1). The reserve accommodates considerable populations of migratory birds, aquatic organisms, tree species, and various perennial plants with significant environmental roles and conservation services (Gurashi *et al.*, 2024; Hagr *et al.*, 2025). The topography of RRRF is flat to semi-flat with average temperatures of 35°C to 45°C . Furthermore, the forest represents the main source of timber, fodder, traditional medicine, fishing, recreation, and site for public events for the local communities and the region's visitors. Three villages including Remaila, Monabaa, and Dewahi neighbor the forest. The common income-generating activities of local people around the reserve include agriculture, livestock keeping, charcoal production, fishing, trading, and traditional medicine.

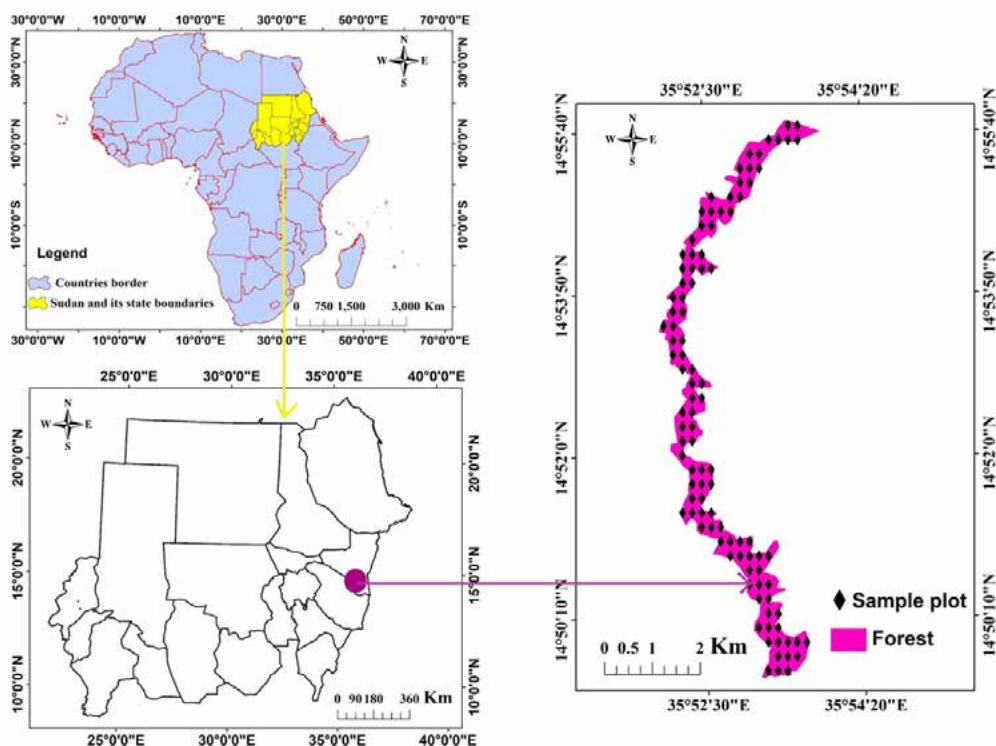


Figure. 1: Study area Map

Data collection

Ground measurement and inventory data

To collect the inventory data, we systematically divided RFR into 39 transects with 102 sample plots of 1000 m² (Fig. 1). The horizontal distance between adjacent sample plots and parallel transect lines was 200m and 300m, respectively. Inside each single sample plot, we identified the tree species, measured the tree diameter, height, and crown width. While tree diameter was measured at breast height (DBH), height was assessed as a total tree height from the tree base up to last growing shoot (Mohammed *et al.*, 2022).

Remote sensing data and image processing

We downloaded a cloud free Landsat images for 2003, 2013, and 2023 from the US Geological Survey (USGS) using Earth Explorer portal as recommended by other researchers (Czerkowski *et al.*, 2023; Dammag *et al.*, 2024). The further Land Use/Land Cover (LULC), Normalized Difference Vegetation Index (NDVI), and Land Surface Temperature (LST) maps were developed based on this data using ArcGIS and ArcMap. The variation of spatial resolutions among different years was fixed using ArcGIS through resampling approach (Hagr *et al.*, 2025).

Data analysis

Ground measurement and inventory data

The study performed ANOVA analysis in JAMOV I to figure out the average values of dendrometric parameters and stand characteristics of mature trees, and compare them across the study area. The tree density, basal area, and volume were computed using equations 1,2, and 3 as supported by references (Assogbadjo *et al.*, 2010; Mohammed *et al.*, 2022).

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$$\text{Density (stem/ha)} = \text{number of stems per ha} \dots\dots\dots (1)$$

$$\text{Basal Area (m}^2\text{)} = \frac{\pi \times \text{DBH}^2}{4} \dots\dots\dots (2)$$

$$\text{Volume (m}^3\text{)} = \text{Basal area} * \text{Total height} * \text{Form factor of } \textit{Acacia nilotica} \dots\dots\dots (3)$$

Remote sensing and image processed data

For LULC analysis, the study utilized the national classification framework to distinguish between the forest sites and other ones, followed by utilization of ESRI 2020 Global LULC framework (FNC, 2021; Hagr *et al.*, 2025). Adopting the methodologies applied by (El fallah *et al.*, 2024; Hagr *et al.*, 2025), the forest was classified into dense, moderate, and open forests with distinct bare and water body classes. The supervised classification was supported by field observation and forest inventory data. Furthermore, for NDVI and LST analysis, the images were preprocessed for geometric correction and radiometric consistency. Due to their availability in surface reflectance and surface temperature formats, the images were not subject to advanced atmospheric correction (Ahmed *et al.*, 2024; Hagr *et al.*, 2025). The NDVI and LST were computed for 2003, 2013, and 2023 using the Table 1 equations. Additionally, the study checked for variation in NDVI and LST applying the non-parametric Kruskal-wallis (Ahmed *et al.*, 2024; Hagr *et al.*, 2025).

Table 1: Description of equations adopted from (Aka *et al.*, 2023; Hagr *et al.*, 2025) for calculation of NDVI and LST for Landsat images

Item	Applied equation
Normalized Difference Vegetation Index (NDVI)	• $NDVI = \{[Near\ Infrared\ Band\ (NIB) - Red\ Band\ (RB)] \div [NIB + RB]\}$ • Landsat 7 Images = $\{[Band\ 4\ (B4) - Band\ 3\ (B3)] \div [B4 + B3]\}$ • Landsat 8 and 9 images = $\{[Band\ 5(B5) - B4] \div [B5 + B4]\}$
Land Surface Temperature (LST, Landsat 7)	• Digital Number (DN) to radiance (Kelvin) = $[SR\ max - SR\ min] \div [255 * (DN + SR\ min)]$ (Deng <i>et al.</i>, 2018; Kumar & Shekhar, 2015) (Deng <i>et al.</i>, 2018; Kumar & Shekhar, 2015) (Deng <i>et al.</i>, 2018; Kumar & Shekhar, 2015) (Deng <i>et al.</i>, 2018; Kumar & Shekhar, 2015) (Deng <i>et al.</i>, 2018; Kumar & Shekhar, 2015) • Radiance to brightness temperature (kelvin) = $1,260.56 \div [\ln(607.67 \div Radiance\ values) + 1]$ • Land Surface Emissivity (LSE) = $1.0094 + 0.047 * \ln(NDVI)$ • Kelvin (k) to Celsius (C) = Temperature in kelvin – 273.15
Land Surface Temperature (LST, Landsat 8 & 9)	• Top of Atmosphere spectral radiance (ToA) = $(RMB * DN) + RAB - correction\ value$ • ToA to brightness temperature ($^{\circ}C$) = $1,260.56 \div \{[\ln(607.67)] \div [(ToA\ spectral\ radiance + 1) - 273.15]\}$ • Proportion of Vegetation (PV) = $[(NDVI - NDVI_{min}) \div (NDVI_{max} - NDVI_{min})]^2$ • Land Surface Emissivity = $0.004 * PV * 0.986$ • LST = $BT \div [1 + (\lambda * BT / c2) * \ln(E)]$

Where, SR max is the Spectral Radiance of Maximum value, SR min is Spectral Radiance of Minimum value, 1,260.56 is the calibration constant1, 607.67 is the calibration constant2, RMB is Radiance Multiplicative Band, RAB is Radiance Add Band, BT is the Brightness Temperature ($^{\circ}C$), λ is the wavelength f emitted radiance (for band 10 is 10.8 μm). 0.986 is corresponds to a correction value of the equation.

RESULTS AND DISCUSSION

Land cover change, LST, and NDVI dynamics

Land cover change analysis illustrated that the areas of dense forest and water decreased with significant degradation near local settlements (Fig. 2). Over 50% of dense forest was lost showing a high proportion of bare lands and degraded sites (Fig. 2). Though literature highlighted similar conclusions in the region (Ahmed *et al.*, 2024; Mohiuddin & Mund, 2024), the current decline underlined the

combined effect of anthropogenic activities and climatic variability. Limited rainfall and extreme temperatures eliminate young trees and drought sensitive species with dramatic impact on forest composition and stand structure (Mohammed *et al.*, 2022; Musa *et al.*, 2023). Moreover, human disturbances reduce species diversity and accelerate habitat fragmentation (Beche *et al.*, 2016; John *et al.*, 2020).

The LST findings display that western part of RFR have higher temperatures compared to eastern and northern ones with lower

vegetation cover and large bare lands (Fig. 2 and 3). The average temperature increased dramatically from 2003 to 2023 with notable raise in 2023 (Fig. 3), which is consistent with regional trends (Hagr *et al.*, 2025; Minière *et al.*, 2023). This observed increase in surface temperature can be directly attributed to agricultural expansion and other domestic land uses including settlements and animal husbandry (Mohammed *et al.*, 2022; Musa *et al.*, 2023). Such patterns can severely influence the ecosystem services and local community livelihoods resulting in limited income and more pressure on forest resources (Liu *et al.*, 2021; Sandeep *et al.*, 2021; Song *et al.*, 2021).

The NDVI values temporarily showed a stable pattern in 2003 and 2013 with notable decline in 2023, particularly in the southern part of the reserve, with NDVI values close to zero (Fig. 4). The areas near human settlement in the western and southern parts exhibited lower NDVI values compared to northern ones (Fig. 4). The detected slight decline in vegetation cover and forest health raised an alarm for further intervention and monitoring to avoid a severe degradation of forest resources and tree species diversity. Such reduction directly attributed to

unauthorized harvesting and severe livestock rearing that affect both natural regeneration and mature trees. Literature documented that over utilization, intensive grazing, severe browsing, and crown debranching reduce tree density, stand population, seed production, and sapling recruitment (Alzubair & Hamdan, 2020; Derebe & Girma, 2020; Eltahir *et al.*, 2025; Valdes *et al.*, 2025). Similar trends were observed in Burkina Faso, Tanzania, and Ethiopia (Bondé *et al.*, 2019; Dimobe *et al.*, 2019; Ewunetu *et al.*, 2021; Kingazi *et al.*, 2020; Mpondo *et al.*, 2023). These findings call for restoration programs that will rehabilitate the degraded sites and severely disturbed ones.

The perceived intensive browsing pressure and wood gathering processes in Remaila forest supported by limited vegetation cover and unmanaged utilization highlights the significance of introducing a restoration package that rehabilitates the degraded and open lands of the reserve and enhance its growing stock and regeneration capacity. However, due to diversified stakeholders benefiting from the reserve, initiating new private and community forests can reduce the ongoing pressure on RFR and enrich its sustainability.

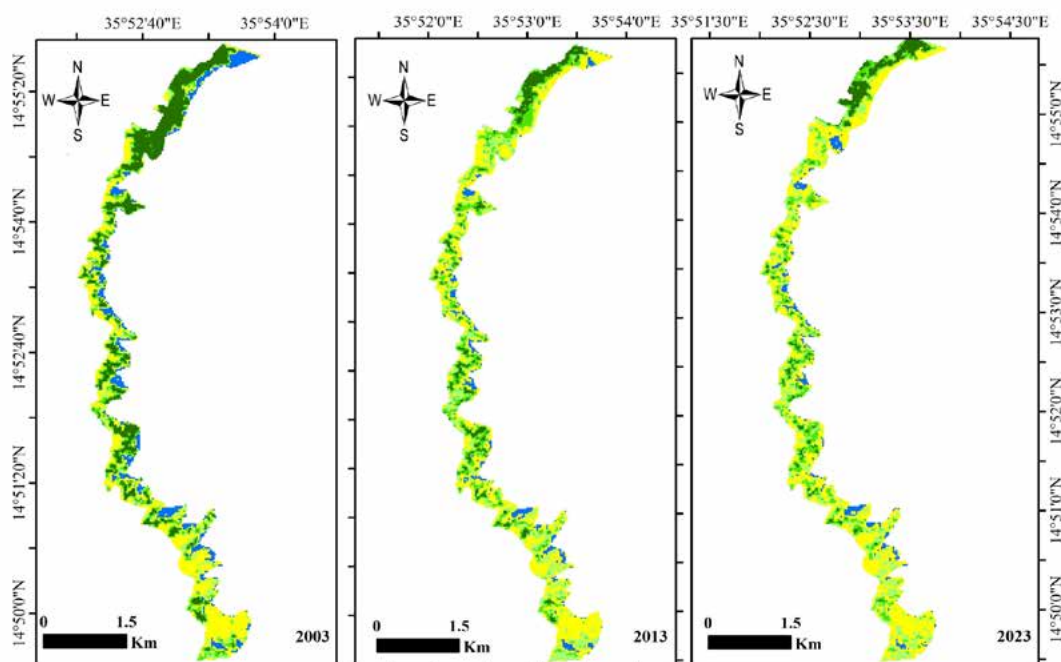


Fig. 2 Land Use Land Cover (LULC) maps for 2003, 2013, and 2023.

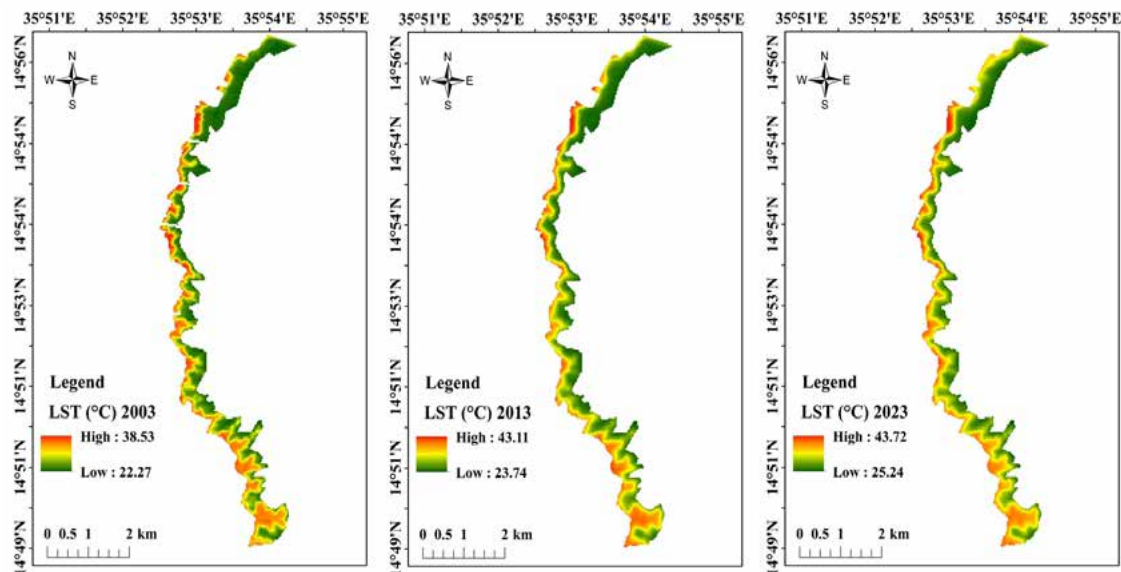


Fig. 3. Land Surface Temperature (LST) maps for 2003, 2013, and 2023.

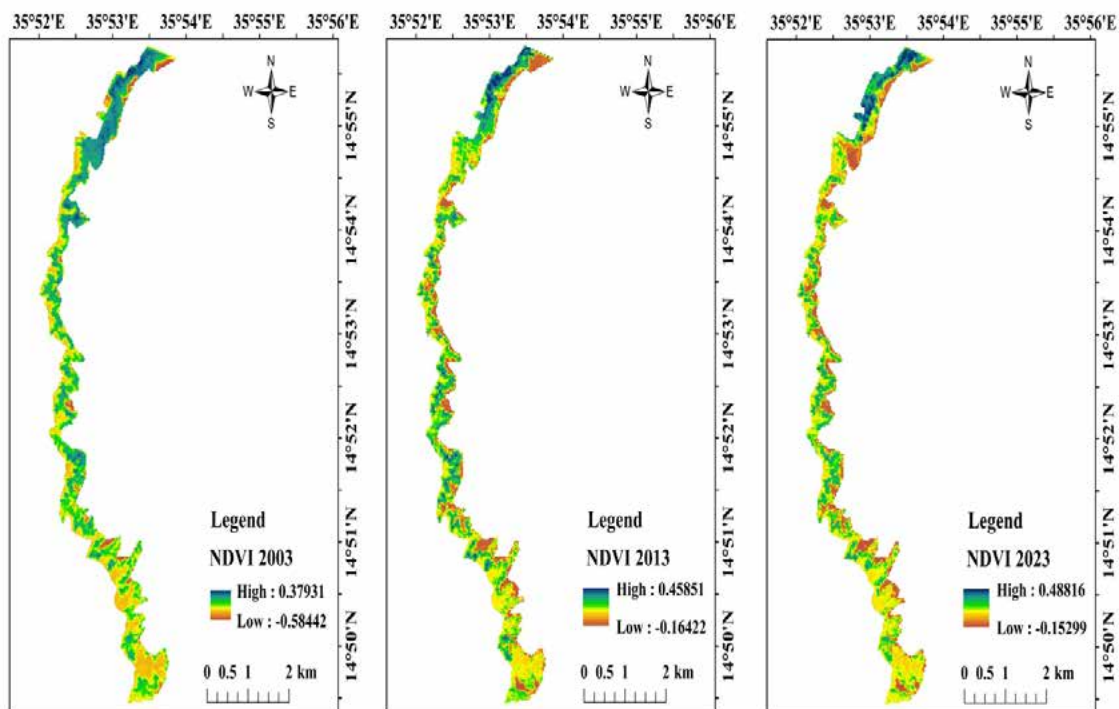


Fig. 4: Normalized Difference Vegetation Index maps (NDVI) maps for 2003, 2013, and 2023.

Density and volume of *Acacia nilotica* trees based on the ground measurement

The outcomes showed that open forest sites have smaller tree density and volume compared to dense ones (Fig. 5). While the tree density in dense forest sites was three times that of open forests, the tree volume in dense forest sites was double that of moderate forests with significant differences across the sites (Fig. 5). Although the limited volume of adult trees in open forest sites can result from their small stem diameter and height (Ibrahim *et al.*, 2018; Mohammed *et al.*, 2022), the limited tree

density might be due to the extensive harvesting of adult trees and rigorous pollarding activities of local communities neighboring the forest (Mohammed *et al.*, 2023; Sahoo & Musa, 2023). These findings support the NDVI and LST results, and in line with other studies accomplished in similar environments (Ahmed *et al.*, 2024; Hagr *et al.*, 2025; Lameck *et al.*, 2025). However, the inadequate density of adult and over-story trees in Remaila riverine forest reserve indicate signs of high human pressure and explains why the reserve attains low regeneration.

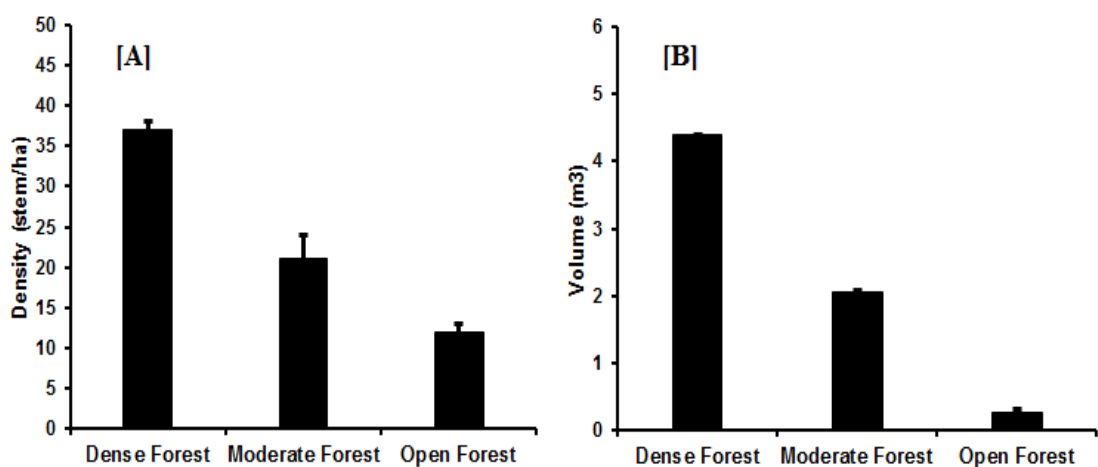


Fig. 5: The average [A] Tree density (stem/ha) and [B] Tree volume (m³) measured in RFR.

CONCLUSION

The study demonstrated the effectiveness of NDVI and LST analysis combined with field inventory data for monitoring vegetation dynamics in riverine forests. Significant decline in vegetation cover and poor regeneration of *Acacia nilotica* were detected, particularly in areas near human settlements. These changes are associated with increased human pressure, grazing and illegal harvesting. The findings highlight the urgent need for restoration programs, improved forest governance, and community-based conservation initiatives to ensure the sustainability of riverine forest ecosystems.

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THE LAND USE AND LAND COVER CHANGES AT MINJA FOREST RESERVE USING REMOTE SENSING TECHNIQUES, 1995–2025

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ABSTRACT

This study examined land use and land cover changes (LULCC) at Minja Forest Reserve (MFR) using remote sensing techniques. LULCC are among the major environmental drivers affecting biodiversity and climate systems at global, regional, and local scales. The study applied a mixed approach combining quantitative data from satellite imagery with qualitative data from ground truthing through field observation surveys. Moderate-resolution (30 m) Landsat images from 1995, 2005, 2015, and 2025 dry seasons were analysed and classified using Google Earth Engine (GEE) and ArcGIS Pro version 2.8.29 to assess spatial and temporal changes in dense forest, bare land, shrubland, and water bodies. The findings revealed significant changes in land cover classes, particularly an increase in dense forest from 64 hectares (ha) in 1995 to 291.79 ha in 2015, followed by a sharp decline to 87.94 ha by 2025. Bare land, which dominated at 281.62 ha in 1995, decreased to 59.86 ha by 2015 but increased again to 120.16 ha in 2025. Shrubland fluctuated, reaching 311.87 ha (59.7%) by 2025. Water bodies declined consistently from 3.7 ha in 1995 to 0.18 ha in 2025. Key drivers identified through field survey include logging (40%), livestock grazing (20%), water infrastructure installations (20%), farmland expansion (10%), and fire (10%). These findings provide valuable insights to support decision-making, forest policy improvements, and sustainable resource use at MFR.

Keywords: Minja Forest Reserve; biodiversity; drivers; land cover; remote sensing

INTRODUCTION

Changes in land use and land cover (LULC), which have an impact on ecosystem services, biodiversity, and climatic patterns, are important indicators of environmental change (Turner *et al.*, 2007). In Tanzania, the main causes of these changes include logging activities, charcoal manufacturing, and agricultural expansion, which result in habitat loss and deforestation (Msofe *et al.*, 2019). For well-informed land management and conservation initiatives, it is imperative to monitor such changes and trends. Satellite imagery and other remote sensing technologies have demonstrated efficiency in identifying and evaluating LULC changes, yielding useful information for sustainable forest management (Msofe *et al.*, 2019).

Globally, LULC changes are critical, as human activities have significantly altered landscapes (Lambin *et al.*, 2003). Deforestation, urbanisation, and agricultural expansion are among the leading causes of these changes, with profound implications for biodiversity, ecosystem services, and the global climate system (Foley *et al.*, 2005). Remote sensing techniques, including the Landsat programme, have been widely used to track LULC changes over time and space, providing valuable insights into trends of land degradation and deforestation (Hansen *et al.*, 2013).

In Africa, LULC changes are occurring at an alarming rate due to factors such as population growth, urbanisation, and agricultural expansion (Seto *et al.*, 2012). These changes, particularly deforestation and land degradation,

have detrimental effects on biodiversity, climate regulation, and the livelihoods of millions of people who rely on natural resources. In Tanzania, the primary drivers of LULC changes include agricultural expansion, logging, and charcoal production, which have led to substantial deforestation, loss of biodiversity, and disruption of ecosystem services (Msofe *et al.*, 2019).

Minja Forest Reserve (MFR), managed by the Tanzania Forest Services Agency (TFS), is located in Mwanga District of Kilimanjaro Region and is a vital conservation area within the Eastern Arc Mountains facing significant threats from human activities (Hansen *et al.*, 2013). Despite its ecological importance, there is limited documented information on the spatial and temporal extent of LULC changes within MFR over recent decades. This study aimed to: (i) assess the extent of LULC change at MFR from 1995 to 2025; (ii) identify the drivers of LULC change at MFR; and (iii) quantify the frequency of drivers of LULC change within the reserve. The findings are intended to support evidence-based decision-making for forest conservation and sustainable resource management.

MATERIALS AND METHODS

Study area

Minja Forest Reserve is located in Mwanga District, Kilimanjaro Region, within the Eastern Arc Mountain ecosystem (Fig.1). The

forest lies between 3°35' and 3°37' S latitude and 37°40' and 37°41' E longitude, covering an area of approximately 520 hectares. The altitudinal range is 1,500 to 1,850 metres above sea level. The area experiences an oceanic climate, with annual rainfall ranging from 700 to 1,300 mm and a dry season extending from June to September.

Satellite image acquisition and processing

Four sets of satellite images were acquired for the years 1995, 2005, 2015, and 2025, selected during dry seasons with minimal cloud cover. Satellite data were sourced from Landsat 5 TM (1995), Landsat 7 ETM+ (2005), Landsat 8 OLI/TIRS (2015), and Landsat 9 OLI-2/TIRS-2 (2025). All images were accessed through Google Earth Engine (GEE) for processing and downloading. Classification was performed using supervised classification in ArcGIS Pro version 2.8.29.

Ground truthing

Ground truthing was conducted in May 2025 using a handheld GPS device to collect geographic coordinates across different land cover types: dense forest, shrubland, bare land, and water bodies. This field data was essential for validating the supervised classification of satellite images. During field visits, photographs and GPS coordinates were recorded for visible drivers of land cover change.

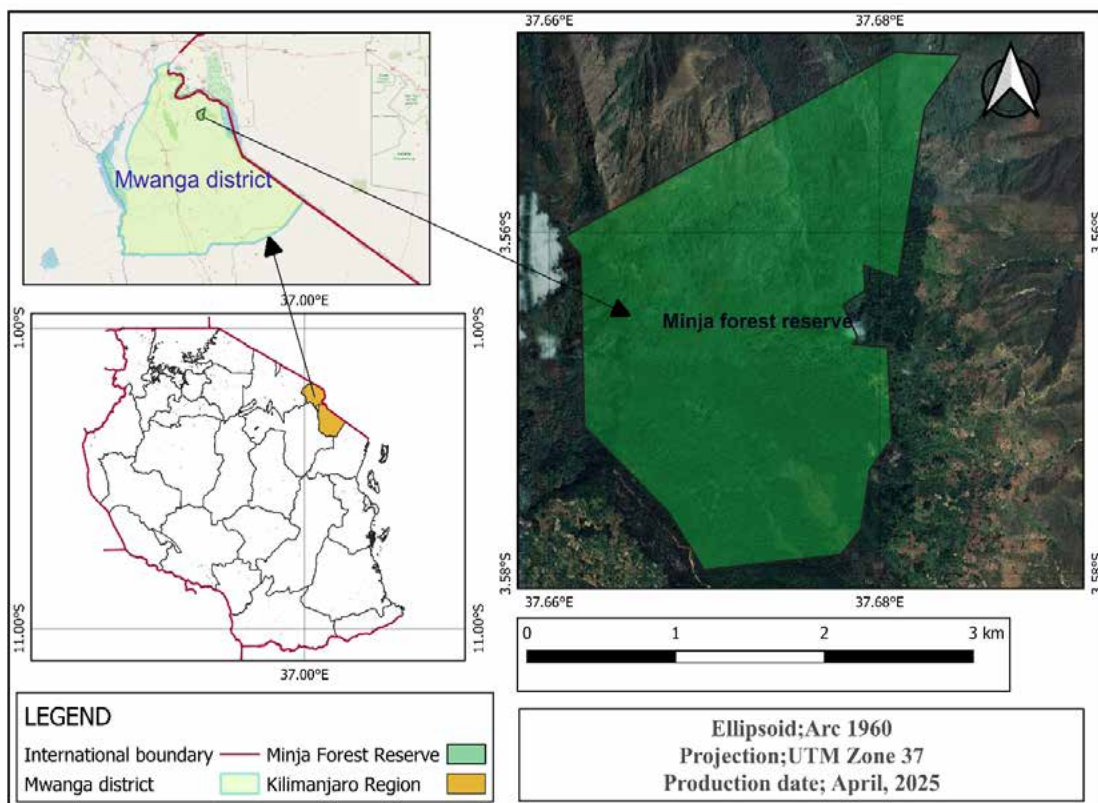


Figure 30: A map of Minja Forest Reserve in Mwanga District, Source: Author 2025

RESULTS

Extent of LULC change at Minja Forest Reserve from 1995 to 2025

The LULC classification revealed significant changes across four land cover types over the 30-year study period (Table 1). In 1995, bare land dominated the landscape at 281.62 ha (53.38%), followed by shrubland at 170.83 ha (32.64%), dense forest at 64 ha (12.64%), and water bodies at 3.7 ha (1.35%). By 2015,

dense forest had increased substantially to 291.79 ha (55.28%), representing the dominant land cover class, whilst bare land decreased to 59.86 ha (11.86%). However, by 2025, dense forest declined sharply to 87.94 ha (17.12%), shrubland increased to 311.87 ha (59.7%), and bare land rose to 120.16 ha (23.15%). Water bodies showed a consistent decrease from 3.7 ha in 1995 to 0.18 ha in 2025. Land cover maps are shown in Figure 2-5, and corresponding statistics in Table 1.

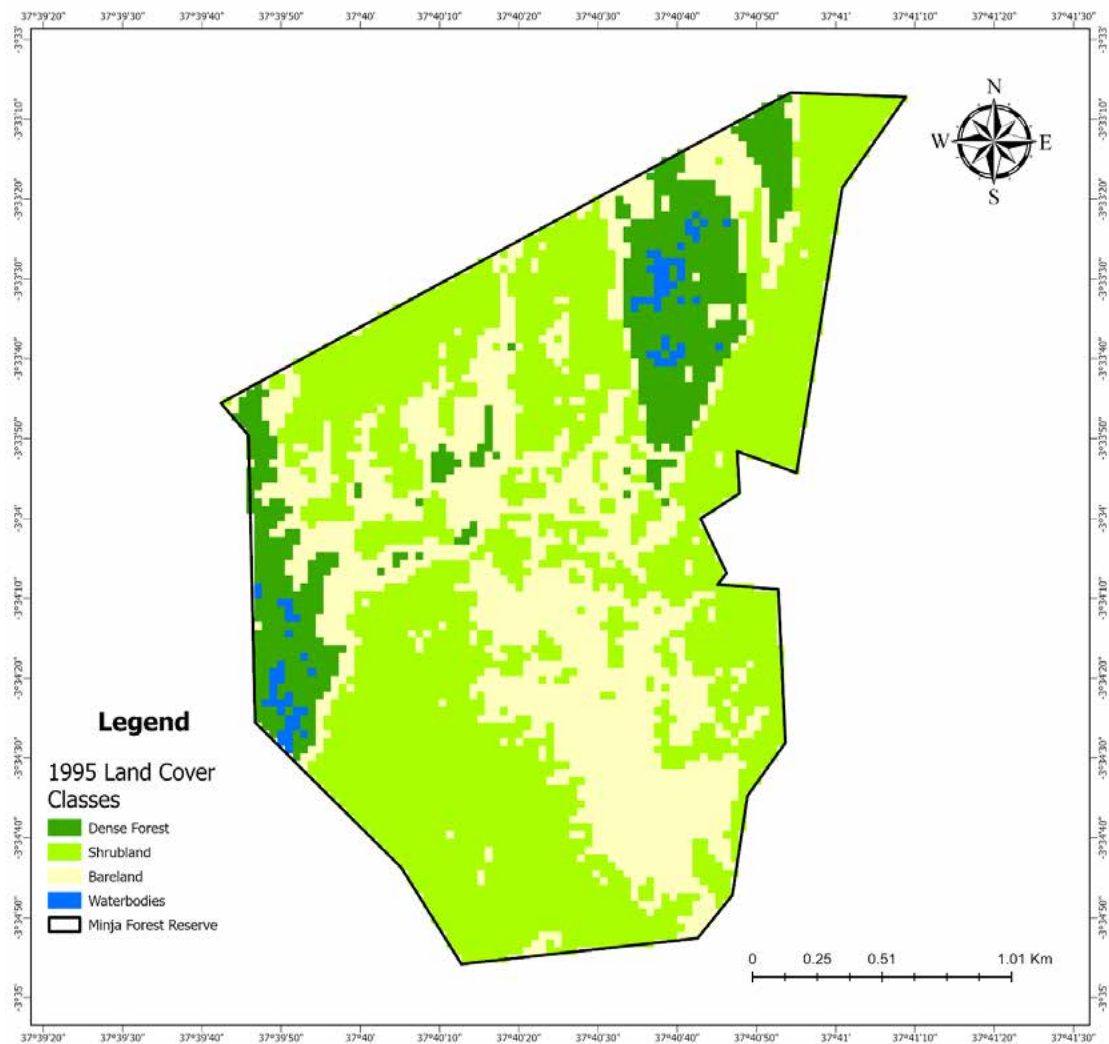


Figure 2: LUC map of MFR in 1995, Source: Classification of Landsat image, 1995

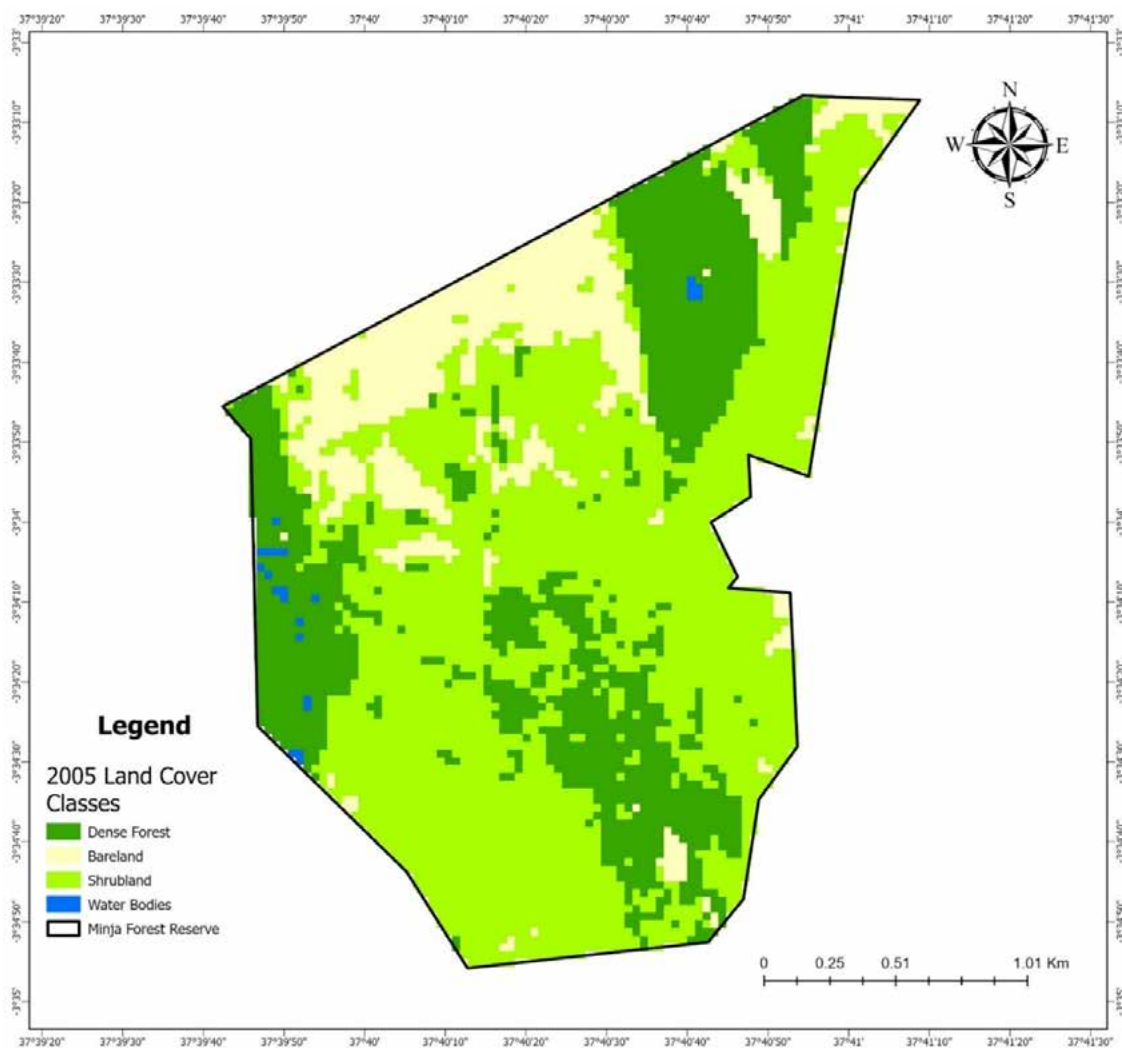


Figure 3: LUC map of MFR in 2005, Source: Classification of Landsat image, 2005

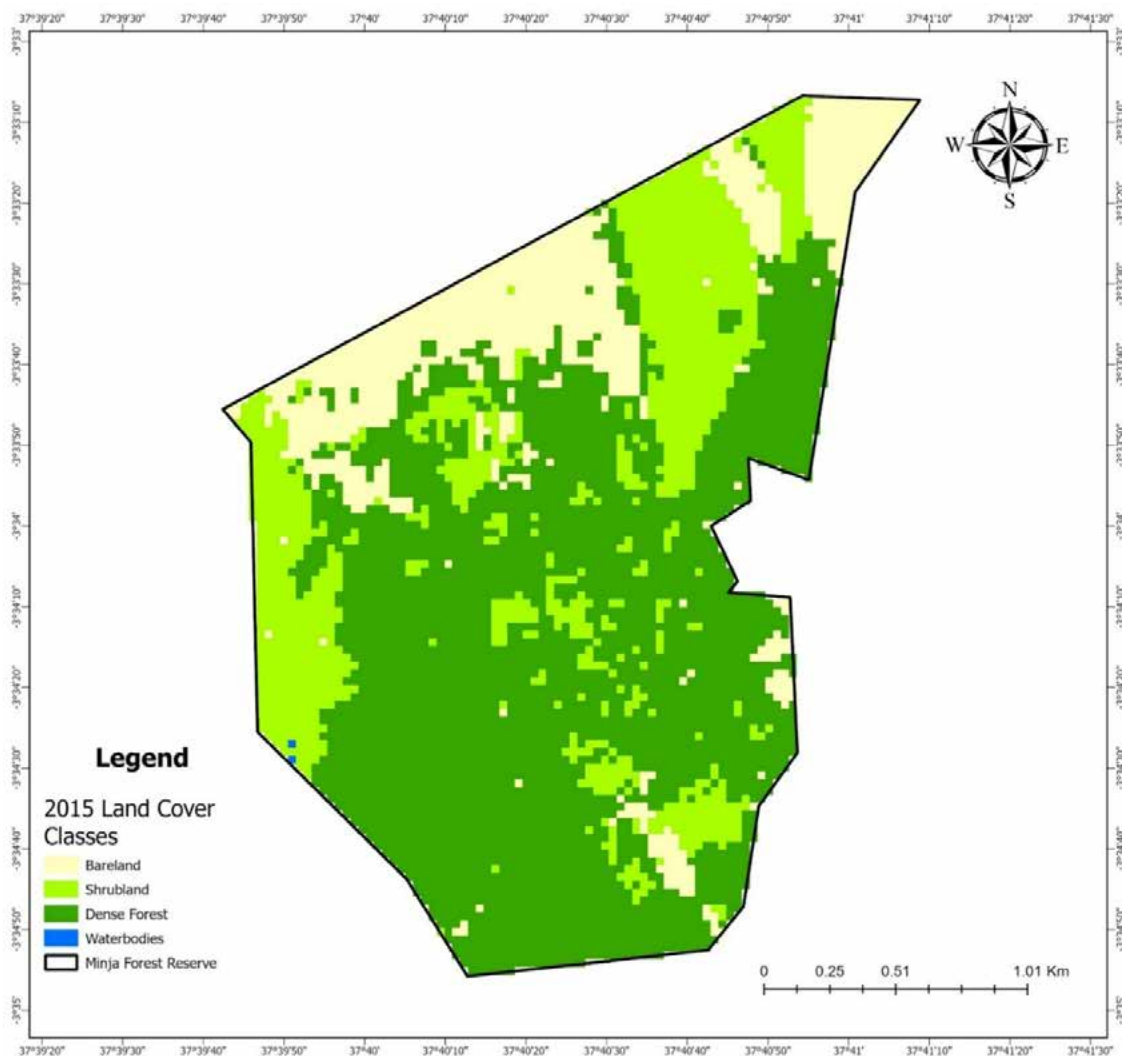


Figure 4: LUC map of MFR in 2015, Source: Classification of Landsat image, 2025

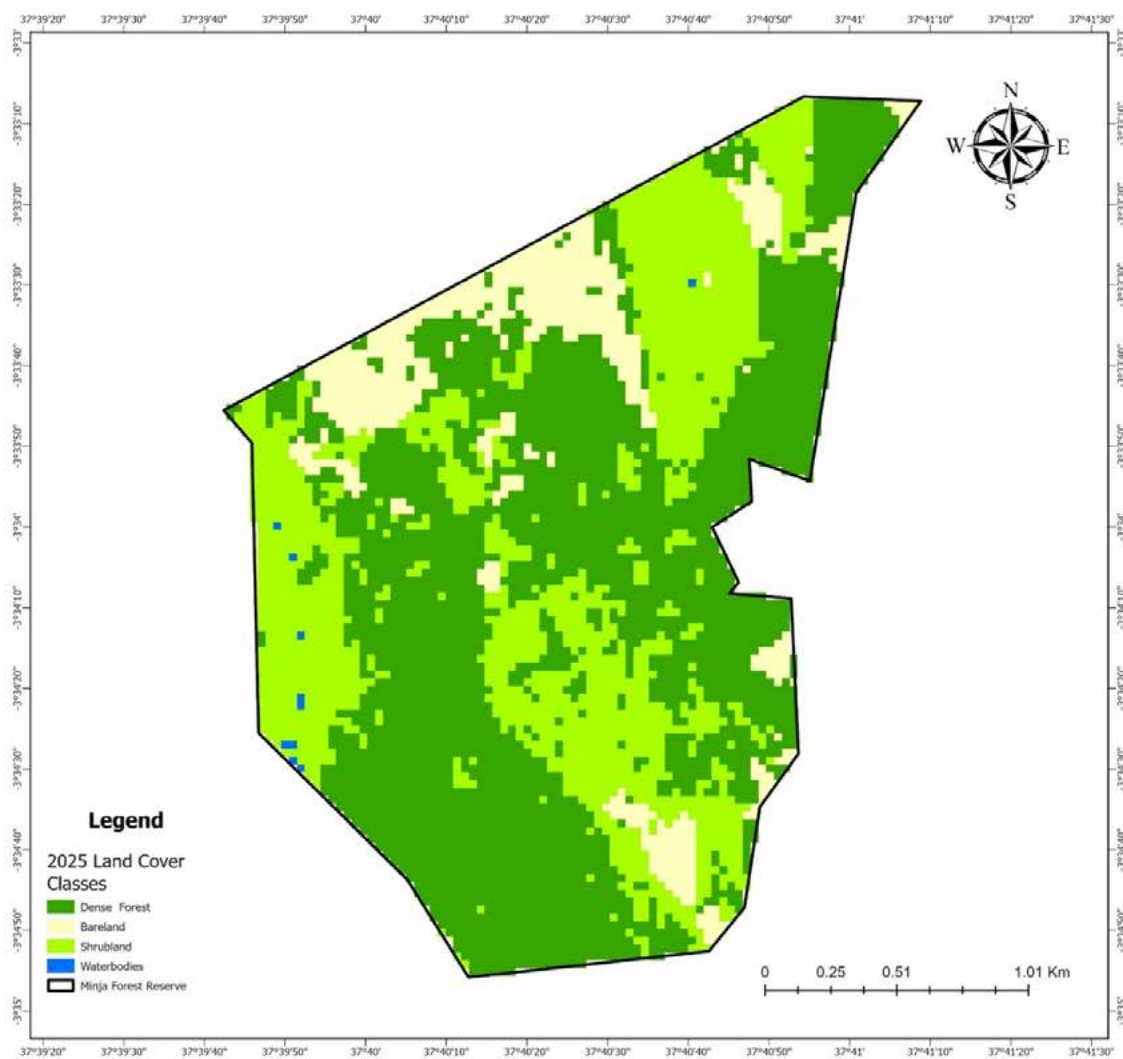


Figure 5: LUC map of MFR in 2025, Source: Classification of Landsat image, 2025

Table 1: Land cover types, land cover change, and change in area between 1995 and 2025 at Minja Forest Reserve

Land Cover	1995 (ha)	1995 (%)	2005 (ha)	2005 (%)	2015 (ha)	2015 (%)	2025 (ha)	2025 (%)
Dense Forest	64.00	12.64	146.71	28.12	291.79	55.28	87.94	17.12
Bare Land	281.62	53.38	77.50	15.16	59.86	11.86	120.16	23.15
Shrubland	170.83	32.64	293.87	56.33	167.60	32.69	311.87	59.70
Water Bodies	3.70	1.35	2.07	0.39	0.90	0.17	0.18	0.03
Total	520.15	100	520.15	100	520.15	100	520.15	100

Drivers of LULC change at Minja Forest Reserve

Based on the ground truthing survey conducted within MFR, several key drivers of LULC change were identified through direct observation. These include livestock grazing, logging, fire points, water pipe installations, and expansion of farmlands. Logging had the highest frequency of occurrence at 40%, followed by livestock grazing at 20%, water pipe installations at 20%, farmland expansion at 10%, and fire points at 10%.

DISCUSSION

Changes in dense forest area

The dense forest area at MFR experienced dramatic fluctuations over the study period. The forest increased from approximately 64 ha in 1995 to 291.79 ha in 2015, a positive sign implying natural vegetation recovery and possibly effective forest management during that period. These trends mirror documented patterns of initial forest recovery followed by rapid degradation in East Africa (Ahrends *et al.*, 2021). However, by 2025, dense forest declined sharply to approximately 87.94 ha, implying a return of deforestation pressure from logging activity and livestock grazing within the reserve (Bhattarai *et al.*, 2020).

Fluctuations in bare land area

Bare land, representing areas cleared by anthropogenic activities or naturally degraded, was the dominant land cover in 1995 at 281.62 ha. It decreased significantly to 59.86 ha by 2015, possibly due to natural vegetation recovery. However, bare land increased again to 120.16 ha by 2025, signifying renewed human activities including logging, farm expansion, and livestock grazing within MFR. These patterns reflect common land degradation problems in semi-arid areas of Tanzania (Msofe *et al.*, 2019).

Shrubland and water body dynamics

Shrubland fluctuated throughout the study period, reaching 311.87 ha by 2025, indicating that shrubland expands when forests are degraded or bare land begins to regenerate. Similar patterns have been observed in other dryland regions of Tanzania (Bhattarai *et al.*, 2020). Water bodies showed a consistent decrease from 3.7 ha in 1995 to 0.18 ha in 2025. Although small in absolute area, their decline is concerning for both wildlife within MFR and nearby local communities. The decline may result from deforestation, which leads to reduced rainfall infiltration and increased surface runoff, as well as sedimentation. Similar impacts have been documented in the Wami-Ruvu and Kilombero river basins (Twisa *et al.*, 2020).

Quantifying the drivers of LULC change

Illegal logging, the most frequently observed driver at 40%, has a significant impact on forest cover, leading to habitat loss and forest degradation (Bhattarai *et al.*, 2020). Livestock grazing was commonly observed within the forest, where domestic animals feed on grasses and shrubs, preventing natural plant regeneration and compacting soil (Msofe *et al.*, 2019). Water pipe installations create footpaths and cause vegetation disturbance, increasing human presence and land clearance. Farmland expansion near the forest area is a response to local population pressure, with crops replacing native vegetation, a trend documented across East Africa (Foley *et al.*, 2005). Fire points, often used by hunters or to clear land, pose a serious risk to biodiversity and often transform dense forest into shrubland or bare land (Twisa *et al.*, 2020).

CONCLUSION AND RECOMMENDATIONS

The findings demonstrate that illegal human activities, particularly logging and livestock grazing, have negatively impacted MFR. The increase in bare land and shrubland

between 2015 and 2025 indicates that current conservation strategies require enhancement. Through strengthened protection measures by TFS management, illegal activities may decrease and flora species habitats can recover. The following recommendations are made:

- i. Strengthen forest protection and law enforcement by increasing day and night patrols and enforcing regulations to stop illegal logging, livestock grazing, and farm encroachment.
- ii. Engage local communities, including leaders and residents of Vuchama-Ngofi village, through education and awareness programmes and support with alternative livelihoods.
- iii. Establish buffer zones and implement reforestation programmes for heavily degraded areas, involving local communities in tree planting.
- iv. Enhance water source protection by preventing vegetation clearing near streams and springs within the reserve.
- v. Conduct continuous monitoring and evaluation using satellite technology to track environmental changes and assess conservation effectiveness.

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ASSESSMENT OF LAND USE AND LAND COVER CHANGE: A CASE STUDY OF KITENDENI WILDLIFE CORRIDOR, TANZANIA

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ABSTRACT

Land use and land cover (LULC) changes significantly impact habitats and biodiversity, affecting wildlife distribution and movement. Understanding these changes is vital for effective conservation, especially in wildlife corridors. This study assessed LULC changes over 30 years (1994–2024) at the Kitendeni Wildlife Corridor in northern Tanzania. LULC changes between 1994, 2004, 2014, and 2024 were classified and monitored for spatial-temporal patterns using Google Earth Engine (GEE) and ArcGIS Pro, employing a supervised Random Forest classification approach with ground truthing. Classification accuracy was validated with confusion matrices, achieving high overall accuracies (90–91.5%) and Kappa Indices of Agreement (KIA) ranging from 0.85 to 0.90. Predictions for 2034 and 2044 were generated using the MOLUSCE plugin with Cellular Automata simulation in QGIS. Change detection analysis revealed that grassland exhibited the highest positive change at 28.8%, followed by forest at 11.7%. Conversely, bushland and bare land declined by 24.9% and 15.6%, respectively. Projections indicate continued grassland expansion and forest stabilisation through 2044, whilst bushland may nearly disappear. These results suggest a shift towards a more vegetated ecosystem, though the near-disappearance of bushland raises concerns about habitat diversity. The study recommends that the government formulate a comprehensive land-use management plan and implement a community-centred natural resources management strategy to enhance resource utilisation and diminish human activities within the corridor.

Keywords: Google Earth Engine; Kitendeni; land cover change; modelling; wildlife corridor

INTRODUCTION

Land use and land cover (LULC) change involves the quantitative increase or decrease in specific land types, significantly affecting the quantity and quality of terrestrial vertebrate habitats worldwide (Briassoulis, 2019). As human-environment interactions increase, the demand to meet human needs often leads to altering, transforming, or even destroying natural vegetation and landscapes (Kanianska, 2016). Land use can influence land cover, and changes in land cover can in turn affect land conditions, structure, and function, ultimately impacting future land use (Khalid Kija *et al.*, 2020).

Globally, land degradation, habitat fragmentation, species decline, and LULC changes threaten biodiversity by affecting wildlife distribution and reducing ecosystem services (Liu *et al.*, 2020; Powers & Jetz, 2019). African countries are generally experiencing declines in terrestrial species, including migratory mammals, as habitat loss, fragmentation, and overexploitation continue to increase due to LULC changes resulting from human-induced factors (Bolger *et al.*, 2008). Blockage of wildlife corridors has reached a level that threatens some wildlife species with extinction, causing various protected areas in Africa to become isolated islands within an anthropogenically altered landscape (Bevanda *et al.*, 2014; Newmark, 2008). East African

countries, including Tanzania, have reported an increase in biodiversity loss due to land-use changes (Maitima *et al.*, 2009; Msoffe *et al.*, 2011).

The Kitendeni Wildlife Corridor forms a critical linkage between Kilimanjaro National Park in Tanzania and Amboseli National Park in Kenya. However, ongoing land use changes raise serious concerns about its long-term viability. Encroachment through settlement expansion and agriculture has significantly altered the natural landscape, reducing available habitat and food for migratory species such as large herbivores (Schübler *et al.*, 2018). Local communities have converted areas once utilised and inhabited by wildlife (Pittiglio *et al.*, 2012), leading to a noticeable decline in wildlife populations using the corridor (Kiringe & Okello, 2007). If this trend is not controlled, Kilimanjaro National Park and Amboseli National Park are likely to lose their ecological connectivity, thereby becoming less ecologically viable (Maitima *et al.*, 2009; Mbane *et al.*, 2019). This study aimed to assess LULC changes in the Kitendeni Wildlife Corridor from 1994 to 2024 and predict future land cover scenarios to 2044.

MATERIALS AND METHODS

Study area description

The Kitendeni Wildlife Corridor (KWC) is located in West Kilimanjaro, northern Tanzania, within the broader Amboseli-Kilimanjaro ecosystem (Fig.1). It forms part of the Enduimet Wildlife Management Area (EWMA), a community-managed conservation area that supports eco-tourism and sustainable land use. The corridor spans approximately 27.3 km², serving as a critical linkage between Kilimanjaro National Park and Amboseli National Park in Kenya, facilitating the movement of wildlife, especially elephants and large herbivores. Geographically, it lies between approximately 2.88°S and 3.05°S in latitude and 37.10°E and 37.25°E in longitude. The area is characterised by a semi-arid climate and bimodal rainfall patterns, with dry seasons from June to September and January to February. The Maasai are the dominant community, with livelihoods dependent on livestock and limited farming. The corridor borders Kitendeni and Irkiswa villages.

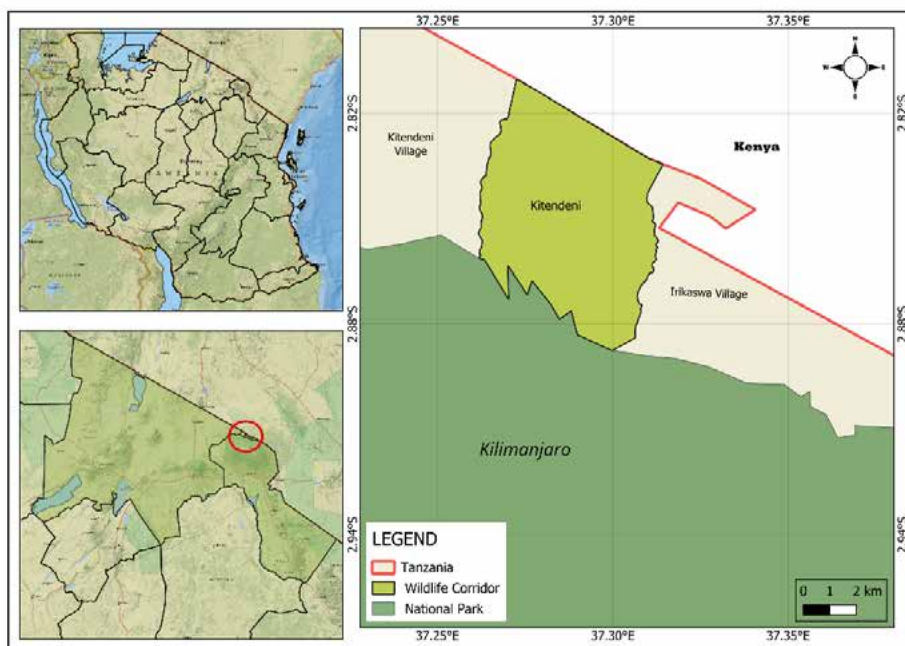


Figure 31: A map showing the study area at Kitendeni Wildlife Corridor. Source: Author, 2025

Data collection methods

The study employed a multiple-method research design integrating quantitative satellite-based remote sensing and qualitative field-based observations (Kamal *et al.*, 2020). Spatial data for 1994, 2004, 2014, and 2024 were obtained from Landsat satellite images using GEE (Borges *et al.*, 2022). All satellite images were downloaded during the dry season at a spatial resolution of 30 m, as dry-season imagery has less cloud cover (Khalid Kija *et al.*, 2020). Satellite images were obtained from Landsat 5 TM (1994), Landsat 7 ETM+ (2004), and Landsat 8 OLI (2014, 2024) sensors, using Collection 2 Level 2 surface reflectance data that were geometrically, radiometrically, and atmospherically corrected (Kashaigili & Majaliwa, 2010). Only imagery with less than

10% cloud cover was selected (Symeonakis & Higginbottom, 2014).

Land cover classification was performed using the Random Forest (RF) algorithm in GEE, employing the `ee.classifier.smileRandomForest` function (Noi Phan *et al.*, 2020). RF is widely recognised as more efficient and accurate for land cover classification compared to traditional methods such as maximum likelihood and support vector machines (Abdel-Rahman *et al.*, 2014; Li *et al.*, 2016). Training data were gathered via the GEE platform across the KWC, covering all four land cover classes: bare land, bushland, forest, and grassland (Table 1). Table 2 provides specification of Landsat images used. Ground truth data were collected from accessible research sites using GPS (Bai *et al.*, 2015).

Table 1: Description of land cover types at Kitendeni Wildlife Corridor

Land Cover Type	Description
Bare Land	Areas with no vegetation cover, characterised by exposed soil
Bushland	Woody plants with shrub canopy <6 m high, mainly acacia, sometimes mixed with scattered trees
Forest	Areas covered with trees forming closed canopies, heights 2–40 m
Grassland	Areas dominated by grasses 22–150 cm tall, may include herbs, shrubs, or scattered trees

Table 9: Specification of the Landsats that were used in classification

Satellite/Sensor	Date	Spatial Resolution	Spectral Resolution
Landsat 5 (ETM+)	1995/03/19	30 m	Multispectral (19 bands)
Landsat 5 (ETM+)	2004/11/04	30 m	Multispectral (19 bands)
Landsat 8 OLI	2014/03/12	30m	Multispectral (9 bands)
Copernicus/S2_SR	2023/01/28	10 m	Multispectral (12 bands)

LULC modelling

Future LULC predictions were generated using the MOLUSCE plugin within QGIS, incorporating a Cellular Automata (CA) simulation approach (Mondal *et al.*, 2016;

Tahir *et al.*, 2025). Historical land cover data, environmental drivers, and spatial dynamics were combined with the CA model to produce spatially explicit predictions of future land use trends for 2034 and 2044 (Jana *et al.*, 2022; Asif *et al.*, 2023).

RESULTS

Land use and land cover change from 1994 to 2024

The results of the LULC classification for 1994–2024, together with projections for 2034 and 2044, are presented in Table 3. In 1994, bare land dominated the landscape at 11.16 km² (40.83%), followed by bushland at 8.17 km² (29.87%), grassland at 6.52 km² (23.86%), and forest at 1.48 km² (5.42%). By 2024, grassland had become the dominant land cover at 13.32 km² (48.74%), forest increased

to 4.55 km² (16.65%), bare land increased to 8.25 km² (30.18%), whilst bushland declined sharply to 1.21 km² (4.41%), representing an approximately 85% loss from its 1994 extent. Projections for 2034 indicate grassland reaching 15.75 km² (57.63%) and forest slightly decreasing to 4.34 km² (15.86%), whilst bare land is expected to decline to 5.97 km² (21.82%). By 2044, forest cover is projected to reach its highest level at 5.35 km² (19.58%), and grassland is expected to cover 16.98 km² (62.18%), approximately 2.5 times its 1994 area. Figures 2-10 illustrates the changes between the years

Table 3: Spatio-temporal changes in land cover area (km²) and percentage (%) from 1994 to 2044

Land Cover	1994 (km ²)	1994 (%)	2004 (km ²)	2004 (%)	2014 (km ²)	2014 (%)	2024 (km ²)	2024 (%)	2034 (km ²)	2034 (%)	2044 (km ²)	2044 (%)
Forest	1.48	5.42	2.32	8.47	2.78	10.17	4.55	16.65	4.34	15.86	5.35	19.58
Grassland	6.52	23.86	10.11	36.98	12.50	45.76	13.32	48.74	15.75	57.63	16.98	62.18
Bushland	8.17	29.87	7.55	27.63	5.61	20.52	1.21	4.41	1.28	4.66	1.11	4.06
Bare Land	11.16	40.83	7.36	26.91	6.43	23.50	8.25	30.18	5.97	21.82	3.87	14.16

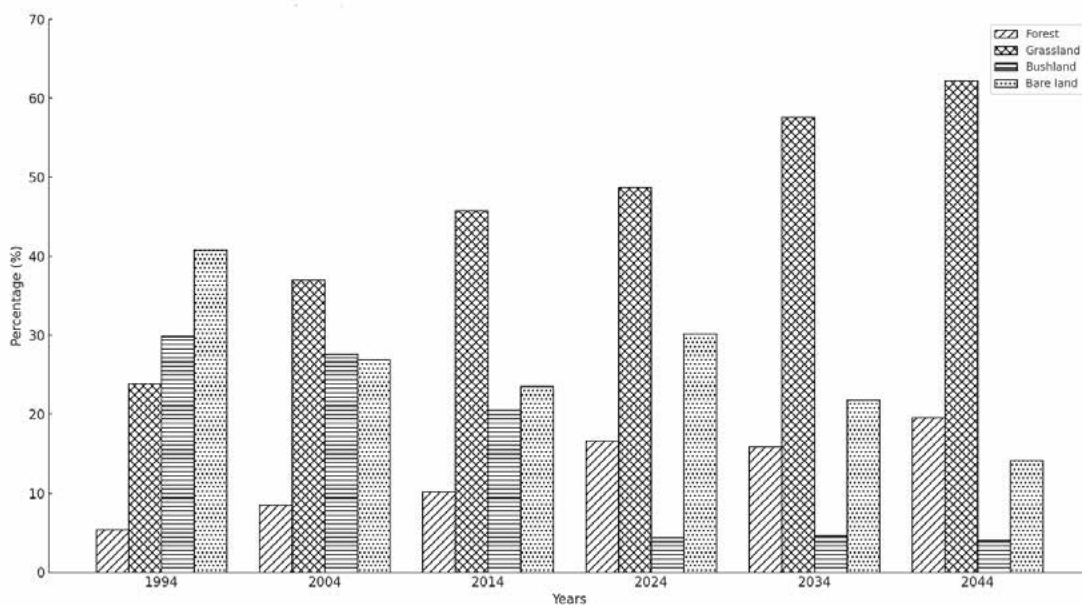


Figure 2: A Clustered Bar Chart showing the percentage distribution of Land cover classes from 1994 to 2044

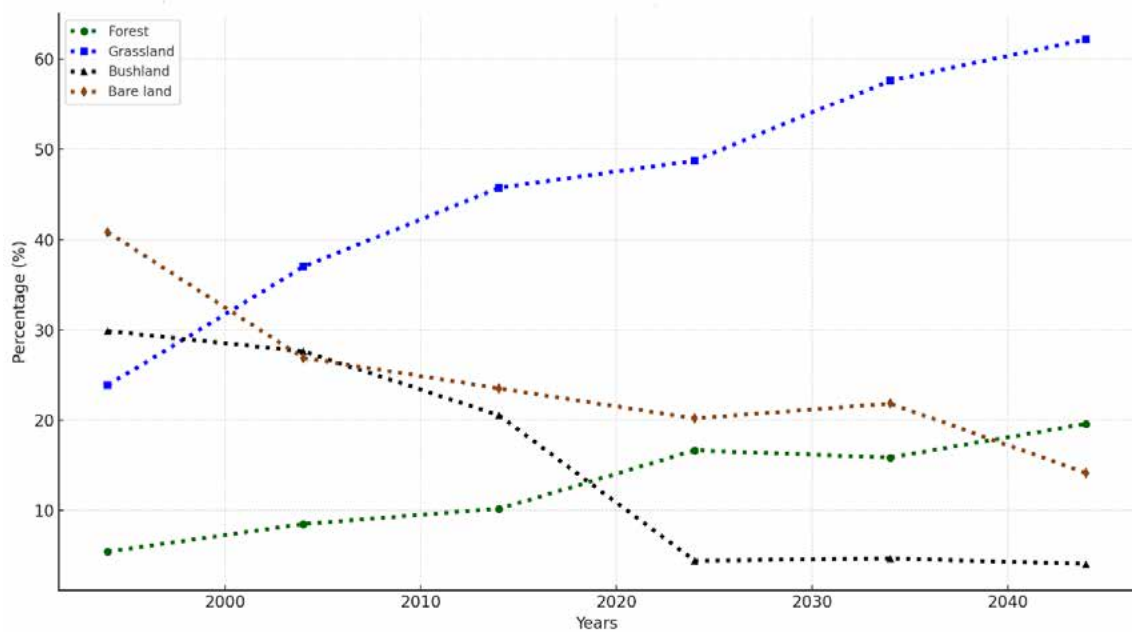


Figure 3: A Line Graph (Dotted style with markers) showing the trends in Land cover percentage from 1994 to 2044

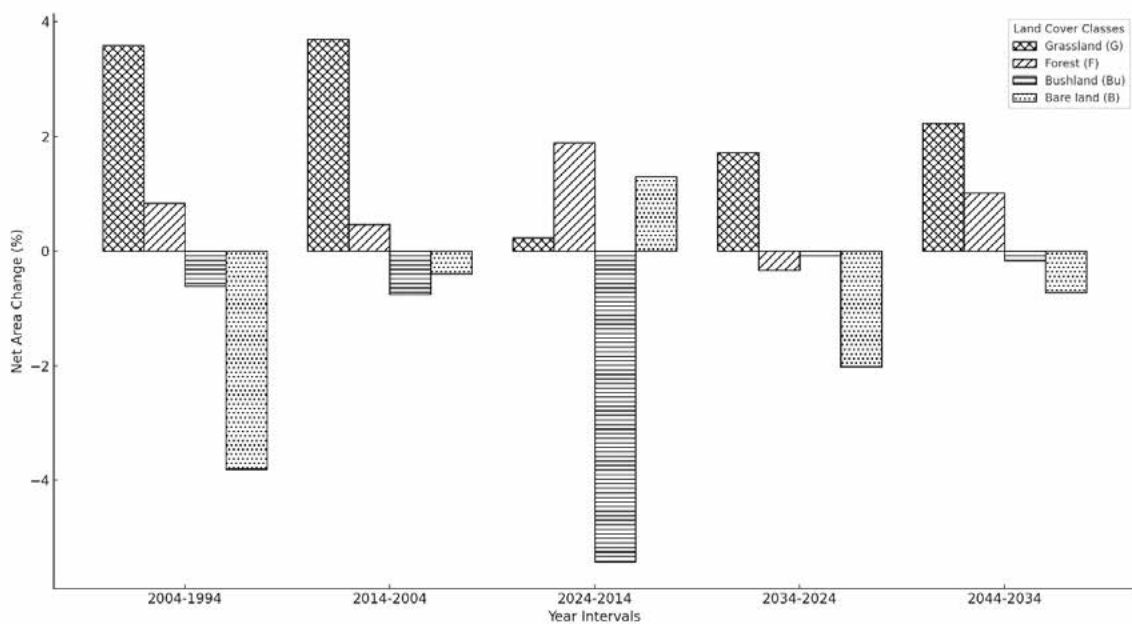


Figure 4: A Clustered Bar Chart showing Net Area Change (%) of Land cover classes across time intervals from 1994 to 2044

The patterns of changes in land use and land cover over space and time.

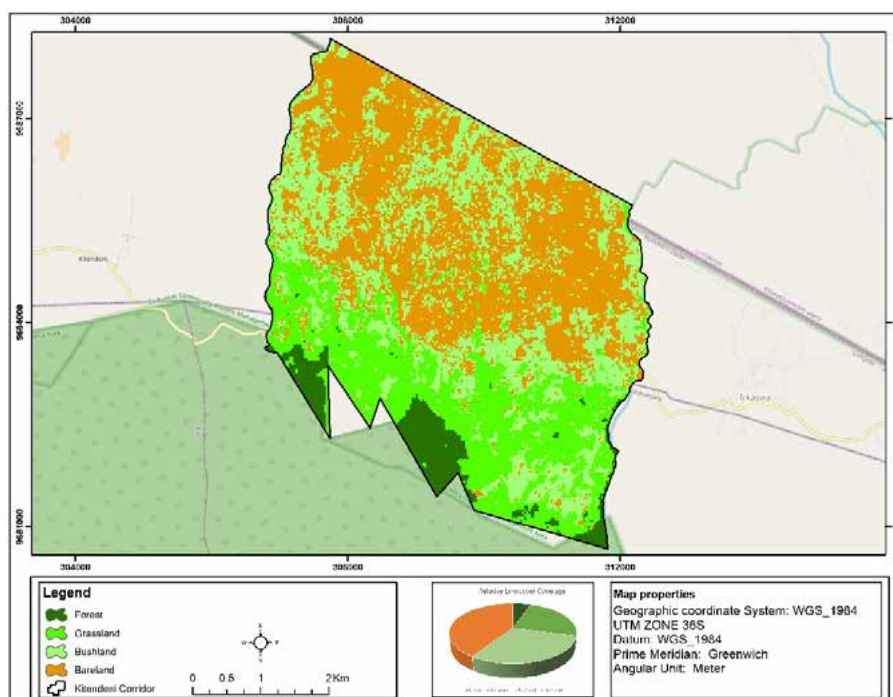


Figure 5: A map showing a classified satellite image of 1994. Source: Landsat thematic classification, 1994

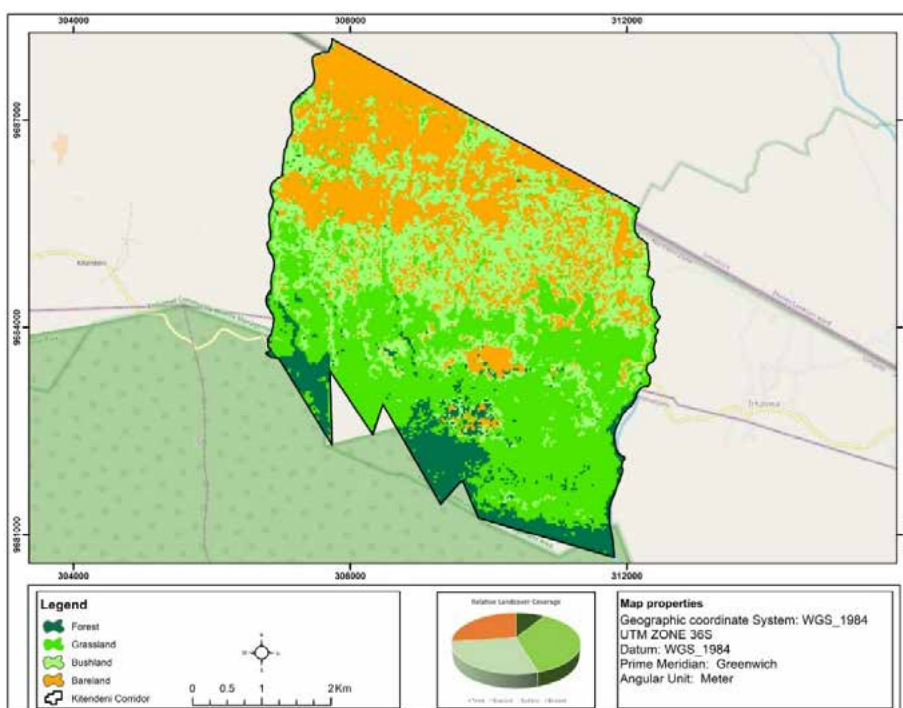


Figure 6: A map showing a classified satellite image of 2004. Source: Landsat thematic classification, 2004

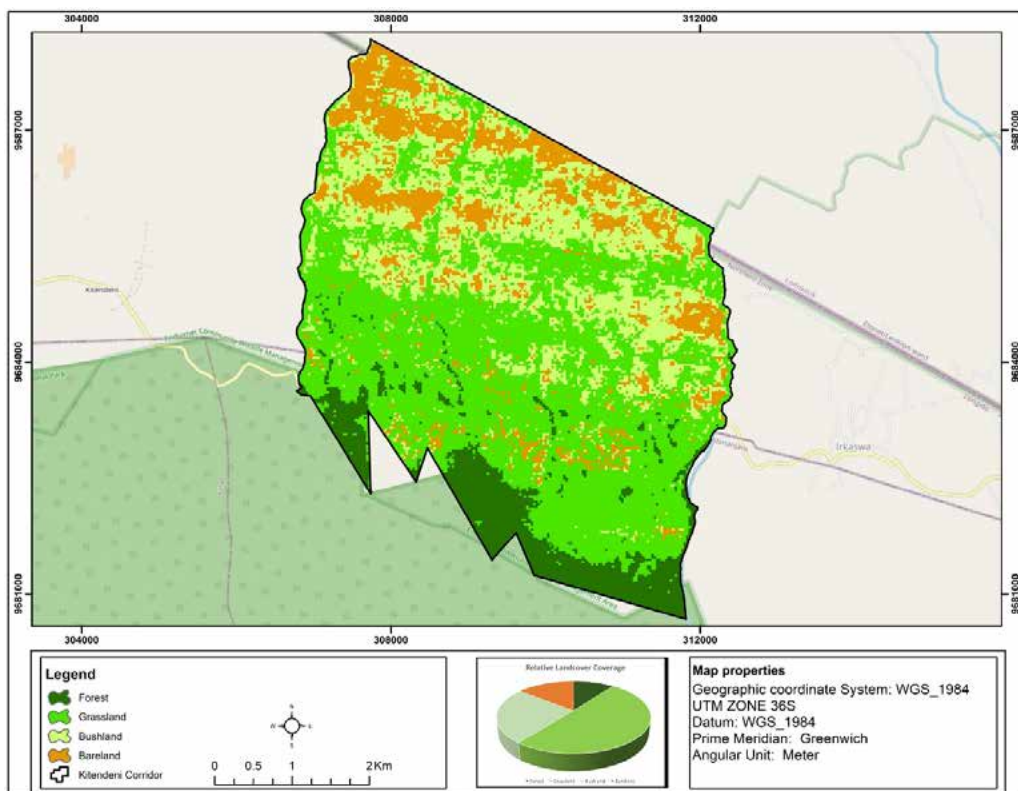


Figure 7: A map showing a classified satellite image of 2014. Source: Landsat thematic classification, 2014

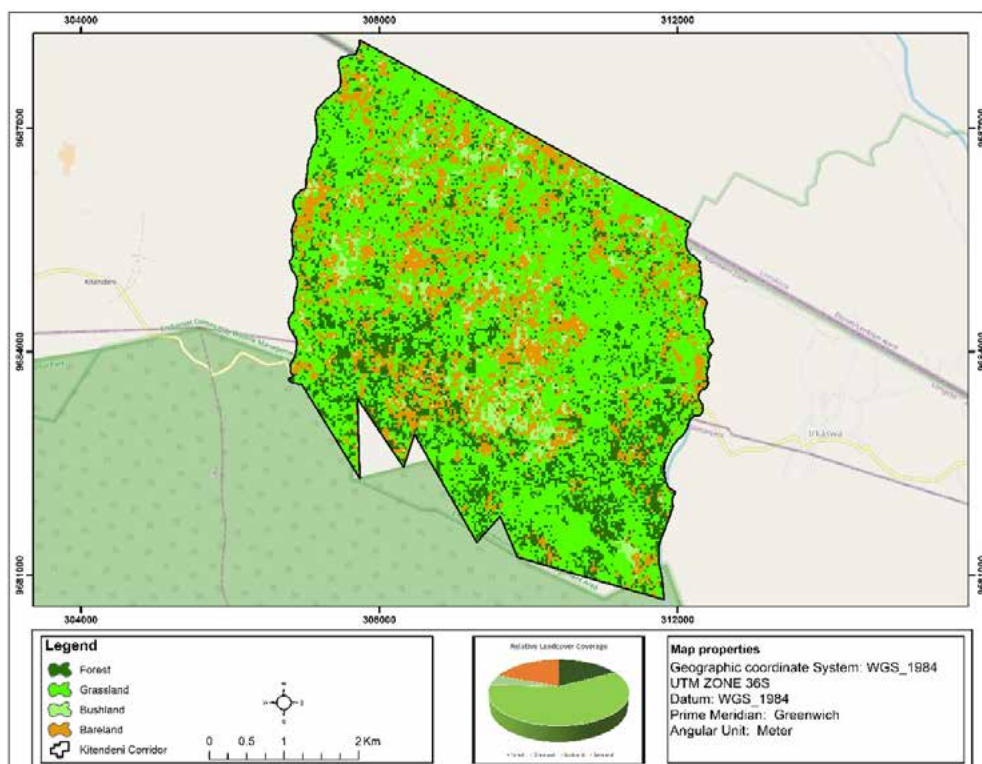


Figure 8: A map showing a classified satellite image of 2024. Source: Landsat thematic *classification*, 2024

Predicting future land use, land cover changes in Kitendeni Wildlife Corridor

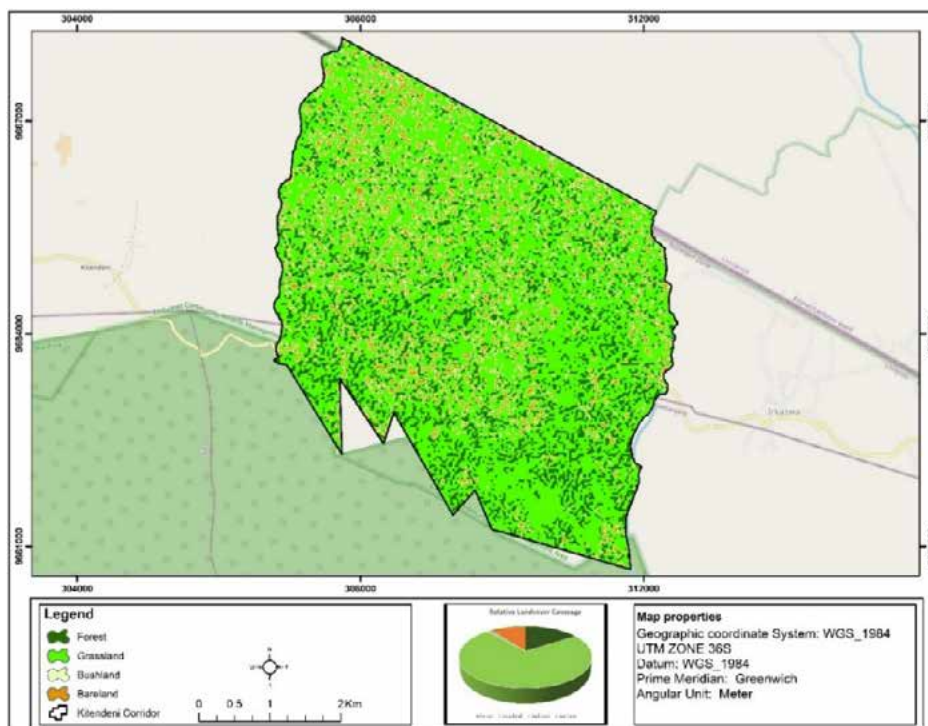


Figure 9: A map showing a classified satellite image of 2034. Source: Landsat thematic classification

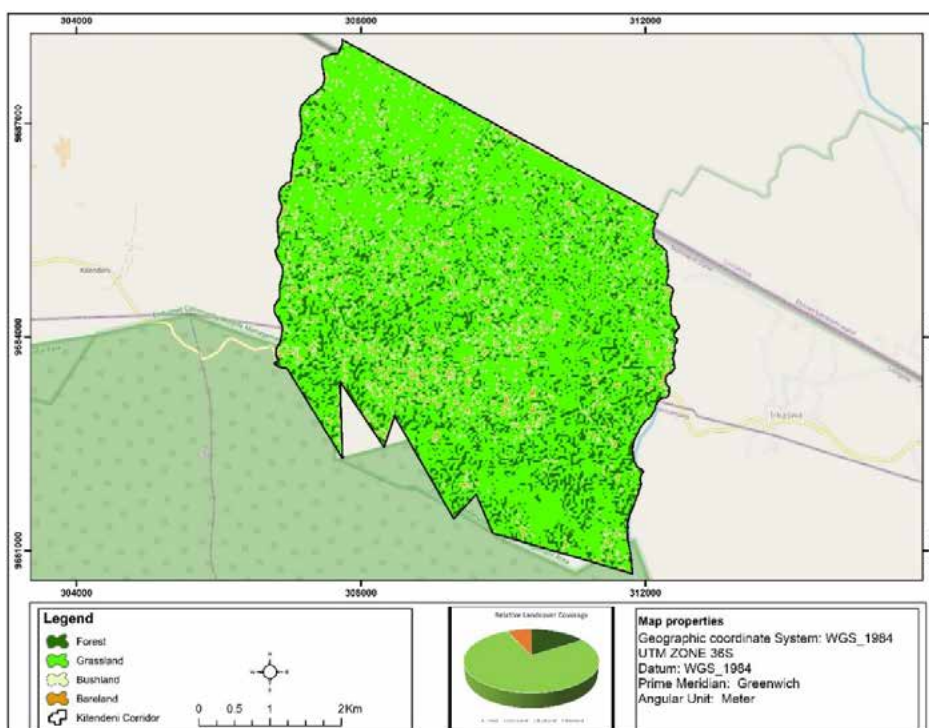


Figure 10: A map showing a classified satellite image of 2044. Source: Landsat thematic classification

Drivers of LULC change identified during ground truthing at Kitendeni wildlife corridor

Drivers of LULC change

Ground truthing identified several drivers of LULC change within the KWC, including agricultural expansion, settlement growth, livestock grazing, and charcoal production. These drivers were spatially mapped (Fig. 11) and contribute to the observed trends in land cover transformation.

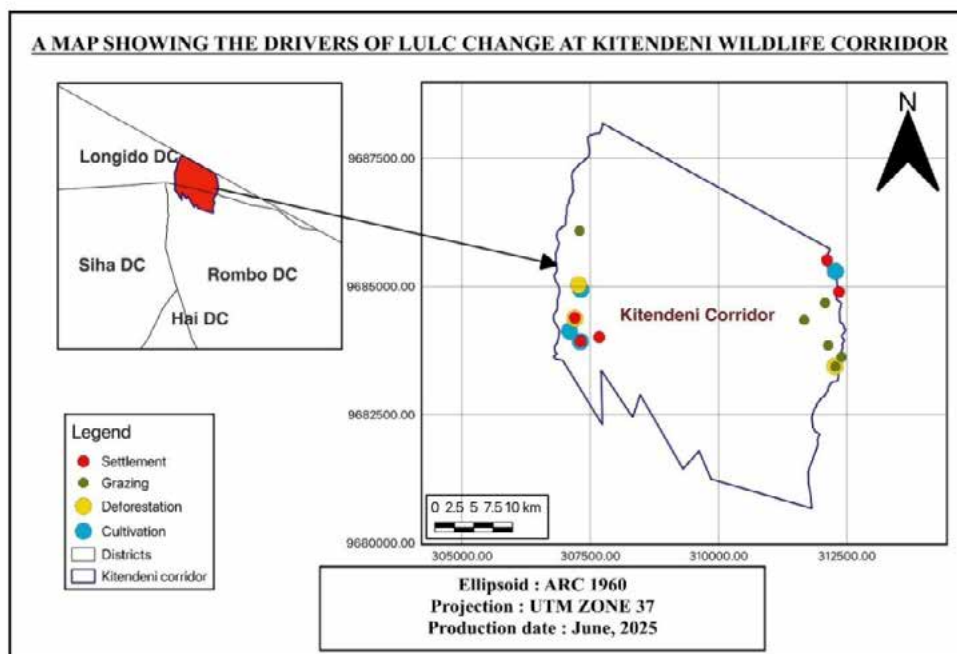


Figure 11: A map that shows the drivers identified during Ground Truthing at Kitendeni Wildlife Corridor. Source QGIS Version 3.4

DISCUSSION

Forest area in the KWC increased consistently from 1.48 km² in 1994 to 4.55 km² in 2024, suggesting effective conservation initiatives or favourable environmental factors promoting forest expansion (Yadav *et al.*, 2012). Projections show continued forest growth, with a slight dip in 2034 followed by recovery to 5.35 km² by 2044, reflecting potential fluctuations due to ecological or anthropogenic factors (Mallegowda *et al.*, 2015). Grassland expanded substantially from 6.52 km² in 1994 to 13.32 km² in 2024, suggesting improved land management and a reduced rate of encroachment (Schüßler *et al.*, 2018). Projections indicate continued grassland expansion to 16.98 km² by 2044, possibly driven by ongoing conservation efforts

(Saharan & Nehra, 2025). However, expanding grasslands could encroach upon other vital habitats, potentially affecting biodiversity.

Bushland experienced a dramatic reduction of approximately 83.3%, from 8.17 km² in 1994 to 1.21 km² in 2024, primarily resulting from land conversion for agricultural use and settlement (Muriithi, 2016; Nzunda *et al.*, 2013). Although the decline rate appears to be slowing after 2024, this sharp loss directly affects species dependent on bushland ecosystems and highlights the urgent need for targeted conservation strategies.

Bare land decreased from 11.16 km² in 1994 to 6.43 km² by 2014, indicating ecological recovery driven by land management strategies

(Waswa, 2012). However, a resurgence to 8.25 km² by 2024 suggests renewed human activities. Projections for 2034 and 2044 indicate continued decline to 3.87 km², reflecting a positive trend towards land recovery and reforestation.

CONCLUSION AND RECOMMENDATIONS

Over the past 30 years, the Kitendeni Wildlife Corridor has experienced significant LULC changes within its total area of 27.30 km². Forest cover increased from 5.4% to 17.1%, whilst grassland became the dominant land cover, rising from 23.9% to 48.7%. Bushland and bare land decreased substantially. Projections for 2034 and 2044 indicate continued grassland expansion and forest stabilisation, although bushland may nearly disappear. Whilst forest and grassland recovery is encouraging, the near-disappearance of bushland and the resurgence of bare land in 2024 raise concerns about habitat diversity and ecological balance. The following recommendations are made:

- i. Community engagement and sustainable resource management should be encouraged, incorporating traditional ecological knowledge and offering incentives for conservation-oriented practices.
- ii. Regular ecological monitoring and adaptive management protocols should be implemented to track changes in biodiversity, evaluate restoration success, and guide future conservation strategies.
- iii. Government authorities should formulate a comprehensive land-use management plan for the corridor to protect ecological connectivity between Kilimanjaro National Park and Amboseli National Park.

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DEMOGRAPHIC DYNAMICS OF YELLOW BABOONS (*PAPIO CYNOCEPHALUS*) IN MIKUMI NATIONAL PARK, TANZANIA

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ABSTRACT

Field studies of natural mammal populations present powerful opportunities to investigate the determinants of health and aging using fine-grained observations of known individuals across the life course. This study investigated the demographic dynamics of two habituated yellow baboon (*Papio cynocephalus*) troops, namely V5 and V6, which were monitored four days per week from 06:30 am to 4:00 pm in the Viramba baboon range of Mikumi National Park, Tanzania, between January 2020 and December 2024. Troop follows enabled the collection of data on population structure, including individual counts, numbers of births and deaths, and transfers. Descriptive analyses were used to generate frequency distribution graphs illustrating the troop's composition and population trends. Troop V6 maintained a larger and more stable adult female cohort (mean = 8.4) and exhibited strong population growth, increasing from 30 to 51 individuals (net gain = +21), driven primarily by high natality (26 births) and low mortality (3 deaths). Troop V5, which began with 19 individuals, showed greater fluctuation and ended with 21 individuals (net gain = +7) despite 19 recorded births, reflecting higher mortality (11 deaths, largely among infants) and a net emigration of 4 males. Over the five years, births were the principal driver of population increase in both troops, whereas mortality and male dispersal exerted counteracting effects. These contrasting demographic trajectories underscore the central role of adult female number and cohort stability in shaping reproductive success and long-term troop viability in savanna baboons.

These findings provide insights into the interplay of reproduction, mortality, and male-driven dispersal in a savanna ecosystem and offer an important baseline for future conservation and management strategies.

Keywords: *Male dispersal, Population trends, Reproductive success, Troop composition, Yellow baboon*

INTRODUCTION

Yellow baboons (*Papio cynocephalus*) are social primates of East African savannahs, with population dynamics shaped by reproduction, mortality, and dispersal (Silk *et al.*, 2010; Alberts, 2019; Zippel *et al.*, 2019; Campos *et al.*, 2021). Their social structure, is characterized by stable female cohorts and male dispersal, which influences troop stability and growth (Tung *et al.*, 2023). In protected areas like Mikumi

National Park, Tanzania, anthropogenic pressures, such as vehicle interactions with roads, pose risks to yellow baboons (Kitegile *et al.*, 2022). Understanding these dynamics is critical for conservation, particularly in human-influenced landscapes where data are limited. Previous studies highlighted that female social bonds and status as key predictors of their survival (Sapolsky, 2005; Campos *et al.*, 2021), while long-term demographic studies of yellow baboons have revealed how ecological

variation, social rank, kinship, and disease interact with individual life histories related to troop population dynamics (Rasmussen, 1981; Kleindorfer & Wasser, 2004).

Mikumi National Park (MNP) hosts one of the most long-studied troops of yellow baboons since 1970s, which has generated valuable information on their population structure, social organization, reproductive strategies, and long-term trends in abundance (Hausfater & Watson, 1976; Wasser *et al.*, 2004). These long-term studies in MNP have revealed significant fluctuations in their population size from 1966 to 2002, likely driven by ecological stressors, disease, predation, or anthropogenic influences (Wasser *et al.*, 2004). Such findings demonstrate that even their populations residing within protected areas can experience demographic instability, necessitating population evaluations and continuous monitoring.

In this study, we used the long-term data for the two habituated troops, namely V5 and V6, in MNP for the period covering January 2020 to December 2024 to assess the demographic dynamics of yellow baboons by: Firstly, establish the troop's age-sex composition, and secondly, evaluate the troop's population trends. We hypothesized that stable female cohorts drive population growth, while road-

related mortality disproportionately affects smaller troops, limiting their viability.

MATERIALS AND METHODS

Study Area

The study was conducted in MNP, located in South-eastern Tanzania and covering an area of 3,230 km². The park lies between 7°05' to 7°50' south, and 36°50' to 37°35' east, approximately 300 km south-west of Dar es Salaam city. MNP forms part of the complex Nyerere-Selous-Mikumi ecosystem (Gunn *et al.*, 2013). The park consists of lowland savannas, miombo woodlands, and floodplain grasslands (Backéus *et al.*, 2004). Mean annual rainfall is about 1,000 mm, with a bimodal wet season spanning from November to May and a dry season from June to October. Temperatures average around 24°C during the wet season and exceed 35°C during the dry season. The A7 (TANZAM) highway bisects roughly one-third of the north portion of the park, creating potential vehicle-collision risks for wildlife, including yellow baboons, and contributing to overall roadkill (Drews, 1991; IPP Media, 2022; Kitegile *et al.*, 2022). The study was conducted within the Viramba range, approximately 10 km north-northeast of the MNP headquarters (Fig. 1).

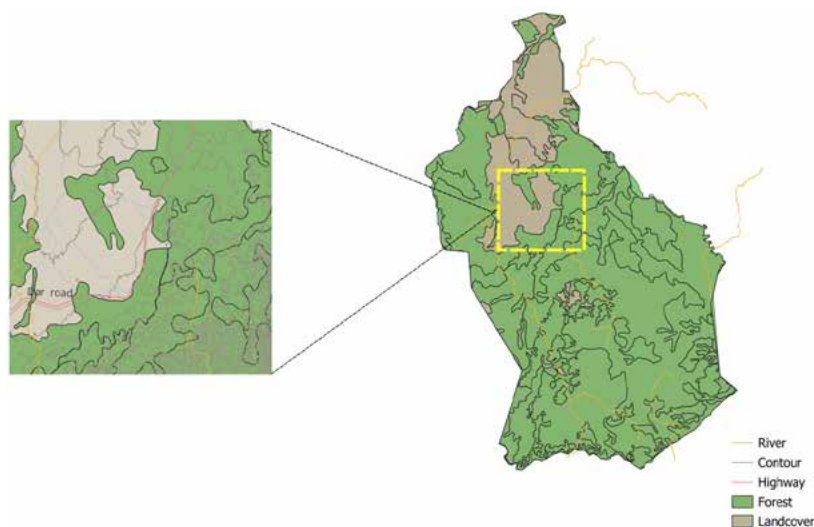


Figure 1: Map of Mikumi National Park showing the location of the Viramba range and A7 TANZAM highway.

Study Design

We used a longitudinal study design in which the data were collected and compiled from two habituated yellow baboon troops, V5 and V6. These troops were monitored within the Viramba ranges of MNP from January 2020 to December 2024. Throughout the study period, information on troop demographics and behavioural activities was systematically recorded.

Data Collection

Data collection during troop follow-up was conducted daily between 06:30–16:00hrs. Observers recorded data on the troop sleeping sites, rolls including births, deaths, transfers, and the total number of individuals present in each troop. In addition to these daily records, overall troop population size was documented on a monthly to annual basis. Age-sex composition was categorized into adult females, juvenile females, infant females, adult males, subadult males, juvenile males, and infant males. Mortality and transfer events (immigration and emigration) were also tracked on daily basis for the whole study period.

Data Analysis

A descriptive analysis was conducted using Ms Excel and R (Core team, 2023) studio version 4.3.1 to produce frequency distribution graphs on age-sex ratios composition and troop's population trends showing mean troop size across the study period. Net population change was calculated as the number of individuals transferred in minus deaths and transfers out.

RESULTS

Groups Composition

Overall, the V6 troop showed consistently higher mean counts across all age-sex classes compared to V5. For the V6 troop, the mean counts were 8.4 adult females, 5.6 juvenile females, 3.3 infant females, 4.7 adult males, 2.2 subadult males, 8.5 juvenile males, and 1.9 infant males. In contrast, the V5 troop recorded a lower mean for each category: 5.5 adult females, 2.2 juvenile females, 1.8 infant females, 3.0 adult males, 2.2 subadult males, 3.8 juvenile males, and 0.9 infant males (Fig. 2).

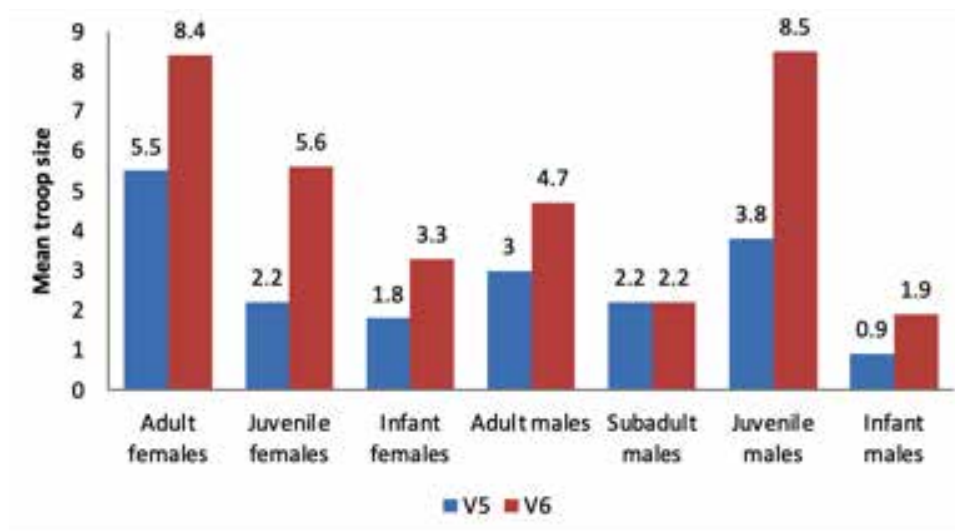


Figure 2: Mean number of individuals per age-sex category for yellow baboon troops V5 and V6 in Mikumi National Park, Tanzania (2020–2024).

Yellow Baboon troops Population Trends

Troop V6 maintained a larger and more stable adult female cohort (mean = 8.4) and exhibited strong population growth, increasing from 30 individuals in January 2020 to 51 individuals by December 2024 (net gain +21), driven primarily by high natality (26 births) and low mortality (3 deaths). In contrast, the V5 troop

began with 19 individuals and showed greater fluctuation, ending with 21 individuals (net gain +7). Despite 19 births, its growth was constrained by higher mortality (11 deaths, mostly infants) and net emigration of 4 adult/subadult males. Over the five years, births were the principal driver of population increase in both troops, whereas mortality and male dispersal exerted opposing effects (Fig. 3 & 4).



Figure 3: Population trends of yellow baboon troops V5 and V6, showing mean monthly troop sizes, 2020–2024.

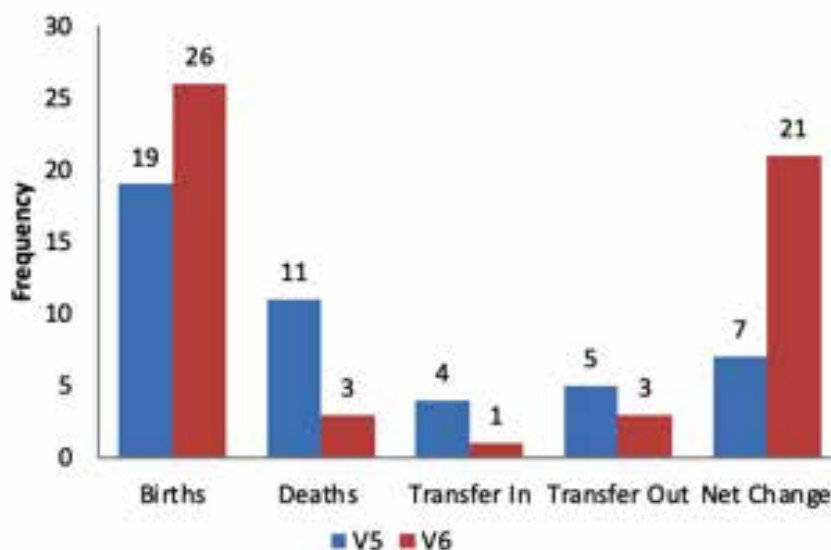


Figure 4: Cumulative mortality and transfer events for yellow baboon troops V5 and V6, 2020–2024.

DISCUSSION

Groups Composition

The findings of this study has shown that the V6's troop consistently had larger adult female cohort and higher juvenile numbers reflecting greater reproductive success and recruitment compared to V5. This is because of the presence of more adult individuals of both sexes in V6 which probably enhanced anti-predator vigilance and reduced per-capita risk, while the greater number of breeding females provided more reproductive opportunities and higher absolute natality (Altmann *et al.*, 2005; Archie *et al.*, 2014; Alberts, 2019). Moreover, the mean ratio of immatures (<4 years) to adult females was markedly higher in V6 (2.7:1) than in V5 (1.8:1), providing a simple yet powerful index of recruitment efficiency (Hausfater & Watson, 1976; Wasser *et al.*, 2004).

These observations agree with those of the long-term studies of yellow baboons in Amboseli National Park, which showed that troops with larger and more stable adult female cohorts exhibit higher birth rates, better infant survival, and stronger demographic resilience (Archie *et al.*, 2014; Alberts, 2019; Altmann *et al.*, 2002). Conversely, troops with fewer adult females or disrupted social bonds suffer reduced reproductive output and slower recovery after disturbances (Altmann *et al.*, 2005; Gesquiere *et al.*, 2018). The marked juvenile surplus in V6 troop and deficit in V5 troop fits this pattern perfectly, emphasizing the critical role of female cohort stability in sustaining recruitment under savannah conditions (Smuts, 1985; Strum, 2012).

Previous studies have shown that Yellow baboons exhibit female philopatry and male dispersal (Altmann, 1980; Smuts, 1985), processes that continually reshape group composition, dominance hierarchies, and reproductive strategies (Strum, 2012; Tung *et al.*, 2023). Troop size and structure are further influenced by habitat quality, predation pressure, and anthropogenic factors (Cheney

& Wrangham, 1987; Dunbar, 1988), with fissioning and temporary fusions reorganizing matrilineal across the landscape (Rasmussen, 1981; Norton, 1987). In MNP, clear juvenile and subadult cohorts reflect recent reproductive success and survival rates, with seasonal resource pulses and ecological stress producing visible cohort effects (Norton, 1987; Rhine, 2000). Together, female longevity and immature survival emerge as principal determinants of age-structure stability and population recovery potential (Archie *et al.*, 2006; Gesquiere *et al.*, 2018).

Yellow Baboon: Troop Population Trends

Over the five years in MNP, births were the primary driver of population growth in both troops, but their impact was heavily modulated by mortality and dispersal (Altmann *et al.*, 2005; Archie *et al.*, 2014; Alberts, 2019). Although the V5 troop produced 19 births, high mortality (11 deaths, mainly infants) and net loss of four dispersing males cancelled most gains. In contrast, the V6's troop with 26 births was met by only three deaths and a net influx of eight immigrant males, yielding strong positive growth. Notably, per-capita birth rates were virtually identical between the troops (0.68–0.70 births per adult female per year), confirming that the starkly different growth trajectories were driven almost entirely by differences in infant survival and male transfer dynamics rather than fertility itself (Altmann *et al.*, 2005; Gesquiere *et al.*, 2018; Alberts, 2019).

Births were concentrated in the wet season in both troops, confirming that resource abundance and female body condition remain key triggers of reproduction (Altmann, 1980; Archibald *et al.*, 2006). Roadkill and other direct anthropogenic mortality were negligible for these particular troops, suggesting that ranging behaviour and habitat use currently buffer them from the intensifying road impacts observed elsewhere in the park (Drews, 1991; Georgiev *et al.*, 2022; Kitegile *et al.*, 2022).

Long-term studies in Amboseli and Gombe have similarly shown that yellow baboon populations can remain stable for decades, yet local troops frequently experience pronounced fluctuations driven by infant mortality, predation, and dispersal patterns (Cheney & Wrangham, 1987; Altmann *et al.*, 2002; Alberts *et al.*, 2005). Additionally, climatic stress, disease, and infrastructural development can all impair demographic stability (Newmark & Boshe, 1996; Gesquiere *et al.*, 2018), but recruitment success and low mortality ultimately determine resilience (Archie *et al.*, 2006). Therefore, our results reinforce these conclusions and highlight adult female cohort size as the single best predictor of a troop's ability to convert favourable conditions into sustained growth (Archie *et al.*, 2014; Alberts, 2019).

CONCLUSION AND RECOMMENDATIONS

This study highlights that the population structure and demographic stability of yellow baboon troops are strongly influenced by ecological conditions, social organization, and habitat quality. The dominance of adult females and large juveniles is central towards recruitment within troops. These differences emphasize ecological heterogeneity and habitat quality as key determinants of troop performance and long-term viability. From a management perspective, maintaining ecological integrity within the baboons' range is essential for sustaining population resilience. Strategies should prioritize the protection of high-quality foraging zones, regulation of tourism and road traffic to minimize disturbance, and monitoring of troop composition as an indicator of ecosystem health. Long-term population monitoring, coupled with habitat management and community engagement, will be critical for ensuring that yellow baboon populations remain stable under changing environmental and anthropogenic pressures. These findings underscore the importance of sustainable management of primate populations which requires an integrated approach that

combines habitat conservation, ecological monitoring, and mitigation of human impacts to ensure optimal age-sex structure and favourable habitat conditions for maintaining the demographic and ecological stability of yellow baboons.

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ASSESSMENT OF ABUNDANCE AND DISTRIBUTION OF INVASIVE ALIEN PLANT SPECIES IN PUGU – KAZIMZUMBWI NATURE FOREST RESERVE (PKNFR), TANZANIA

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ABSTRACT

The coastal forests of Tanzania are among the global biodiversity hotspots, harbouring dozens of narrow endemic species. However, the biodiversity found in these forests area currently threatened by the spread of invasive alien plant species (IAPS). This study assessed the abundance and distribution of IAPS in Pugu-Kazimzumbwi Nature Forest Reserve (PKNFR) located in Kisarawe district, Tanzania, to support efforts in combating these (IAPS). A combination of three methods (a 14-day reconnaissance survey, systematic vegetation sampling across an area of 2,410 ha in Pugu forest using 58 circular sample plots and 4,887 ha in Kazimzumbwi forest using 77 circular sample plots, and spatial analysis) was employed in the assessment. The results indicated that 19 IAPS were recorded in Pugu forest and 14 IAPS in Kazimzumbwi forest. In Pugu forest, the most dominant and distributed species were *Lantana camara* (with a relative frequency of 15.1%) and *Senna siamea* (RF 9.6%), followed by *Leucaena leucocephala* with RF 5.5%, and the remaining 16 plant species with the RF $\leq 5.4\%$. *Lantana camara* and *Leucaena leucocephala* shows to form distinct clusters, particularly along disturbed edges and near built-up areas, suggesting a preference for areas with anthropogenic influence or reduced forest cover. Likewise, in the Kazimzumbwi forest, *Lantana camara* was identified as the most dominant species, accounting for over 70 % of IAPS stems with RF of 14.3%, followed by *Senna siamea* with the RF of 12.5 and *Leucaena leucocephala* with RF 8.9%. IAPS were most frequently observed to exhibit clustering in specific parts of the reserve, often aligning with areas of disturbed land cover such as grasslands and edges of built-up areas. These results demonstrate that the Kazimzumbwi forest remains at an early to moderate stage of invasion, while the Pugu forest is moderate. Thus, early intervention and adaptive management are essential to maintain the reserve's ecological integrity.

Keywords: *Biodiversity loss, Coastal forests, Human disturbances, Invasive species, Tanzania Forest Services Agency.*

INTRODUCTION

Tanzania's forest reserves and natural forests are vital ecosystems that harbor rich biodiversity, offering sanctuary to numerous species, including endemic flora and fauna. These reserves play essential roles in climate regulation, water conservation, and supporting local communities while fostering ecotourism

and sustainable development (Ract *et al.*, 2023). Despite offering these benefits, the natural ecological system faces a high rate of species invasion of invasive alien species (IAS), including herbaceous and woody plants, thus devastating the stability of productivity of the ecosystem (Ngondya *et al.*, 2017; NISSAP 2019; IUCN, 2020). Among the major impacts associated with the invasive species in the

country include loss of biodiversity, where in some cases has caused disappearance of certain species of fauna and flora, social and economic distress to human being due to the disappearance of plants and animal species that are regarded as important sources of livelihoods (Lymo *et al.*, 2009; IUCN, 2020; Adolfo Perdomo-González *et al.*, 2023). The IAS pose negative impacts on water catchment areas while also suppressing the native plant species composition and structure (Ferraguti *et al.*, 2023), loss of biodiversity, and the alteration of ecosystem functions (NISSAP 2019; Rai and Singh, 2020; Kacheche & Mzuza, 2021; Nininahazwe *et al.*, 2023). However, inadequate information exists on the diversity, density, and distribution of these noxious weeds in the reserves. Assessing the impacts of IAS is a major task in invasion science and vitally important for supporting invasion-related policies (Bartz & Kowarik, 2019). IAS are exotic, non-native, non-indigenous, or foreign plants that have been introduced by people intentionally or unintentionally, into habitats outside of their natural range and beyond the reach of their natural dispersal potential, and in their new environment. They proliferate to the detriment of native biodiversity and local livelihoods and economic activities (Witt & Luke, 2017). However, many studies have reported the presence of alien invasive plants, while very little attention has been paid to determine their existence and map their distribution at large within the protected areas (2008). It has been stated that Kazimzumbwi, together with Pugu Nature Forest Reserves are poorly studied (Philbert & Nyundo, 2024).

Several studies have been conducted at PNFR such as Howell (2008) who described the vegetation type and biodiversity composition, Beda (2013) who dealt with carbon potential, Mwegime *et al.*, (2013) who explored the flora and fauna, and Boussougou *et al.* (2018) who assessed the anthropogenic threats to the Pugu ecosystem. However, none of the above studies explored the extent and spread of invasive alien plant species (IAS) in the area. Kazimzumbwi Nature Forest Reserve has also been reported

for the conspicuous presence of IAS, while little has been done to map their existence in the area. This study, therefore, intended to identify and map the location and distribution of IAS at PKNFR. This would help in setting strategies to evaluate the diversity, distribution, and structure within the study area. It would also, help in planning how to manage them when such a need arises. The study was guided by the following research Questions: What is the list of IAS found in PKNFR? and How are IAS distributed in the PKNFR?

MATERIALS AND METHODS

Study area location

Pugu-Kazimzumbwi Nature Forest Reserve (KNFR) is two coastal forests remnants located in Kisarawe District, Pwani Region in the eastern part of Tanzania (Fig. 1). Both reserves are part of the coastal vegetation forests of Tanzania and host unique species assemblages. Pugu forest spans roughly 2,410 ha (24.1 km²), while Kazimzumbwi forest covers 4, 887 ha. Pugu forest was gazetted in 1954 and up-graded to Nature Forest Reserve status in 2016. The forest is under the management of the Tanzania Forest Service Agency (TFS), under the Ministry of Natural Resources and Tourism (MNRT). It is located in the west of Dar es Salaam city, centred at $\approx 06^{\circ}55' \text{ S}$, $39^{\circ}13' \text{ E}$ at Elevations undulating from $\approx 100 \text{ m}$ to 330 m asl., forming the northern flank of an ancient kaolinitic-sandstone plateau that harbours one of the world's oldest surviving coastal forests. The substrate, deep red sandy clays over kaolin beds, has long attracted small-scale clay and phosphate mining, visible today as abandoned pits in the south-eastern sector. The climate is warm-humid tropical with a bimodal rainfall regime: "long rains" (*Masika*, March–May) and "short rains" (*Vuli*, Oct–Dec). Mean annual precipitation is about 1000–1160 mm, and mean daily temperatures range 24 to 30 °C. Five broad vegetation formations occur: (i) semi-deciduous coastal forest on ridge tops, (ii) evergreen thicket in moist gullies, (iii) open bushland/woodland mosaics, (iv)

wooded grassland and secondary scrub on fire-prone slopes, and (v) small bamboo stands and riparian swamp forest along the Minaki stream. Pugu forest is floristically renowned for at least 14 endemic or near-endemic plants, including *Baphia puguensis* and *Milletia puguensis*, and supports threatened fauna such as the Zanzibar colobus (*Piliocolobus kirkii ssp. tanaeus*), Pugu galago, and over 80 bird species. Extensive bat colonies inhabit old phosphate tunnels and natural limestone caves. Pugu forest faces intense peri-urban pressure: charcoal production, smallholder farms, sand and clay extraction, and rapid settlement along its eastern boundary. Together with Kazimzumbwi NFR to the south, it forms the Pugu–Kazimzumbwi forest block.

Kazimzumbwi forest, on the other hand, was gazetted in 1936 and upgraded to Nature Forest Reserve status in 2016. It lies on the gently undulating coastal hinterland 20 km south-west of Dar es Salaam city, within

Kisarawe District, Pwani Region (06°54' S, 39°09' E) (Fig. 1). The reserve covers 4 887 ha (48.8 km²) and, together with the adjacent Pugu forest, protects remnants of an ancient Miocene kaolinitic sandstone forest, considered one of the oldest surviving blocks of coastal forest worldwide. Altitude ranges from 120 m to 280 m asl. The geology comprises red to brown sandy-clay soils over kaolinitic sandstone (pH 5 - 6), yielding well-drained but nutrient-poor substrates. Kazimzumbwi forest experiences a bimodal rainfall regime: short rains (*vuli*, Oct–Dec) and long rains (*masika*, March–Jun). Mean annual rainfall ranges from $\approx$ 1 000 mm in the western sector to 1400 - 1600 mm in the east, with mean daily temperatures of 24 - 28 °C. Kazimzumbwi forest harbours numerous coastal-forest endemics (e.g., *Baphia puguensis*, *Milletia puguensis*) and threatened fauna such as the White-throated Colobus and Tanzanian. Kazimzumbwi forest faces threats of encroachment from agriculture, charcoal, and rapid peri-urban settlements.

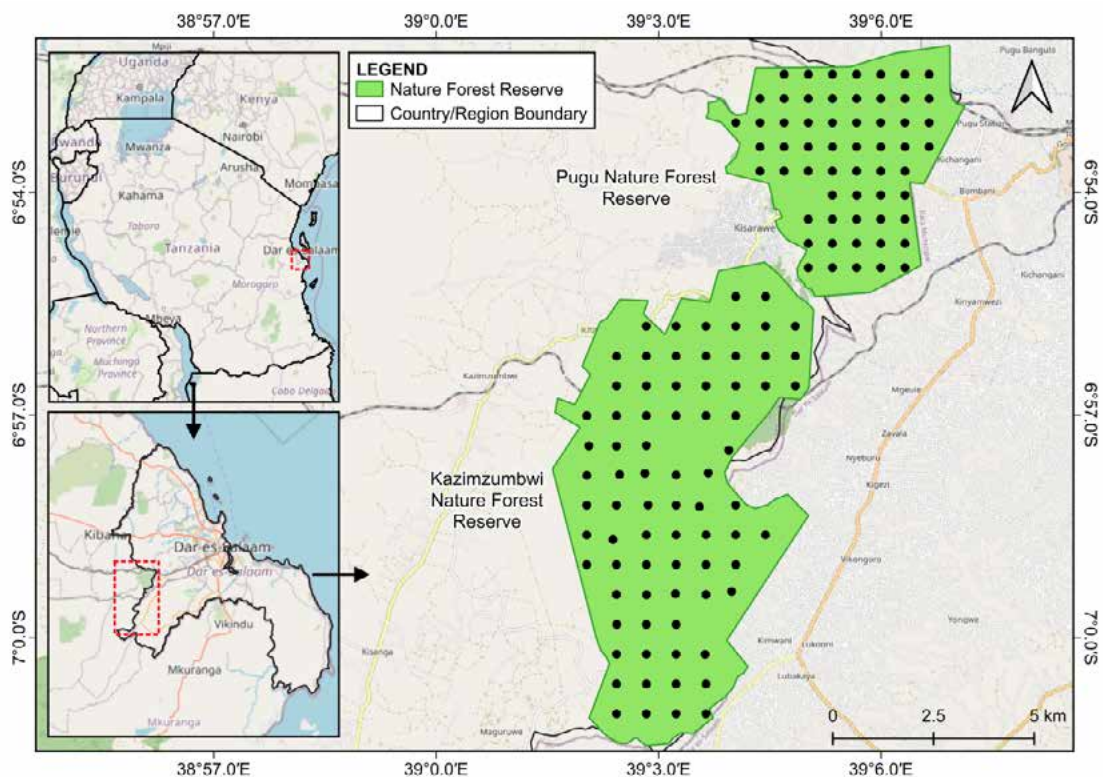


Figure 32. Map of Tanzania showing the location of Pugu-Kazimzumbwi nature forest reserve and distribution of field sample plots in both forests.

Data collection

Desktop Review and Reconnaissance Survey

The study combined a desktop review and reconnaissance survey with a systematic plot inventory, which recorded all trees, shrubs, and herbs, and identified IAS. Field observations were then integrated with location data to produce spatial maps of IAS hotspots to pinpoint the areas where management action is most urgently needed. A targeted literature scan (Frontier 1993; TFCG 2013; Mdemu & Burra 2016; Frontier 2002) and stakeholder discussions with TFS personnel and village scouts provided baseline information on likely IAS hotspots and access routes. An initial two-week reconnaissance in December 2024 validated track conditions and examined known exotic tree plantings and previously reported alien hotspots. During the survey, the research

team purposely visited the already known area to have been planted with exotic plant species by the forest authority as part of a restoration activity focusing on those areas that were badly damaged by human activities (Palys, 2008). The planted shrubs or trees were identified and recorded for their botanical and common names. Also, the survey team extended the search area to a distance of 300 m in four cardinal directions (north, east, south, and west) from the planted area and recorded all encountered alien plant species and marked their locations using GPS (Waheed *et al.*, 2023) (Fig. 2). Geospatial data to map IAS distribution by species (Rajakumari *et al.*, 2010) and coverage in each NFR were recorded. Photographs were taken for each alien plant for pictorial representation of the existing alien taxa (Mesaglio *et al.*, 2023). Furthermore, the vegetation type in which these plants were encountered was also described.



Figure 33: An example 300 m buffer zone from a target point (planted) during data collection

Plot Layout and Sampling Design

A systematic sampling design was used, with plots spaced at 600–700-meter intervals. In total, 80 and 60 plots were planned in Kazimzumbwi and Pugu, respectively. Due to encroachment (recently converted to settlements - “built-up” areas) and inaccessibility, 77 plots (0.07 ha

concentric design: 2 m, 5 m, 10 m, 15 m radii) were surveyed in Kazimzumbwi forest and 58 in Pugu forest (Fig. 1). Each plot measured 15 m radius and was georeferenced using UTM coordinates. Opportunistic plots were added where reconnaissance flagged dense IAS clusters.

Data Collection Protocols

Within each sampling plot the team recorded: Dominant vegetation type, Trees ≥ 1 cm DBH species, DBH (cm), status (alien/indigenous), life-form, count, Understorey (< 1 cm) seedling counts by species inside the 2 m sub-plot, Photographs of alien plants for pictorial representation of the existing alien taxa, and Plot attributes: UTM coordinates (in EPSG 21037), elevation (m asl), slope ($^{\circ}$), canopy cover (% spherical densitometer), litter depth (cm), disturbance category (Undisturbed, Low, Moderate, High). Botanical determinations to the species level were conducted with the aid of a botanist; doubtful taxa were photographed for additional expert verification.

Data Analysis and Processing

The identified alien plants were assessed for whether they were invasive or noninvasive as per Rai and Singh (2020). Invasive species were those that show the potential to bring changes to the native plant species and extended beyond the borders of where they were introduced, while noninvasive species were those which cannot spread aggressively beyond the introduction boundary. The frequency (F) and relative frequency (RF) were used to determine the most spreading alien plant species. The recorded coordinates were used to map their distribution in PKNFR. Field data were entered into Microsoft Excel, cleaned and analysed. All GIS operations were completed in QGIS 3.42.1 (QGIS.org, 2025) with field-collected UTM coordinates.

RESULTS

Presence of Alien Plant Species in PKNFR

A total of 42 alien plant species were recorded at Pugu Nature Forest Reserve (PNFR) (Tables 1 & 2). Of the 42 alien plant species, 19 (45.2%) were IAS (Table 1), and the remaining 23 (54.8%) were just alien but not invasive (NI) (Table 2). Of the recorded 19 invasive alien plant species, *Lantana camara* and *Senna siamea* were the most distributed in PNFR, with the RF of $\leq 15.1 - \geq 9.6$, followed by *Leucaena leucocephala* with RF of $\geq 5.5 - \leq 9.5$, and the

remaining 16 plant species with the RF ≤ 5 . (Appendix 1).

In the second group of 23 noninvasive plants (NI) (Appendix 2), the most occurring plant species were *Azadirachta indica* and *Elaeis guineensis* with the RF of ≥ 5.5 , followed by *Artocarpus heterophyllus*, *Delonix elata*, and *Gmelina arborea* with the RF of $\geq 2.7 - \leq 5.4$, and the remaining 19 species were the least occurring within the sampled areas with the RF of ≤ 1.5 (Appendix 2). Table 1 shows the distribution of alien plant species per surveyed site. The highest number of alien plant species was observed to occur at the Minaki tourist gate to Minaki Pond (20), and the least were at Kimani and Bangulo (2).

Distribution of alien plant species

The mapped distribution of alien and invasive plant species (IAPS) within Pugu Nature Forest Reserve is presented in Fig. 3. The map illustrates the spatial extent and clustering of invasive alien plant species (IAPs) relative to land-cover types, including forest, grassland, built-up areas, and other land uses. By visualizing species occurrences, the map provides insight into potential invasion hotspots within the reserve. From the map, distinct clusters of species such as *Leucaena leucocephala* and *Lantana camara* are observed, particularly along disturbed edges and near built-up areas. The distribution suggests a preference for areas with anthropogenic influence or reduced forest cover. Conversely, certain species, such as *Cedrela odorata* and *Eucalyptus spp.*, are sparsely distributed, possibly indicating isolated introductions. The map identifies > 20 species, indicating the diversity and potential ecological impact of IAPS within the nature forest reserve. The proximity of clusters to major access routes and settlements suggests these areas may serve as entry points for invasive species. Species-specific distribution maps of the most dominant species have been prepared to provide further detail, as can be seen in Fig. 4. These maps allow for focused analysis of individual species and can potentially support targeted management interventions.

Table 1: Total alien plant species per surveyed site in PNFR

Site name	TTPS (total plant species)	Ranked position
Minaki Tourist gate to Minaki pond	20	1
Mwambisi Mashimo ya Popo	14	2
Pugu Mnadani	9	3
Kinyanyiko	5	3
Mambisi	5	4
Golani	5	3
Msimbazi river near SGR	4	5
Pugu/Kazimzubwi Tourism office	3	6
Mwambisi natural vegetation	3	6
Kimani (near Kimani settlement)	2	7
Bangulo hill	2	7
Total	72	-

In Kazimzumbwi Nature Forest Reserve, a total of 26 alien plant species were recorded (Table 2 & Appendix 3) from the eight (8) sites, which were (i) Vigama, (ii) Maguruwe ranger post, (iii) Dahu (iv) Kimani, (v) Vibula, (vi) Maguruwe village, (vii) Nyeburu, and (viii) Kimwani (Table 3). Of the recorded 26 alien plant species, some of them occurred more than the others, and *Lantana camara* was

known to occur in all eight sites with the RF of 14.3, followed by *Senna siamea* with the RF of 12.5. *Leucaena leucocephala*, *Mangifera indica*, *Senna siamea*, and *Tectona grandis* were the most common alien plant species with the RF of $\leq 14 - \geq 7.1$, while the rest got the RF of ≤ 7.0 , and of all those, 14 species (52%) were invasive and 12 (42%) were just alien but not invasive (Appendix 3).

Table 2: Total plant species for invasive alien (IAS) and alien not invasive (NI)

Category	T	S	WC	H	A	F	RF	I	NI
Invasive alien species	7	3	1	3	14	35	62.3	14	0
Alien not invasive (NI)	10	0	0	2	12	21	37.7	0	12
	17	3	1	5	26	56	100	14	12

Key: *T* = tree; *S* = shrub; *WC* = woody climber; *H* = herb; *A* = alien; *F* = frequency; *RF*; relative frequency; *I* = invasive; *NI* = not invasive

Table 3: Total alien plant species per site

Sampled site name	TAS	Rank	Remarks/notes
Maguruwe ranger post	18	1	<i>Gmelina arborea</i> and <i>S. siamea</i> eviction area
Bonde la Mpunga	15	2	Eviction area
Maguruwe village	7	3	Planted at former forest guide house
Dahu	6	4	Signs of very old settlement

Vigama	4	5	<i>Tectona grandis</i> & signs of very old settlement
Nyeburu	3	6	Eviction area
Vibula	2	7	Heavily degraded previously
Kimwani hill	2	7	Jumped in

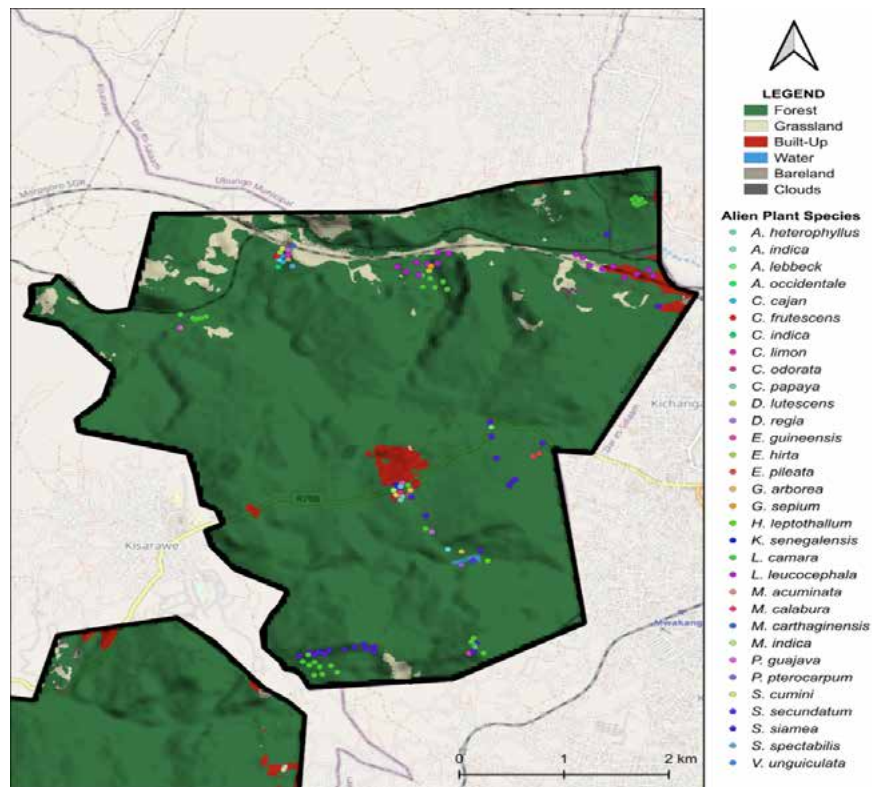


Figure 3: Distribution of alien plant species in the Pugu Nature Forest Reserve (PNFR)

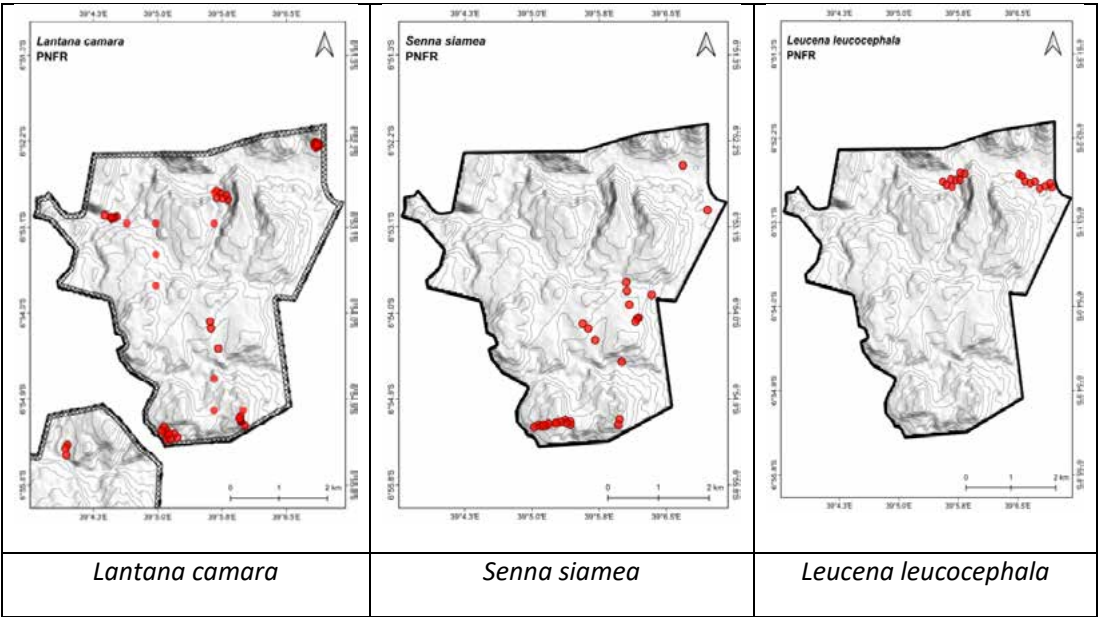


Figure 4: Species specific alien plant distribution maps in Pugu forest reserve

In the Kazimzumbwi Nature Forest Reserve the mapped distribution of alien and invasive alien plant species (IAPS) is presented in Figure 5. This map describes the spatial extent and clustering of different IAPS across various land cover types within the reserve, including

forested areas, rasslands, built-up zones, and other land uses. The species locations are overlaid on a land cover lassification to provide context for their occurrence and potential ecological interactions.

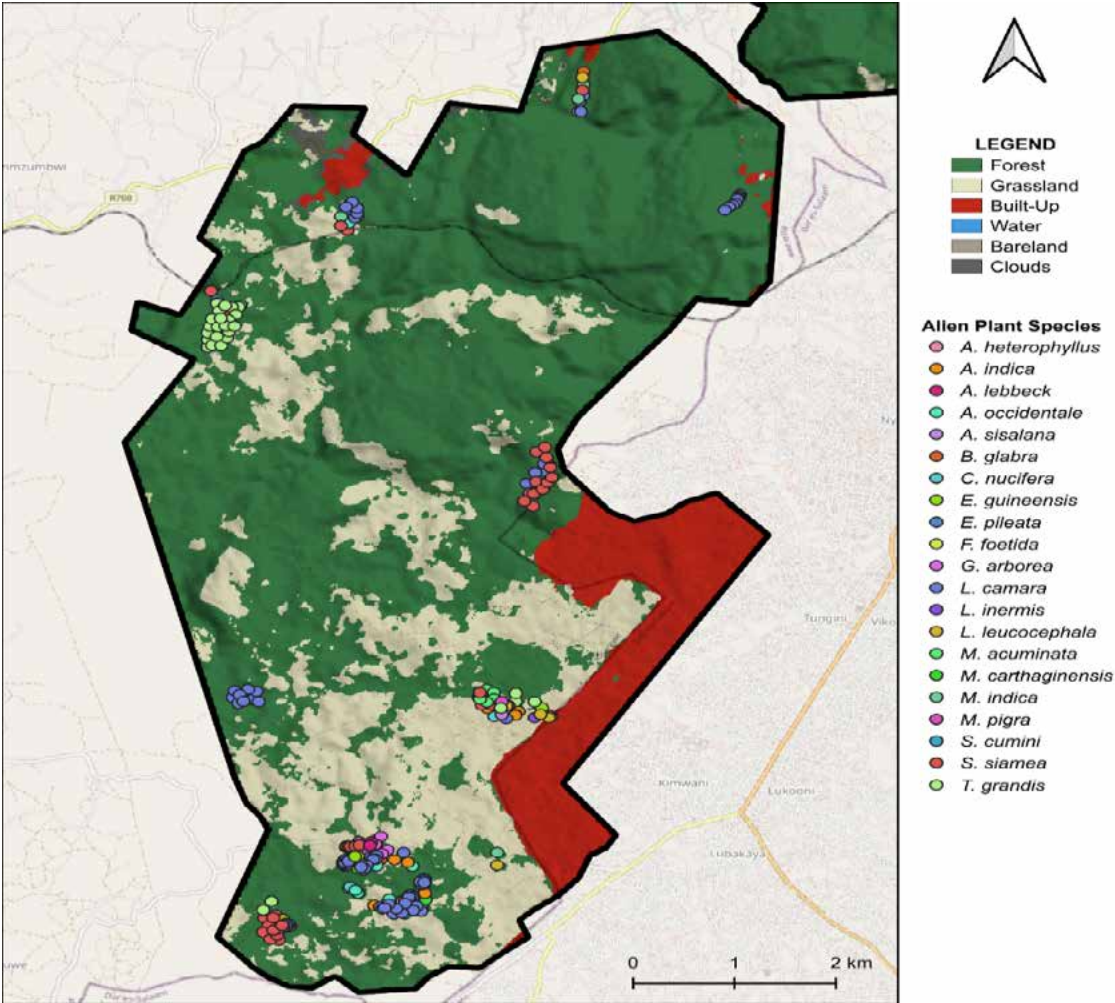


Figure 5: Mapped distribution of alien and invasive alien plant species (IAPS) in the Kazimzumbwi Nature Forest Reserve

From the map, it can be observed that certain species exhibit clustering in specific parts of the nature forest reserve, often aligning with areas of disturbed land cover such as grasslands and edges of built-up areas. This pattern may suggest that these species are exploiting disturbed habitats, which are often more vulnerable to invasion. Distinguished concentrations of species such as *Leucaena leucocephala*, *Anacardium occidentale*, and

Tectona grandis can be observed in areas with less forest cover, suggesting possible preferences for open or semi-open habitats. For a detailed overview, species-specific distribution maps of the dominant invasive species have been generated and presented in Figure 6. These maps provide finer detail on the spatial extent and density of individual species, which may be useful for targeted management interventions.

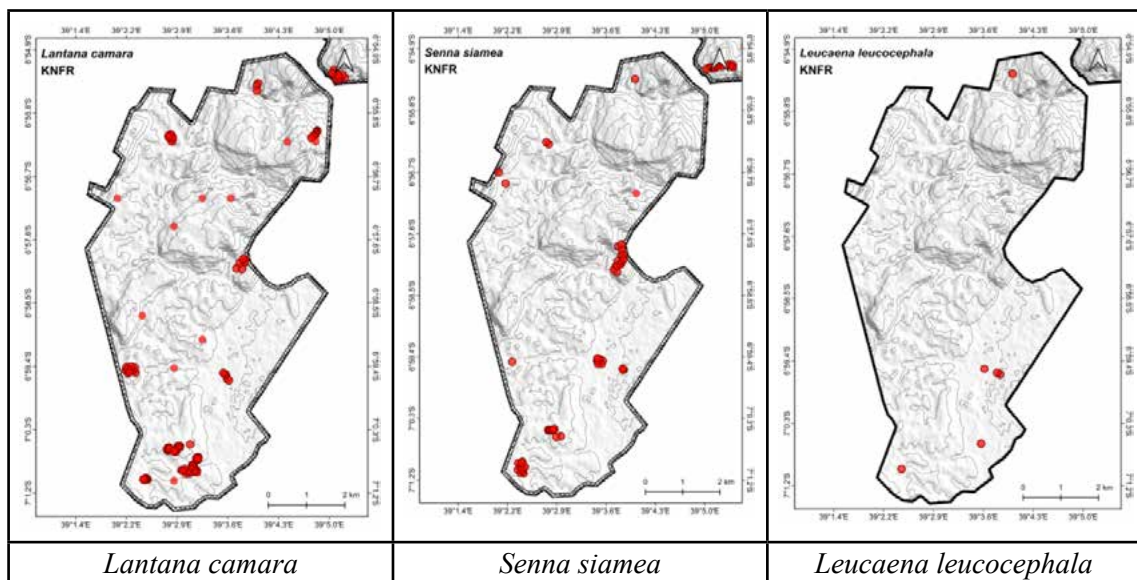


Figure 6: Species specific alien plant distribution maps in the Kazimzumbwi forest reserve

DISCUSSION

Out of the total 19 recorded invasive species in Pugu forest reserve, *Lantana camara* and *Senna siamea* were the most distributed (occurred) species. *Lantana camara* L. is widely distributed in the East African region and can be found in arid regions, and is known to pose a threat to other biodiversity (Stegelmeier & Cook, 2013). According to Sharma *et al.* (2005), *Lantana camara* L. has been considered to be among the world's 10 worst weeds. *Lantana* occurs in diverse habitats and on a variety of soil types, and its spread is encouraged by animal activities and by human disturbances, such as cultivation, road construction, and changes in fire regimes.

Studies have revealed that IAS plant species like *S. siamea* is often planted at gates, roadsides, vehicle parks and staff and tourist accommodation as a useful and attractive tree, but the risk of spread from these areas is significant as it has the ability to become invasive and compete with native vegetation especially in savannah areas (International Union of Conservation of Nature (IUCN, 2013). It has been stated that on average, abundance and diversity of the resident species decreased in invaded sites (Via *et al.*, 2011). The higher

number of species at tree nursery, along roads, and the Minaki pond infrastructure, planted patches at forest gaps implied how the planting focused mostly on the alien plants while less effort was put in selection of native plants. The full economic costs of invasions include not only the direct damage and control costs but also the effects on the ecosystems themselves (Panzavolta *et al.*, 2021). The ecological and environmental impacts of alien invasive species can be felt by all levels of organization including the gene, species, habitat and ecosystem level (Meyerson *et al.*, 2007).

The not invasive (NI) plant species, apart from being mature and producing seeds did not show signs of spreading beyond the borders of where they were introduced. It has been pinpointed that when population sizes and genetic diversity of native species are large enough, natives may be able to evolve traits that allow them to co-occur with invasive species, and also, native species may evolve to become significant competitors with invasive species, and thus affect the fitness of invaders. Invasive species may respond in turn, creating either transient or continuing coevolution between competing species (Leger & Espeland, 2010). However, in general the presence of any alien plant species essentially destroying the environment by harboring rogue

plants that are sapping (weakening) the life out of everything around them (Boy & Witt, 2013). Understanding the potential distribution area of biological species is vital for effective conservation efforts and setting strategies of mitigating its impacts to ecological ecosystem (Wang *et al.*, 2024).

In Kazimzumbwi forest reserve, alien plants as it has been widely defined to mean exotic or non-native organisms that occur outside their natural adapted ranges; a few of them may become invasive (Sekar, 2011; Skocajic & Nesic, 2020). Some of the plant species such as *Tectona grandis*, may not be extending beyond the boundaries of the area it was introduced, but rather impede the regeneration of several native plant species, thus limiting the abundance, height growth, diversity, and growth of native plants (Healey & Gara, 2003). The identified 26 alien plant species with *Lantana camara* occurring in all sampled areas implied how the reserve is highly invaded by such species. The differences in the number of alien plant species (Appendix 3), implied that some of the areas were more occupied than others, even though the number of species did not imply higher invasion than areas with lower number of species because it depended on how a particular species spread in the area. The massive sprouting of live tree stumps implied how severe the area was previously degraded, thus also contributing to spreading of alien plant species which prefer disturbed land or vegetation.

CONCLUSION AND RECOMMENDATIONS

PKNFR is potential in terms of biological diversity even though is facing high rate of alien plant species including the only alien and the invasive alien plant species competing with the native species. Nature reserve is expected to accommodate most of the indigenous biological taxa to attract tourists and conservation of native gene pool. Some of the alien plants were deliberately introduced to fill the open and or damaged areas by people, while others are

spreading by other agents such as wind, rain water, and fauna (eg. Birds). *Lantana camara* is widely distributed in PKNFR at previously disturbed areas and edges, including even where anthropogenic disturbances have not yet taken place. It is thus recommended that further research on biodiversity composition and structure is needed. This can include recording all plant species, determining plant species diversity, basal area, volume, biomass, and carbon. The existing anthropogenic threats should be investigated to be aware of the continuing human activities within such hotspot area. Tree planting should focus on native plant species to maintain the natural scenery. Conservation of live stumps is important because the damaged ecosystem can recover quicker than planting seedlings which when there is shortage of rain, most of them die easily. The existing live stumps, apart from being conserved can be singled (singling) to remain with few stems which will grow quickly. The areas where infrastructures like offices, roads, walk trails have been constructed must be visited regularly and remove any emerging unintended alien plant species.

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APPENDICES

Appendix 1: Invasive alien plant species (IAS) recorded in Pugu Nature Forest Reserve (PNFR)

S/N	Botanical name	Author	Family	Common name	T	S	H/G	A	F	RF
1	<i>Lantana camara</i>	L.	Verbenaceae	East indian welnut	0	1	0	1	11	15.1
2	<i>Senna siamea</i>	(L.) H.S.I & Barn.	Fabaceae	Yellow cassia	1	0	0	1	7	9.6
3	<i>Leocaena leucocephala</i>	(L.)De Wit	Fabaceae	River tamarind	1	0	0	1	4	5.5
4	<i>Muntingia calabura</i>	L.	Muntingiaceae	Jamaica cherry	1	0	0	1	2	2.7
5	<i>Psidium guajava</i>	(L.) Leaf	Myrtaceae	Guava/Mpera	1	0	0	1	2	2.7
6	<i>Stachytarpheta jamaicensis</i>	(L.)Vahl.	Verbenaceae	Blue portedweed	0	0	1	1	2	2.7
7	<i>Calotropis procera</i>	(Ait)Ait.f.	Asclepiadaceae	Sodom apple	0	1	0	1	1	1.4
8	<i>Canna indica</i>	L.	Cannaceae	Wild canna lily	0	0	1	1	1	1.4
9	<i>Cedrela odorata</i>	L.	Meliaceae	Spanish cedar	1	0	0	1	1	1.4
10	<i>Eucalyptus pellita</i>	F.Muell	Myrtaceae	Red mahogan	1	0	0	1	1	1.4
11	<i>Euphorbia hirta</i>	L.	Euphorbiaceae	Milk weed	0	0	1	1	1	1.4
12	<i>Mimosa pigra</i>	L.	Fabaceae	Giant sensitive plant	0	1	0	1	1	1.4
13	<i>Peltophorum pterocarpum</i>	(DC.)K.Heyne	Fabaceae	Copperpod	1	0	0	1	1	1.4
14	<i>Ricinus communis</i>	L.	Euphorbiaceae	Castor oil tree	1	0	0	1	1	1.4
15	<i>Scoparia dulcis</i>	L.	Scrophulariaceae	Sweet broomweed	0	0	1	1	1	1.4
16	<i>Senna spectabilis</i>	(DC.)H.S.Ir.&Barn,	Fabaceae	American cassia	1	0	0	1	1	1.4
17	<i>Stenotaphrum secundatum</i>	(Walter)Kuntze	Poaceae	Buffalo grass	0	0	0	1	1	1.4
18	<i>Tectona grandis</i>	L.f.	Verbenaceae	Teak	1	0	0	1	1	1.4
19	<i>Tridax procumbens</i>	L.	Asteraceae	Coat buttons	0	0	1	1	1	1.4
Total					10	3	5	19	41	56

Key: TTS=Total species; T = tree; S = shrub; H/G = herb or grass/herb and grass; A = alien; F = frequency; RF = relative frequency.

Appendix 2: Alien but not invasive plant species recorded in Pugu Nature Forest Reserve (PNFR)

S/N	Botanical name	Author	Family	Common name	T	S	H/G	A	F	RF
1	<i>Azadirachta indica</i>	A.Juss	Meliaceae	Neem tree	1	0	0	1	4	5.5
2	<i>Elaeis guineensis</i>	Jacq.	Arecaceae	Oil palm	1	0	0	1	4	5.5
3	<i>Artocarpus heterophyllus</i>	Lam	Moraceae	Jack fruit tree	1	0	0	1	2	2.7
4	<i>Delonix elata</i>	(Hook.)Raf.	Fabaceae	Roya poinciana	1	0	0	1	2	2.7
5	<i>Gmelina arborea</i>	Roxb	Verbenaceae	White teak	1	0	0	1	2	2.7
6	<i>Albizia lebbek</i>	(L. Benth	Fabaceae	East indian welnut	1	0	0	1	1	1.4
7	<i>Anacardium occidentale</i>	L.	Combretaceae	Cashewnut/Mkorosho	1	0	0	1	1	1.4
8	<i>Cajanus cajan</i>	Druce	Fabaceae	Pigeon pea	0	1	0	1	1	1.4
9	<i>Capscum frutescens</i>	L.	Solanaceae	Pepper/Pilipili kichaa	0	0	1	1	1	1.4
10	<i>Carica papaya</i>	L.	Caricaceae	Pawpaw	1	0	0	1	1	1.4
11	<i>Chrysalidocarpus lutescens</i>	(H.Wendl.)B&J.D	Arecaceae	Areca palm	1	0	0	1	1	1.4
12	<i>Citrus limon</i>	(L.)Burm.F.	Rutaceae	Lemon	1	0	0	1	1	1.4
13	<i>Cocos nucifera</i>	L.	Arecaceae	Coconut tree	1	0	0	1	1	1.4
14	<i>Codeaeum variegatum</i>	(L.)R.exA.Juss	Euphorbiaceae	Garden croton	1	0	0	1	1	1.4
15	<i>Eupatorium capillifolium</i>	Wald.&Kit	Asteraceae	Dogfennel	0	0	1	1	1	1.4
16	<i>Gliricidia sepium</i>	(Jacq.) Walp	Fabaceae	Gliricidia/Mgiricidia	1	0	0	1	1	1.4
17	<i>Lawsonia inermis</i>	L.	Lythraceae	Henna	0	1	0	1	1	1.4
18	<i>Mangifera indica</i>	L.	Anacardiaceae	Mango tree	1	0	0	1	1	1.4
19	<i>Manihot esculenta</i>	Crantz	Euphorbiaceae	Cassava	0	1	0	1	1	1.4
20	<i>Musa acuminata</i>	Colla	Muntingiaceae	Banana plant	0	0	1	1	1	1.4
21	<i>Syzygium cumini</i>	(L.)Skeels	Myrtaceae	Java plum	1	0	0	1	1	1.4
22	<i>Terminalia catapa</i>	L.	Combretaceae	Indian almond	1	0	0	1	1	1.4
23	<i>Vigna unguiculata</i>	L.	Fabaceae	Black eyed pea	0	0	1	1	1	1.4
Total					16	3	4	23	32	44

Key: TTS=Total species; T = tree; S = shrub; H/G = herb or grass/herb and grass; A = alien; F = frequency; RF = relative frequency.

Appendix 3: Alien and invasive alien plant species frequency (F) and relative frequency (RF) of Kazimzumbwi Nature Forest Reserve

S/N	Botanical name	Author	Family	Common name	T	S	WC	H	A	I	NI	F	RF
1	<i>Lantana camara</i>	L.	Verbenaceae	West indian lantana	0	1	0	0	1	1**	0	8	14.3
2	<i>Senna siamea</i>	(L) H.S.Irwin & Barneby	Fabaceae	Yellow cassia/Mjoholo	1	0	0	0	1	1**	0	7	12.5
3	<i>Leucaena leucocephala</i>	(L.) De Wit	Fabaceae	River tamarind/Mlusina	1	0	0	0	1	1**	0	5	8.9
4	<i>Mangifera indica</i>	L	Anacardiaceae	Mango/Mwembe	1	0	0	0	1	0	1	5	8.9
5	<i>Tectona grandis</i>	L.	Lamiaceae	Teak/Mtiki	1	0	0	0	1	1*	0	4	7.1
6	<i>Agave sisalana</i>	Perrine	Asparagaceae	Sisal/mkonge	0	1	0	0	1	1**	0	2	3.6
7	<i>Anacardium occidentale</i>	L.	Anacardiaceae	Chew nut/Mkorosho	1	0	0	0	1	0	1	2	3.6
8	<i>Azadirachta indica</i>	A. Juss	Meliaceae	Neem Tree/Mwarobaini	1	0	0	0	1	0	1	2	3.6
9	<i>Cocos nucifera</i>	L.	Arecaceae	Coconut/Mnazi	1	0	0	0	1	0	1	2	3.6
10	<i>Elaeis guineensis</i>	Jacq	Arecaceae	African oil palm/ Mchikichi	1	0	0	0	1	0	1	2	3.6
11	<i>Lawsonia inermis</i>	L.	Lythraceae	Henna Tree	1	0	0	0	1	0	1	2	3.6
12	<i>Acacia auriculiformis</i>	A.C ex G.Godon	Fabaceae	Red eye wattle	1	0	0	0	1	1**	0	1	1.8
13	<i>Albizia lebbeck</i>	(L.) Benth	Fabaceae	East Indian walnut	1	0	0	0	1	0	1	1	1.8
14	<i>Artocarpus heterophyllus</i>	Lam	Moraceae	Jackfruit/Mfenesi	1	0	0	0	1	1	0	1	1.8
15	<i>Bougainvillea spectabilis</i>	Willd	Nyctaginaceae	Great bougainvillea	0	0	1	0	1	1**	0	1	1.8
16	<i>Eucalyptus pellita</i>	F. Muell	Myrtaceae	Red mahogany/Mkaratusi	1	0	0	0	1	1**	0	1	1.8
17	<i>Furcraea foetida</i>	L	Asparagaceae	Mauritius hemp	0	0	0	1	1	1**	0	1	1.8
18	<i>Gmelina arborea</i>	Roxb. ex Sm.	Lamiaceae	White teak/Teak nyeupe	1	0	0	0	1	0	1	1	1.8
19	<i>Manihot glaziovii</i>	(Mull.Arg) Allen	Euphorbiaceae	Tree cassava/Kisamvu	1	0	0	0	1	0	1	1	1.8
20	<i>Mimosa pigra</i>	L.	Fabaceae	Sensitive giant plant	0	1	0	0	1	1**	0	1	1.8
21	<i>Musa acuminata</i>	Colla	Musaceae	Banana/Mgomba	0	0	0	1	1	0	1	1	1.8
22	<i>Musa xparadisiaca</i>	L	Musaceae	Banana/Mgomba	0	0	0	1	1	0	1	1	1.8
23	<i>Psidium guajava</i>	L.	Myrtaceae	Guava/Mpera	1	0	0	0	1	1**	0	1	1.8

S/N	Botanical name	Author	Family	Common name	T	S	WC	H	A	I	NI	F	RF
24	<i>Richardia brassiliensis</i>	Gomes	Rubiaceae	White eye	0	0	0	1	1	1**	0	1	1.8
25	<i>Scoparia dulcis</i>	L.	Scrophulariaaeae	Licorice weed	0	0	0	1	1	1**	0	1	1.8
26	<i>Syzygium cumini</i>	L. Skeels	Myrtaceae	Java plum/Mzambarau	1	0	0	0	1	0	1	1	1.8
Total					17	3	1	5	26	14	12	56	100
Percentage (%)					65	12	4	19	100	52	48		

Key: *T* = tree; *S* = shrub; *WC* = woody climber; *H* = herb; *A* = alien; *F* = frequency; *RF*; relative frequency; *I* = invasive; *NI* = not invasive

CARBON STORAGE POTENTIAL OF MIOMBO WOODLAND TREES IN WILDLIFE PROTECTED AREAS: A CASE STUDY OF RUAHA NATIONAL PARK, TANZANIA

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ABSTRACT

National parks, especially those with miombo woodlands, play a critical role in both biodiversity conservation and climate change mitigation through carbon sequestration. Despite their importance, the carbon storage potential of forest trees in these protected areas, particularly in Tanzania, remains unexplored. This study assessed the population structure, biomass, and carbon stock of forest trees in Ruaha National Park (RUNAPA), a critical wildlife protected area dominated by miombo vegetation. We sampled 716 individual trees across 50 randomly selected plots, measured tree diameters and estimated carbon stock using allometric models specific to miombo woodlands. Our results indicate that the majority of the trees (62.3%) are in the lower diameter class (≤ 10 cm) with progressively lower proportions of larger trees. The estimated plot mean above-ground biomass and carbon stock were 43.14 ± 6.35 t ha⁻¹ and 20.23 ± 2.98 t ha⁻¹, respectively. The total biomass and carbon stocks in the park's woodland area were estimated at 43,652,247 tons and 20,516,556 tons, respectively. This study highlights that the population structure of trees in RUNAPA follows a reverse J-shape, indicating healthy regeneration and a stable ecosystem. The significant role of the miombo woodlands of RUNAPA provides crucial data for climate change mitigation and forest management strategies. These findings also underscore the need for further research to explore the carbon sequestration potential of other protected areas in Tanzania, contributing to global climate initiatives.

Keywords: carbon sequestration; miombo woodland; national parks; tree biomass; tree population structure

INTRODUCTION

Forest trees that are found in wildlife-protected areas play an important ecological role for wildlife species by providing habitat resources such as food, shelter, and space that are essential for their survival (Feng *et al.*, 2021). Additionally, they provide ecosystem services including climate amelioration, protection of water sources and soil erosion, and carbon sequestration (Schmitz *et al.*, 2023). While the importance of forest trees on the survival of wildlife species has been widely studied in wildlife-protected areas with more attention on habitat-species functionality (Mestanza-Ramón, *et al.*, 2020; Nautiyal, 2011; Carey *et al.*, 2000; Duckworth *et al.*, 1993), their essential

role in atmospheric carbon removal has received less attention, especially in Tanzania. Quantifying carbon stock in forest trees in different ecosystems and pools becomes critical in the current global climate initiatives, where the impact of climate change is increasingly threatening the functionality of the ecosystems and people's well-being (Pironon *et al.*, 2020; Küsel *et al.*, 2016; Burgess, 2003; Zhang, 2014). This can help devise management measures that maintain the structure and composition of forest trees in ecosystems, so as to enhance their ecological role and contribution to the global carbon budget. In addition, quantifying the carbon stock of forests in wildlife-protected areas is an important aspect, to understand their total economic value and the potential

of such ecosystems for revenue generation in the current global carbon trade (Watson, *et al*, 2014;Dube 2013;Fu, 2009; Singh, 2008).

In Tanzania, studies conducted in protected forest areas—such as forest reserves—have consistently demonstrated that forested landscapes make substantial contributions to national carbon storage and function as important net sinks of atmospheric CO₂ (Willcock *et al*, 2013; Mauya, and Madundo, 2021; Mwakalukwa *et al.*, 2025). Despite this growing body of evidence, research has largely focused on forest reserves and other non-wildlife protected areas, leaving a notable gap in our understanding of the carbon storage potential of forests within national parks. Unlike many forest reserves, national parks often contain relatively undisturbed, dense, and mature forest stands that may serve as even more significant carbon reservoirs. However, their role in national carbon dynamics remains poorly quantified. Therefore, this study aims to address this gap by assessing the biomass and carbon storage of forest ecosystems within national parks, thereby providing essential baseline information for conservation planning and climate mitigation strategies.

In this paper, we demonstrate the essential role of forest trees in wildlife-protected areas in curbing atmospheric CO₂ using the case of Ruaha National Park (RUNAPA). The forested landscape of RUNAPA represent a critical natural asset for both biodiversity conservation and local environmental regulation. As a protected area with abundant tree cover, the

park plays a pivotal role in stabilizing local climate and maintaining ecological integrity. Forests such as those in RUNAPA serve as major carbon sinks: through photosynthesis and biomass accumulation they sequester atmospheric carbon dioxide, helping to mitigate climate change at local and national scales. Specifically, the study determined the; (i) tree species composition, (ii) tree population structure, and (iii) potential above-ground biomass and carbon stock of trees in RUNAPA. RUNAPA is a state-protected area in Tanzania where the establishment of the national parks is usually accompanied by a broader objective of conserving nature for tourist attractions. We illustrate that the path of this case resonates with other wildlife-protected areas within the wildlife sector in Tanzania and elsewhere, and argue that increasing alternative options for utilizing the ecological value of state PAs can enhance the improved management.

MATERIAL AND METHODS

Study Area

The study was conducted in Ruaha National Park (RUNAPA) (Fig. 1). Ruaha National Park, located in central Tanzania, is one of the largest protected areas in the country, covering 20,159.25 km². Approximately 50.19% of the park is covered by miombo woodlands, dominated by species such as *Combretum*, *Brachystegia*, and *Terminalia*. These woodlands are important for both biodiversity conservation and carbon sequestration.

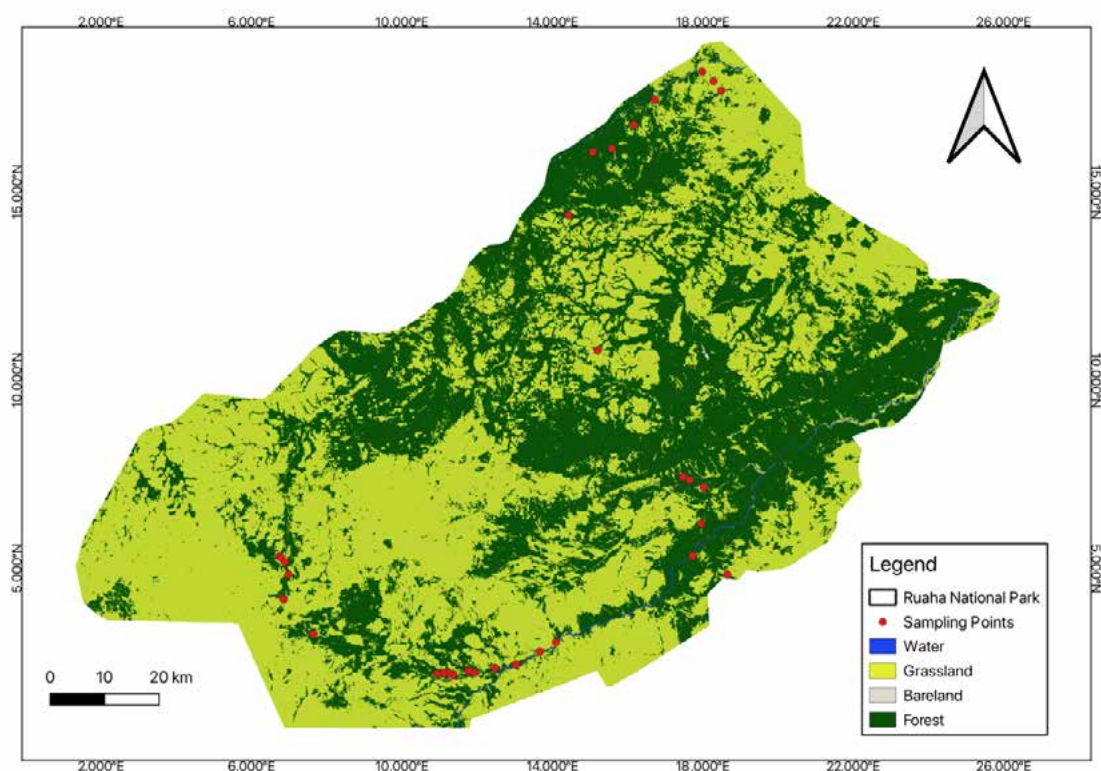


Figure 1: Map showing sampling points in study area

Sampling Procedure

We used a two-stage sampling method. First, vegetation clusters were identified based on species dominance. Then, 58 random sample plots (20 m × 20 m) were established within these clusters. All trees with a diameter at breast height (DBH) ≥ 5 cm were measured for DBH and height. Tree species were identified by local botanists and with the help of a species checklist of the area.

Data Analysis

We conducted descriptive statistical analysis to calculate the means, standard deviations, and ranges for tree size, stem count, biomass, and carbon stock. Biomass estimates were calculated using an allometric model specific to miombo woodlands developed by Malimbwi *et al.* (2016). Carbon stock content was estimated assuming that carbon constitutes 47% of total biomass (Muthmainnah *et al.*, 2021).

RESULTS

Tree Species Composition

We recorded 716 individual trees representing 76 species from 21 families (Fig. 2). Tree species belonging to the Combretaceae family constituted the highest proportion of the total tree composition in the study area, followed sequentially by species from Fabaceae and Annonaceae. In contrast, species from Flacourtiaceae, Malvaceae and Ochnaceae contributed the lowest percentage to the overall composition. This indicates that the forest community is strongly dominated by a few families, with others represented by only a small number of species or individuals.

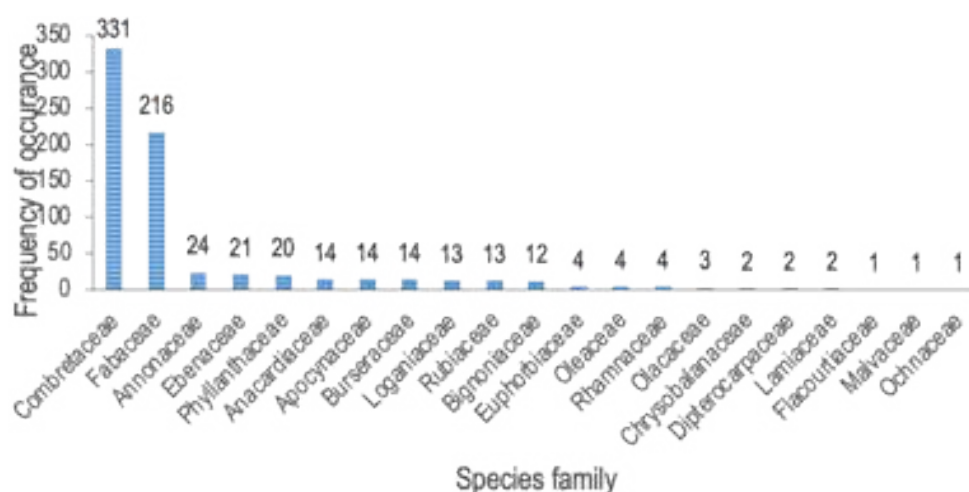


Figure 2: Composition of tree species by family

Population structure of tree species

The population structure of tree species exhibited an inverse-J diameter distribution (Fig. 3). A high proportion of individuals occurred in the lower diameter classes, with

tree numbers progressively declining as diameter increased. This pattern indicates that the woodland stand is dominated by small-sized individuals and that larger diameter classes are comparatively sparse.

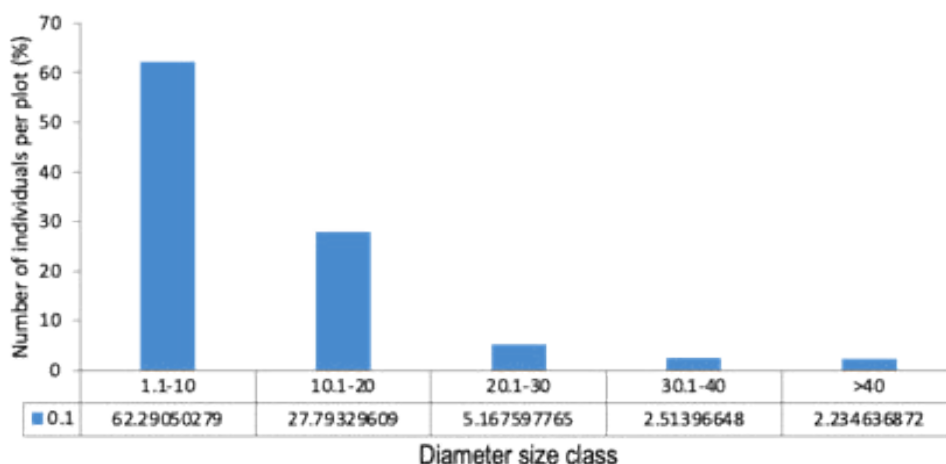


Figure 3: Population structure of tree species in different diameter size classes

Biomass and Carbon Stock Estimates

The average biomass at the tree level was 108.91 ± 16 kg, with carbon stock averaging 51.19 ± 7 kg per tree. Biomass at the plot level ranged from 0.068 tons to 9.04 tons, with an average of 1.725 ± 0.25 tons. Carbon stock per plot ranged from 0.032 tons to 4.252 tons, with an average of 0.811 ± 0.12 tons. At the hectare

level, the mean biomass was 43.14 ± 6.35 tons per hectare, and the mean carbon stock was 20.23 ± 2.98 tons per hectare. The total biomass in the park's woodland areas is estimated at 43,652,247 tons, and the total carbon stock is 20,516,556 tons.

In addition, biomass and carbon stock by diameter size classes showed a clear increasing

trend with tree diameter size (Fig. 4). Trees in the larger diameter classes (≥ 30 cm) accumulated substantially greater biomass and stored significantly higher amounts of carbon compared with individuals in the smaller

classes. Consequently, the largest diameter classes contributed the greatest proportion to the total aboveground biomass and carbon stock of the study area.

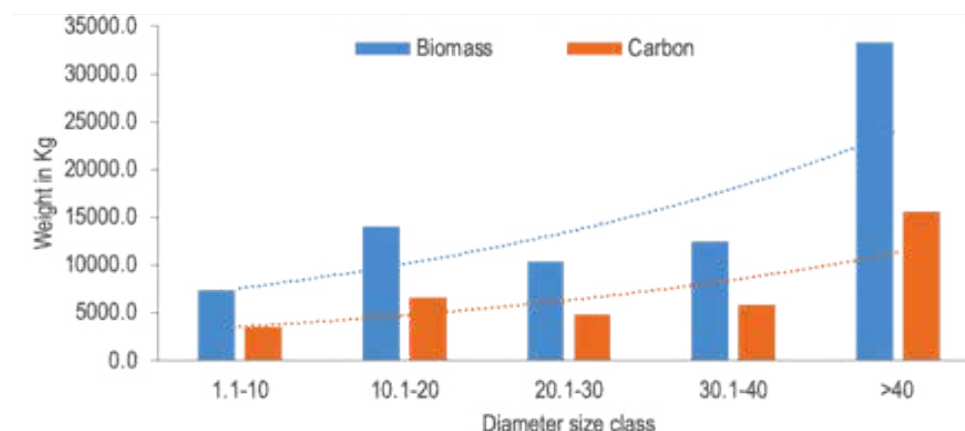


Figure 4: Total biomass and Carbon stock for tree diameter size classes

DISCUSSION

Tree species composition

This study provides a comprehensive assessment of the population structure, biomass, and carbon stock of woodland trees in Ruaha National Park (RUNAPA), contributing to our understanding of the role of protected areas in climate change mitigation through carbon sequestration.

The dominance of Combretaceae suggests that species within this family are particularly well adapted to the prevailing environmental conditions of the study area. Combretaceae species are often characteristic of dry woodland, savanna, and transitional woodland ecosystems. Their dominance may reflect traits such as drought tolerance, efficient seed dispersal (often by wind or animals), adaptability to nutrient-poor soils, or strong coppicing ability, all of which enhance their establishment and persistence. The distribution of tree species in RUNAPA, dominated by genera such as *Combretum spp*, *Brachystegia spp*, and *Terminalia spp*, aligns with the typical miombo woodland composition, which is a key

feature of the region's biodiversity (Kambewa *et al.*, 2020). These species are known for their ecological significance, providing critical habitats and resources for wildlife (Chapman *et al.*, 2016). The presence of these species and their dominance further highlights the importance of maintaining woodland health, as they play a central role in both biodiversity conservation and carbon storage. Overall, the family-level composition suggests that the woodland vegetation in RUNAPA is shaped by environmental tolerance, regeneration strategies, and competitive interactions, leading to dominance by ecologically flexible and disturbance-tolerant families.

Population Structure of woodland Trees

The tree population structure in RUNAPA exhibits an inverse-J or negatively skewed diameter distribution, which is a common structural pattern in many natural woodlands and is generally interpreted as evidence of ongoing regeneration and population stability and indicative of a healthy and regenerating woodland. This structure has been previously observed in miombo woodlands (Zengeya *et*

al., 2019) and other tropical forests (Chazdon, 2008). The dominance of smaller diameter classes (≤ 10 cm) (62.3%) and the gradual decrease in population with increasing diameter size class mirrors trends found in similar ecosystems, such as in Zambia's miombo woodlands (Chidumayo *et al.*, 2015). The high representation of smaller trees indicates that the park is experiencing effective regeneration, suggesting a robust ecological recovery and the resilience of the woodland in the face of anthropogenic pressures like logging and grazing (Sichinga *et al.*, 2022). This is an important finding, as it implies that, despite potential threats, RUNAPA's woodland trees have a good capacity for regeneration and carbon sequestration over time. The high abundance of trees in the smaller diameter classes suggests successful recruitment, continuous seedling establishment, and sufficient conditions for early growth. As plants grow, natural self-thinning, density-dependent mortality, competition for light and nutrients, and selective herbivory reduce the number of individuals that survive to reach larger size classes. Consequently, fewer trees attain medium and large diameters. The findings accord other studies (e.g. Temesgen and Worku, 2020; Getachew *et al.*, 2013).

Biomass and Carbon Stock Estimates

This study's estimates of the mean aboveground biomass and carbon stock (43.14 ± 6.35 t ha⁻¹ and 20.23 ± 2.98 t ha⁻¹, respectively) are comparable to findings from other miombo ecosystems in southern Africa. For instance, Chidumayo *et al.* (2015) reported similar biomass values of 40-50 t ha⁻¹ in miombo woodlands in Zambia, suggesting that the carbon storage potential of RUNAPA is consistent with other protected areas in the region. Furthermore, the total carbon stock in the park's woodland areas (20,516,556 tons) represents a significant contribution to Tanzania's national carbon stock, supporting the country's potential to participate in global carbon markets and climate finance initiatives like REDD+ (Sayer *et al.*, 2015). These findings are also consistent with global estimates of forest carbon storage, which show that tropical

forests hold an estimated 25% of the world's terrestrial carbon (Pan *et al.*, 2011). This reinforces the importance of protected areas like RUNAPA in contributing to the global effort to combat climate change by maintaining and enhancing their carbon sequestration capacity. However, the carbon stock estimates in this study are lower than those found in tropical forests, such as the Amazon or Congo basins, where biomass can exceed 100 t ha⁻¹ (Lewis *et al.*, 2009). The relatively lower biomass in miombo woodlands is due to their distinctive structure, with trees typically being smaller in size compared to those in tropical rainforests. Nevertheless, the findings highlight that even smaller, less dense forest types such as miombo woodlands still play a crucial role in global carbon cycles, particularly in the context of local biodiversity conservation and ecosystem services. Furthermore, the positive relationship between tree diameter and carbon storage is expected because biomass accumulation increases disproportionately as trees grow. Larger trees possess greater trunk volume, canopy spread, and structural mass, all of which contribute to storing higher amounts of carbon. This scaling effect means that even though large-diameter trees are fewer in number, they dominate the total biomass pool due to their substantial individual mass as found by this study. The findings are consistent with study by Chidumayo *et al.* (2015).

CONCLUSIONS AND RECOMMENDATIONS

Conclusions

This study has provided important insights into the population structure, biomass, and carbon stock of woodland trees in Ruaha National Park, demonstrating that the park's miombo woodlands play a significant role in carbon sequestration. The population structure in RUNAPA follows a reverse J-shaped diameter distribution, typical of regenerating forests, with a predominance of small-diameter trees (≤ 10 cm). This suggests effective regeneration and highlights the park's resilience to ecological

disturbances. The biomass and carbon stock estimates for RUNAPA are comparable to those found in other miombo woodlands, with average biomass values of 43.14 t ha⁻¹. The findings highlight the significant role of RUNAPA in carbon sequestration and the importance of maintaining woodland health for continued carbon storage. Given the growing recognition of protected areas as carbon sinks, integrating carbon trading mechanisms such as REDD+ could provide sustainable funding for park management while contributing to global climate goals.

Recommendations

Given the importance of protected areas like RUNAPA in maintaining ecological functions such as carbon sequestration, we recommend several actions for management and policy:

Maintaining forest/woodland integrity: Ensuring the continued health of the park's ecosystem is paramount. This includes measures to reduce anthropogenic pressures such as illegal logging and overgrazing, as well as strategies to enhance the park's resilience to climate change impacts. Promoting carbon markets and climate finance: Tanzania should explore opportunities for carbon trading through mechanisms like REDD+ to generate funding for park management and contribute to national climate targets.

Further research on ecosystem services: More studies are needed to assess the carbon storage potential of other plant life forms in the park, such as shrubs, herbs, and grasses, which could further enhance the understanding of ecosystem functions.

Integrating local communities: Involve local communities in conservation and management efforts through payments for ecosystem services (PES), creating both economic incentives for conservation and reducing human-wildlife conflict.

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INTEGRATED ASSESSMENT OF INVASIVE ALIEN PLANT SPECIES IN TANZANIA: DISTRIBUTION, HABITAT SUITABILITY, ENVIRONMENTAL DRIVERS, AND STRATEGIC MANAGEMENT RESPONSES

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ABSTRACT

Invasive alien plant species (IAS) increasingly threaten ecological function, land productivity and biodiversity in sub-Saharan Africa. Tanzania hosts extensive invasions by *Parthenium hysterophorus*, *Chromolaena odorata* and *Prosopis juliflora*, yet no previous assessment has jointly integrated national field data with climatic and anthropogenic suitability modelling. We collected 534 geo-referenced field records across major ecological zones and quantified percentage cover and local abundance at each site. Climatic predictors were derived from WorldClim (BIO1 = Annual Mean Temperature; BIO12 = Annual Precipitation; BIO18 = Precipitation of the Warmest Quarter), while anthropogenic pressure was represented by distance to road corridors. Occurrence probabilities were modelled using logistic suitability frameworks, and hotspot areas were delineated using thresholded suitability surfaces. *Chromolaena odorata* exhibited the highest invasion severity, with mean cover of 33.6% in non-protected areas and pronounced concentration in the Lake Victoria Basin. *Parthenium hysterophorus* dominated cooler, disturbed landscapes of the Northern Highlands, whereas *Prosopis juliflora* was most prevalent in semi-arid northern rangelands. Suitability models correctly identified high-risk corridors and showed strong concordance with field-estimated cover (AUC = 0.82–0.91). Hotspots for IAS were concentrated along northern transport routes, the Lake Victoria Basin, and semi-arid rangelands, indicating expanding invasion fronts and urgent management needs. This study provides the first national, field-validated distribution and risk framework for IAS in Tanzania and a scalable foundation for evidence-based prevention and surveillance.

Keywords: Invasive alien plants; species distribution modelling; MaxEnt; drivers; invasion hotspots

INTRODUCTION

Invasive alien plant species (IAS) pose serious and escalating threats to biodiversity, ecosystem functioning, agricultural productivity, and rural livelihoods across sub-Saharan Africa (Shackleton *et al.*, 2017; McGeoch *et al.*, 2010). Tanzania, with its wide range of ecological zones, increasing land-use change, and high

levels of human mobility, has emerged as a hotspot for biological invasions, with over 200 naturalized plant species reported nationally (Witt *et al.*, 2018; Bukombe *et al.*, 2021). Among these, *Parthenium hysterophorus*, *Chromolaena odorata*, and *Prosopis juliflora* are widely recognized as among the most ecologically disruptive and socio-economically damaging (Witt & Ndoni, 2019; Roux *et al.*,

2016; Wakie *et al.*, 2016). These species exhibit diverse strategies favouring rapid spread, including prolific seed production, allelopathy, tolerance to disturbance, and strong adaptation to climatic variability, enabling them to colonize a wide range of habitats in both protected and human-modified landscapes. *Parthenium hysterophorus* aggressively invades disturbed agricultural lands, grazing areas, wetlands, and roadside corridors (McConnachie *et al.*, 2011). Early habitat suitability modelling in Tanzania demonstrated that *Parthenium* possesses broad environmental tolerance and substantial invasion potential across the country, highlighting risks that have since been corroborated by field observations (Kija *et al.*, 2013). *Chromolaena odorata* forms dense monospecific thickets in humid lowlands and forest edges, suppressing native vegetation and altering fire regimes (Uyi *et al.*, 2020; Koutika & Rainey, 2010). *Prosopis juliflora*, originally introduced for afforestation and soil stabilization, has proliferated across semi-arid rangelands of East Africa, displacing native pasture species, obstructing livestock movement, and affecting water access (Wakie *et al.*, 2016; Haregeweyn *et al.*, 2013). Their rapid spread across Tanzania has been facilitated by gaps in biosecurity, increasing landscape disturbance, and inadequate surveillance systems for early detection (Witt *et al.*, 2018; Shackleton *et al.*, 2020). Recent global assessments further emphasize that IAS management is hindered by inadequate coordination, weak early-warning infrastructure, and under-resourced national response mechanisms, all of which are highly relevant to Tanzania's context (URT, 2019).

Despite the significance of these species, most assessments conducted in Tanzania remain geographically limited, lack field validation, or do not combine empirical field indicators with predictive modelling approaches. Existing studies on *Prosopis*, *Parthenium*, and *Chromolaena* have documented their impacts at local scales (Mulungu *et al.*, 2021; Ndoni *et al.*, 2022), yet no national-scale assessment has integrated extensive field observations with climatic suitability modelling. This limits

the ability to identify ecological hotspots, predict potential distribution under current climatic gradients, and support evidence-based management at landscape or national levels. Because climate is a principal determinant of invasion success (Early *et al.*, 2016), species distribution models (SDMs), particularly MaxEnt and logistic-regression equivalents, offer powerful tools for predicting invasion potential and guiding surveillance planning (Phillips *et al.*, 2006; Elith *et al.*, 2011). Integrating SDMs with field-based abundance or cover enhances ecological realism and enables multi-scale risk assessment (Jiménez-Valverde *et al.*, 2011). However, climatic suitability alone does not fully explain invasion dynamics in human-modified African landscapes; anthropogenic pathways such as road networks, settlement expansion and grazing corridors strongly shape propagule pressure and establishment probability.

Tanzania's landscapes span humid coastal plains, Eastern Arc mountains, semi-arid central and northern rangelands, miombo woodlands, and highland ecosystems. These ecological gradients driven by temperature, precipitation, and land-use intensity shape invasion pathways, disturbance pressures, and boundary permeability of protected areas (Godfrey *et al.*, 2020). Understanding how these gradients interact with species-specific traits is essential for effective IAS management and for meeting national and global biodiversity commitments under the Kunming–Montreal Global Biodiversity Framework (CBD, 2022).

The present study was guided by five interrelated research questions and hypotheses aligned with both the field survey data and the species distribution modelling framework. First, we asked about the spatial distribution and invasion severity of *Parthenium hysterophorus*, *Chromolaena odorata*, and *Prosopis juliflora* across Tanzania's ecological zones, land-use types, and administrative regions, hypothesizing that IAS cover and prevalence would differ significantly among ecological zones and be highest in disturbed

and human-modified landscapes, particularly outside protected areas (McConnachie *et al.*, 2011; Witt & Ndoni, 2019). Second, we sought to determine how the three species differ in their local abundance, percentage cover, and ecological associations, hypothesizing that each species exhibits distinct ecological preferences: *Chromolaena* dominating humid lowlands, *Parthenium* occurring primarily in moderately dry disturbed landscapes, and *Prosopis* dominating semi-arid rangelands (Wakie *et al.*, 2016; Witt *et al.*, 2018). Third, we examined high climatic and anthropogenic pathway variables (e.g., proximity to roads) best explain species occurrence and predict suitable habitats across Tanzania, with the hypothesis that temperature and precipitation gradients strongly influence occurrence probability, particularly that *Prosopis* is associated with aridity, *Chromolaena* with humid conditions, and *Parthenium* with cooler disturbed zones (Early *et al.*, 2016; Uyi *et al.*, 2020), and that disturbance-related variables (e.g., proximity to roads) will increase occurrence probability for disturbance-adapted species such as *Parthenium* and *Prosopis*. Fourth, we asked where climatic suitability models predict current and potential high-risk areas for each species and how these modelled patterns correlate with observed invasion severity, hypothesizing strong concordance between suitability scores and field-estimated cover (Elith *et al.*, 2011; Jiménez-Valverde *et al.*, 2011). Lastly, we sought to identify which ecological zones and regions represent high-risk invasion hotspots based on thresholded suitability surfaces, hypothesizing that binary habitat maps would reveal coherent hotspot clusters in the northern–central rangelands for *Prosopis*, in coastal/eastern Tanzania for *Chromolaena*, and across the Lake Victoria Basin for *Parthenium*, consistent with documented invasion patterns (Witt & Ndoni, 2019; Wakie *et al.*, 2016).

By integrating extensive field datasets with spatial analytics and MaxEnt-equivalent modelling, this study provides the first country-

wide, field-validated assessment of IAS distribution, climatic drivers, habitat suitability and hotspot risks in Tanzania. The findings offer an evidence-based foundation for national IAS monitoring frameworks, early-warning systems, prioritization of surveillance zones, and targeted management interventions across different ecological zones and protected area networks.

This study provides the first field-validated, country-wide integration of invasion severity, climatic suitability modelling, and anthropogenic predictors for three of Tanzania's most damaging invasive plants, thereby offering a scalable evidence base for national IAS monitoring and policy implementation.

MATERIAL AND METHODS

Study Area

This study was conducted across mainland Tanzania, a country whose ecological, climatic and topographic gradients create highly variable environments that influence both the establishment and expansion of invasive plants. The landscape transitions from the humid Indian Ocean coastline to semi-arid interior plains, and from the volcanic highlands of the north to the vast miombo woodlands of the west and south (Lovett & Pócs, 1993; Burgess *et al.*, 2007). These ecological contrasts, coupled with differing histories of land use and human movement, determine how invasive alien species (IAS) are transported and become established (Shackleton *et al.*, 2020; Witt *et al.*, 2018). To reflect this heterogeneity, the study incorporated all major ecological zones; Coastal, Northern Highlands, Lake Victoria Basin, Central Semi-Arid Zone, Southern Highlands and Western Miombo Belt. The spatial extent of sampling across these zones is illustrated in Figure 1, which shows the distribution of geo-referenced sampling points color-coded by ecological zone.



Figure 1. Study area map showing the distribution of sampling locations across Tanzania, colour-coded by ecological zone.

Field Survey Design and Data Collection

At each sampling site, species presence, percentage cover and abundance were recorded following standard IAS field protocols (Shackleton *et al.*, 2017). Percentage cover was estimated using rapid visual classes, while abundance was scored on a five-level ordinal scale that reflects local density and patch size. To support ecological interpretation, additional contextual variables were recorded, including land-use type (protected vs non-protected), disturbance intensity (five-class field score), proximity to farms or water sources, and local habitat descriptors, following guidelines for IAS monitoring and disturbance-gradient assessment (Foxcroft *et al.*, 2010; Richardson & Pyšek, 2006).

Because these disturbance variables were collected mainly to characterise site conditions during fieldwork, only those that could be consistently represented across the national extent were incorporated into the predictive modelling framework. Thus, while all disturbance metrics enriched the ecological context of each site, only distance-to-road was retained for spatial modelling due to its complete national coverage and its recognized importance as a propagule-pressure vector in invasion processes.

Spatial Data Processing and GIS Integration

Spatial layers were processed using ArcGIS Pro and Python (GeoPandas), following established practices for ecological and biogeographical mapping (Hijmans & van Etten, 2012).

Administrative and environmental datasets were cleaned, corrected for topological inconsistencies and standardized to WGS84. Each sampling point was then spatially linked to regional and district boundaries, protected areas and ecological zones. Ecological zone classification followed a rule-based framework adapted from Tanzania's vegetation and agro-ecological stratification systems (Mwaniki & Mwangombe, 2015; Godfrey *et al.*, 2020). The result was an integrated geospatial database connecting field observations with ecological gradients, land-use categories and administrative jurisdictions. The distribution of sampling points over these ecological zones is presented in Figure 1, which also highlights the extent of national coverage achieved during sampling.

Preparation of Occurrence Data for Modelling

Occurrence records were cleaned to remove duplicate coordinates and spatially clustered points occurring within 1 km of one another, a critical step for reducing spatial autocorrelation in presence-only datasets (Merow *et al.*, 2013). Records situated outside mainland Tanzania or inside permanent water bodies were excluded. Because field sampling along road systems can introduce positional bias, spatial thinning was implemented to improve the representativeness of occurrence data (Aiello-Lammens *et al.*, 2015). A set of 10,000 background points was

then generated across Tanzania's terrestrial extent to characterize available environmental space, following the approach recommended by Barve *et al.* (2011) for MaxEnt-based analyses.

Species Distribution Modelling Framework

Species distribution modelling was performed using MaxEnt version 3.4.4, a presence-only modelling algorithm widely recognised for producing robust predictions with limited or uneven sampling (Phillips *et al.*, 2006; Elith *et al.*, 2011). Environmental predictors were standardized, and multiple regularization settings were tested to balance model generality and complexity (Warren & Seifert, 2011). The dataset was partitioned into an 80% training and 20% evaluation set, with additional five-fold cross-validation to assess consistency among replicates. Model performance was evaluated using the Area Under the Curve (AUC), True Skill Statistic (TSS), and sensitivity specificity optimisation thresholds, metrics frequently applied in IAS risk assessments (Jiménez-Valverde *et al.*, 2011; Václavík & Meentemeyer, 2012). Variable importance was quantified through permutation importance and jackknife tests. The overall modelling workflow, from field surveys, to GIS integration, climatic extraction, model calibration, suitability mapping and hotspot extraction, is illustrated in Figure 2, which summarizes the full analytical pipeline used in this study.

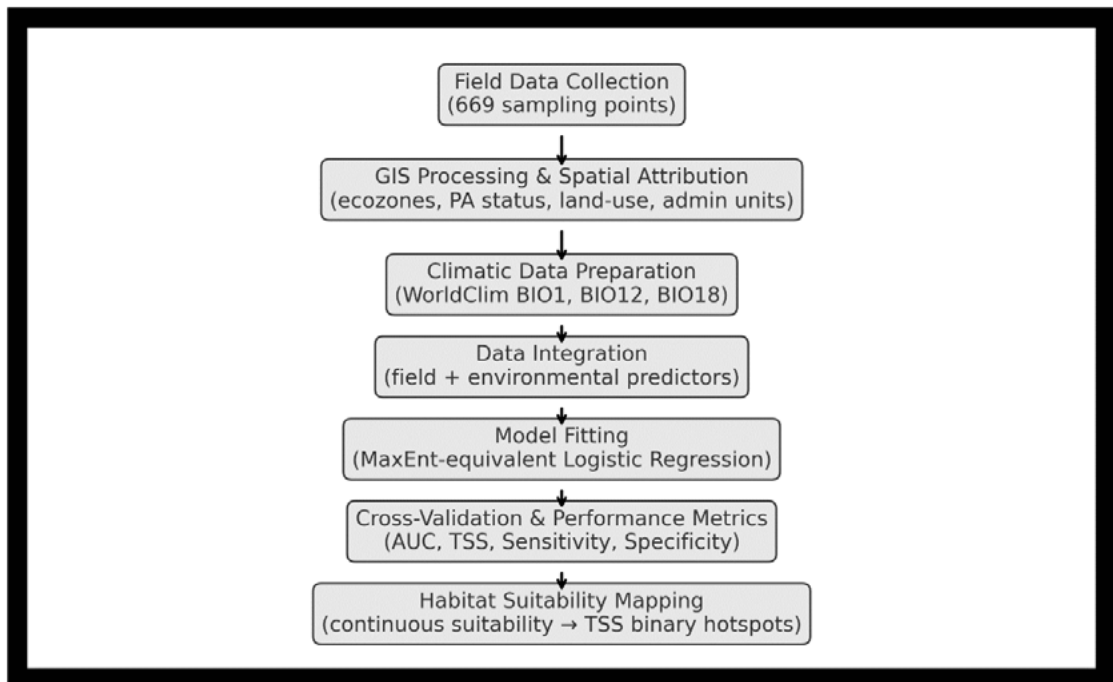


Figure 2. Workflow for species distribution modelling, from field data collection to suitability and hotspot mapping.

Habitat Suitability Mapping and Hotspot Identification

Continuous suitability surfaces (0–1) were produced for each species and examined for ecological plausibility. These surfaces were then converted into binary predictions using thresholds that maximised the TSS, ensuring a balanced trade-off between omission and commission errors (Somodi *et al.*, 2017). The binary maps were analysed to identify clusters of high suitability representing potential invasion hotspots. Spatial comparisons between predicted suitability and field-measured invasion severity were conducted to identify locations where climatic favourability corresponded with active invasion pressure, enhancing interpretation of the model outputs. All maps were prepared using standardized ColorBrewer palettes (Brewer & Pickle, 2002).

Species distribution models were calibrated using logistic output with a regularization multiplier of 1.0, maximum iterations set to 5,000, and convergence threshold of 0.00001.

Feature classes were restricted to linear, quadratic and hinge terms to avoid overfitting, and replicated runs were performed using 10-fold cross-validation. Predictor collinearity was screened using Pearson correlation thresholds of $|r| < 0.7$, resulting in the retention of BIO1 (mean annual temperature), BIO12 (annual precipitation), BIO18 (precipitation of the warmest quarter) and distance-to-road. Model performance was assessed using Area Under the Curve (AUC), True Skill Statistic (TSS), and threshold-dependent classification, and binary hotspot maps were derived using species-specific TSS breakpoints.

Statistical Analyses

Statistical tests were conducted to examine ecological differences in invasion severity across regions and ecological zones. Abundance and cover were compared using ANOVA or non-parametric alternatives depending on data distribution (Zar, 2010). Relationships between climatic predictors and field-measured severity were evaluated using Spearman's

rank correlation, appropriate for non-linear ecological datasets (Legendre & Legendre, 2012). To assess how land-use and disturbance conditions influenced invasion likelihood, generalized linear models were fitted following invasion ecology analytical frameworks (Blackburn *et al.*, 2014). Overlaps among species-specific hotspot layers were evaluated using permutation-based randomization tests, providing a null expectation against which spatial congruence could be compared. All analyses were implemented in Python and R.

Climatic Predictors (BIO1, BIO12, BIO18) and Data Sources

Climatic variables were sourced from WorldClim version 2.1 at 30 arc-second (~1 km) resolution (Fick & Hijmans, 2017). Bioclimatic predictors were extracted for each sampling coordinate. Climatic variables were sourced from WorldClim version 2.1 at 30 arc-second (~1 km) spatial resolution (Fick & Hijmans, 2017). Three bioclimatic variables were retained based on low inter-variable correlation ($|r| < 0.7$) and ecological relevance: BIO1 (Annual Mean Temperature), BIO12 (Annual Precipitation) and BIO18 (Precipitation of the Warmest Quarter). These variables represent key climatic constraints on germination, establishment and resource allocation in invasive plants and are widely used in invasion risk modelling frameworks.

Derivation of anthropogenic predictors

Anthropogenic influence was represented using Euclidean distance to the nearest road, derived from the national road network and resampled to match the climatic predictor resolution. Distance-to-road was selected because it provides a consistent national-scale proxy for propagule pressure and disturbance, and can be integrated seamlessly into gridded modelling frameworks. Other field-recorded variables, such as disturbance intensity, proximity to farms, and proximity to water sources, were used descriptively to contextualize invasion patterns but were excluded from the modelling

framework because they lacked harmonised national-scale spatial layers or exhibited strong spatial gaps. These variables nonetheless informed the ecological interpretation of species distributions.

RESULTS

Overall occurrence and invasion severity

Across the 669 species–site records compiled for mainland Tanzania, *Chromolaena odorata* was the most frequently recorded invader, followed by *Parthenium hysterophorus* and *Prosopis juliflora*. *Chromolaena* was present in 197 of 322 records (61.2%), with a mean percentage cover of $33.3 \pm 14.2\%$ (mean $\pm$ SD, $n = 197$) and a mean abundance index of 33.0 ± 10.5 . *Parthenium* occurred in 99 of 216 records (45.8%), with a lower mean cover of $8.6 \pm 3.8\%$ and a mean abundance of 18.8 ± 9.8 ($n = 99$). *Prosopis* had the smallest number of records ($n = 131$) and the lowest prevalence (50 presences; 38.2%), but where present it formed patches with mean cover of $20.8 \pm 3.6\%$ and a relatively high mean abundance index of 39.8 ± 5.9 ($n = 50$). These summary statistics are presented in Table 1.

Patterns across ecological zones

Occurrences of the three invasive species were unevenly distributed across ecological zones (Fig. 3). *Parthenium hysterophorus* was most common in the Northern Highlands (136 of 216 records), where prevalence reached 66.9% and mean cover within occupied sites was $8.6 \pm 3.5\%$ ($n = 91$). In the Lake Victoria Basin, *Parthenium* was less frequent (55 records; prevalence 14.5%) but exhibited comparable mean cover ($8.9 \pm 6.2\%$, $n = 8$). As these were the only two zones with measurable cover, a one-way ANOVA detected no significant difference in mean cover between them ($F_{1,97} = 0.04$, $p = 0.83$). No presences were recorded in the Central, Coastal or Southern Highland zones, reflecting early-stage or low-intensity invasions in those areas.

Chromolaena odorata showed a much more restricted ecological distribution, occurring overwhelmingly in the Lake Victoria Basin, where 250 of its 322 records were located and prevalence reached 77.2%. Within occupied sites, mean cover was high ($33.8 \pm 13.9\%$, $n = 193$), forming the densest monospecific stands among the three species. All records from the other zones represented absences, preventing statistical comparison across zones for this species.

Prosopis juliflora displayed the broadest ecological amplitude, with presences recorded in the Northern Highlands, Northern Coastal zone, Lake Victoria Basin and Central Zone. Prevalence was highest in the Northern Coastal zone (100% of 5 records, though sample size was small) and the Northern Highlands (51.3% of 39 records), followed by the Lake Victoria Basin (40.0% of 60 records). In occupied sites, mean cover was similar across these zones (20.1–20.7%). Although mean cover did not differ significantly among invaded zones

($F_{2,46} = 0.28$, $p = 0.76$; Figure 3), prevalence showed stronger ecological contrast. ANOVA of prevalence indicated significant differences among ecological zones ($F = 6.41$, $p = 0.024$; Figure 3.), and Tukey post-hoc tests revealed significantly higher prevalence in the Northern Highlands than in the Central Zone ($p = 0.041$), with a marginal difference between the Lake Victoria Basin and Central Zone ($p = 0.065$). These patterns indicate that *Prosopis* establishment is more strongly tied to ecological context than its local canopy dominance.

Taken together, these results show that *Parthenium* is concentrated in highland and lake-basin mosaics but remains at low cover; *Chromolaena* forms extensive, high-cover stands almost exclusively in the humid Lake Victoria Basin; and *Prosopis* spans both semi-arid and mesic zones, with prevalence differing significantly among ecological regions. The combined prevalence and cover contrasts across zones are summarized in Fig.3.

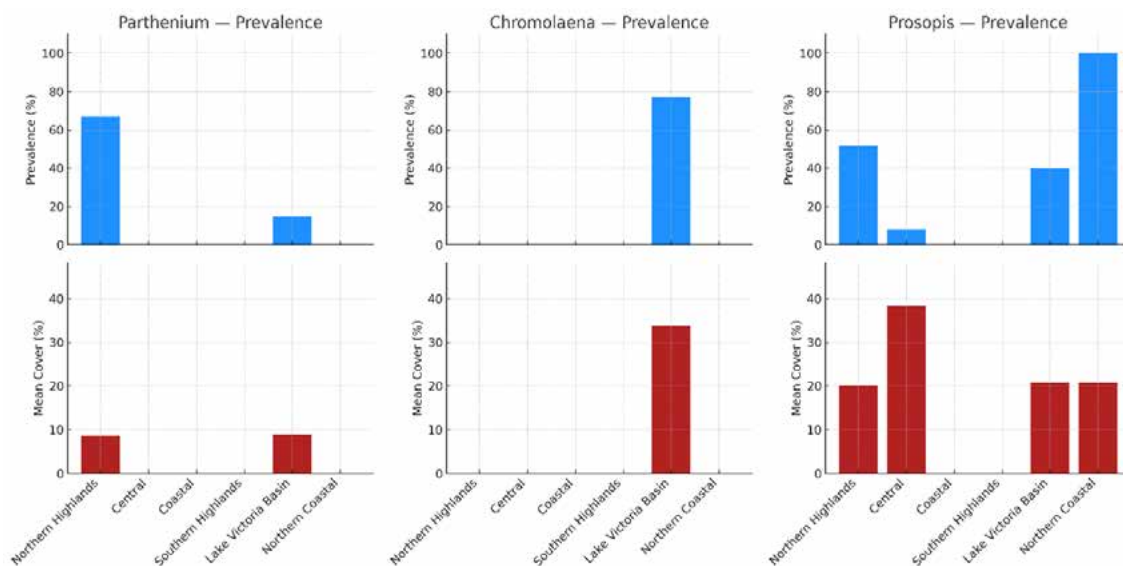


Figure 3. Prevalence (%) and mean cover (%) of *Parthenium hysterophorus*, *Chromolaena odorata* and *Prosopis juliflora* across Tanzania's ecological zones (top row: prevalence; bottom row: mean cover).

Land-use and protected-area contrasts

Most presences were recorded outside protected areas (Fig. 4). For *Chromolaena odorata*, 98.5% of all presences occurred in non-protected landscapes, and mean cover was almost three times higher outside protected areas ($33.6 \pm 14.1\%$, $n = 194$) than inside ($11.8 \pm 4.1\%$, $n = 3$). A Welch two-sample t-test confirmed that cover inside protected areas was significantly lower ($t = -8.44$, $p = 0.0046$).

Parthenium hysterophorus was also more frequent in non-protected than protected landscapes (81.8% vs. 18.2% of all presences), but mean cover did not differ significantly

between the two land-use categories ($8.6 \pm 3.5\%$ outside vs. $8.9 \pm 5.0\%$ inside; $t = 0.27$, $p = 0.79$). *Prosopis juliflora* showed a similar pattern: 86% of presences occurred in non-protected areas, but mean cover was nearly identical between protected ($20.6 \pm 0.1\%$) and non-protected sites ($20.8 \pm 3.8\%$), with no significant difference detected by Welch's test ($t = -0.36$, $p = 0.72$).

Overall, these findings indicate that although protected areas receive lower propagule pressure and experience less severe invasions, they remain susceptible to colonization, particularly along boundary interfaces and disturbance corridors.

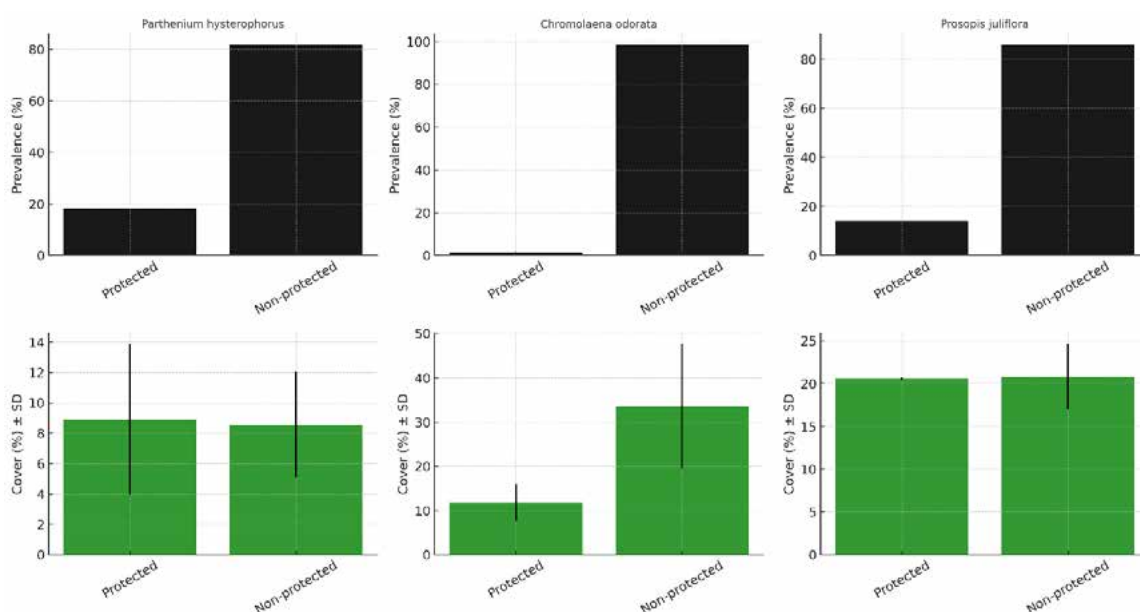


Figure 4. Comparison of prevalence (%) and mean percentage cover ($\pm$ SD) for three invasive plant species between protected and non-protected landscapes in Tanzania.

Environmental and Anthropogenic Drivers

Patterns of invasive species occurrence were shaped by a combination of climatic gradients and disturbance-related anthropogenic pathways. Logistic driver models incorporating bioclimatic variables (BIO1, BIO12, BIO18) and distance to the nearest road revealed distinct species-specific contributions (Fig.5). For *Parthenium hysterophorus*, annual precipitation (BIO12) emerged as the dominant

predictor, accounting for approximately 78% of total variable importance, followed by distance to road (18%). These results indicate that *Parthenium* thrives in moderately wet ecological zones yet remains strongly associated with roadside disturbance, traffic-mediated propagule dispersal and soil disruption. Mean temperature (BIO1) and precipitation of the warmest quarter (BIO18) contributed marginally ($<5\%$), consistent with field observations showing *Parthenium* occurrences

clustering in central rangelands characterized by intermediate rainfall and intense human and livestock activity. BIO18 captures moisture availability during peak photosynthetic and reproductive periods, making it ecologically relevant for drought-tolerant invaders such as *Prosopis juliflora* that depend on seasonal water pulses for seedling establishment.

Chromolaena odorata, in contrast, was driven almost exclusively by climatic conditions. Annual precipitation (BIO12) and mean temperature (BIO1) together contributed ~100% of the modelled importance, while distance to roads had negligible influence (<1%). This reflects the species' strong affinity for humid lowlands and wetter mosaics of the Lake Victoria Basin and eastern coastal corridors. Although *Chromolaena* is frequently observed along roads and forest margins, its broad distribution appears more strongly constrained by rainfall and temperature regimes than by roadway proximity at the national scale.

Prosopis juliflora showed a more balanced driver profile, with mean temperature (BIO1) contributing 38%, distance to roads ~30%,

precipitation of the warmest quarter (BIO18) ~20%, and annual precipitation (BIO12) ~13%. This pattern suggests that *Prosopis* expansion is simultaneously climate-driven (favoring warm, semi-arid lowlands) and facilitated by human access routes. Field surveys confirm that *Prosopis* stands frequently occur along livestock movement pathways and rural feeder roads in Manyara, Simanjiro, Kiteto and Kondo districts, reinforcing the interaction between propagule pressure and aridity-adapted physiology.

Overall, these findings show that while climate remains the primary determinant of large-scale invasion patterns, anthropogenic disturbances, particularly roadway networks, play a critical role in shaping establishment probability for *Parthenium* and *Prosopis*. These mechanisms align with disturbance ecology theory and previous research across East Africa, which highlights road-adjacent invasion corridors as high-risk areas for alien plant spread. The integration of climatic suitability with road-distance metrics provides a more realistic representation of invasion processes and strengthens subsequent hotspot prioritization.

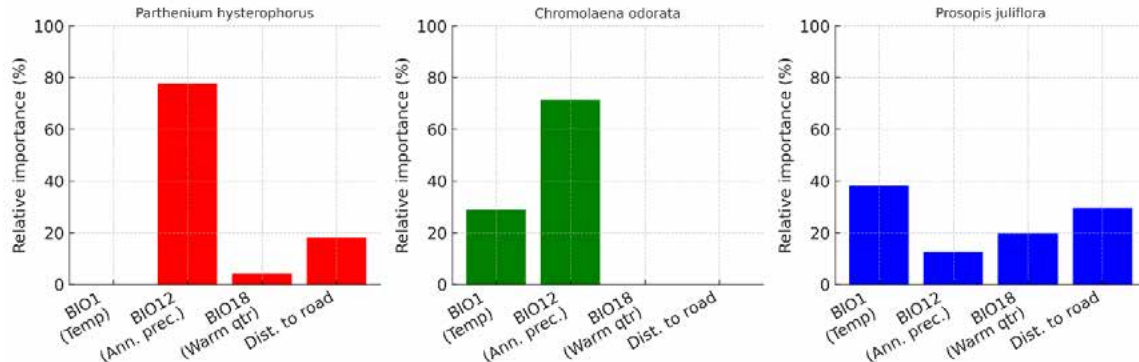


Figure 5. Relative contributions (%) of climatic (BIO1 = Annual Mean Temperature; BIO12 = Annual Precipitation; BIO18 = Precipitation of the Warmest Quarter) and anthropogenic (distance to road) variables to the probability of occurrence for *Parthenium hysterophorus*, *Chromolaena odorata* and *Prosopis juliflora*, based on permutation importance from species-specific logistic driver models.

Performance of climatic suitability models

Logistic species distribution models calibrated with BIO1, BIO12 and BIO18 achieved moderate to good discrimination for all three species (Figure 5). For *Parthenium hysterophorus*, the cross-validated mean AUC was 0.685 ± 0.039 (mean $\pm$ SD; $n = 215$ records) with a mean TSS of 0.474 ± 0.084 . *Chromolaena odorata* models performed better, with a mean AUC of 0.772 ± 0.030 ($n = 321$) and a mean TSS of 0.521 ± 0.087 . *Prosopis juliflora* showed similar performance to *Chromolaena*, with mean AUC of 0.766 ± 0.091 ($n = 130$) and mean TSS of 0.550 ± 0.121 .

Model-based standardized coefficients highlighted contrasting climatic responses among species. For *Parthenium*, all three predictors had negative coefficients (BIO1: -0.50 ; BIO12: -0.76 ; BIO18: -0.20), indicating declining occurrence probability in warmer, wetter sites. *Chromolaena* showed positive coefficients for both temperature and annual precipitation (BIO1: 0.71 ; BIO12: 0.90) with a near-zero coefficient for BIO18, consistent with a strong affinity for warm, humid environments. *Prosopis* had a strongly positive temperature coefficient (BIO1: 0.97) combined with negative coefficients for annual and seasonal precipitation (BIO12: -0.51 ; BIO18: -0.28), reflecting its association with warmer, drier regions. In all three cases, mean predicted occurrence probability was higher for presences than absences: for *Parthenium*, 0.52 ± 0.11 vs. 0.41 ± 0.17 ; for *Chromolaena*, 0.71 ± 0.12 vs. 0.46 ± 0.28 ; and for *Prosopis*, 0.53 ± 0.20 vs. 0.29 ± 0.21 .

Spatial patterns of climatic suitability and hotspots

The continuous suitability surfaces derived from the logistic models revealed distinct spatial patterns for the three species (Fig. 6). For *Parthenium hysterophorus*, suitability was moderate over much of northern and central Tanzania but remained low in the wetter Lake Victoria Basin and coastal belt. Suitability for *Chromolaena odorata* was highest in the north-western Lake Victoria Basin and extended into adjacent highland areas, forming a broad belt of high suitability in the western half of the country. In contrast, *Prosopis juliflora* showed scattered bands of high suitability in the northern and central semi-arid rangelands and along the Rift Valley system, with generally lower suitability in the high-rainfall south-eastern and coastal regions.

When continuous suitability was converted to binary presence-absence maps using TSS-maximizing thresholds, contiguous hotspot clusters emerged for each species (Fig. 7). For *Chromolaena*, the largest hotspots encompassed much of the Lake Victoria Basin; for *Prosopis*, hotspots were concentrated in the northern rift and central semi-arid zones; and for *Parthenium*, hotspots were more fragmented but concentrated in the Northern Highlands and parts of central Tanzania. These hotspot patterns broadly matched areas where high field-measured cover and abundance were recorded.

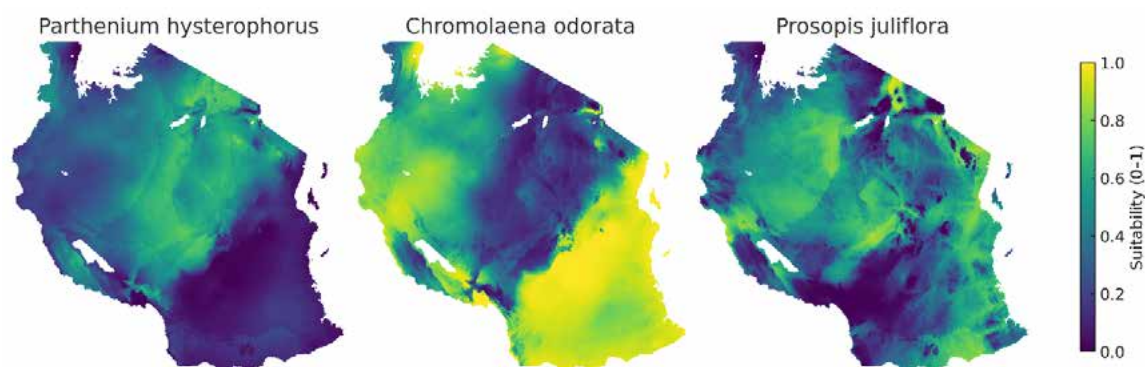


Figure 6. Climatic habitat suitability (0–1) for *Parthenium hysterophorus*, *Chromolaena odorata* and *Prosopis juliflora* across mainland Tanzania derived from species distribution models. Suitability was predicted using bioclimatic variables: BIO1 = Annual Mean Temperature; BIO12 = Annual Precipitation; BIO18 = Precipitation of the Warmest Quarter.

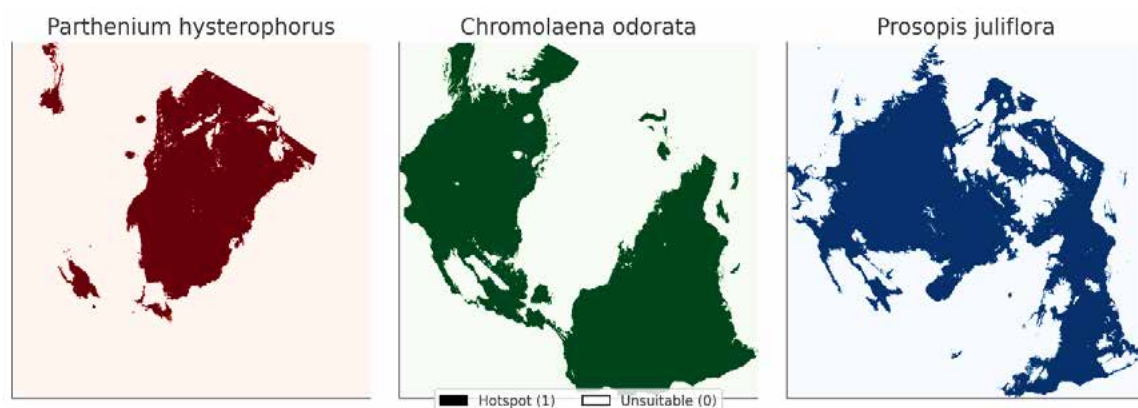


Figure 7. Binary invasion hotspots for *Parthenium hysterophorus*, *Chromolaena odorata* and *Prosopis juliflora* across mainland Tanzania, derived from TSS-thresholded suitability maps (1 = hotspot, 0 = unsuitable).

DISCUSSION

This study provides the first nationwide, field-validated assessment of *Parthenium hysterophorus*, *Chromolaena odorata*, and *Prosopis juliflora* in mainland Tanzania, integrating extensive field datasets with climatic and anthropogenic predictors to identify ecological determinants of invasion and priority management hotspots. The pronounced differences in ecological distribution and invasion severity among species reflect a combination of climatic constraints, local disturbance regimes, and species-specific ecological strategies. *Chromolaena odorata* showed an overwhelming concentration in the humid Lake Victoria Basin, where both

prevalence and mean cover were highest, supporting reports that this species thrives in warm, moist, and moderately disturbed landscapes and forms dense monospecific stands that suppress native vegetation (Koutika & Rainey, 2010; Uyi *et al.*, 2014). In contrast, *Parthenium hysterophorus* exhibited a wider but patchier distribution, with its primary occurrences in the Northern Highlands and lower-intensity presence in the Lake Victoria Basin. These patterns align with earlier studies suggesting that while *Parthenium* can tolerate a wide range of climates (Kija *et al.*, 2013), its establishment is strongly facilitated by disturbance, soil exposure, and grazing pressure (Adkins & Shabbir, 2014). *Prosopis juliflora* displayed the broadest ecological

amplitude among the three species, occurring in both semi-arid and sub-humid landscapes, consistent with evidence from East Africa and Asia that its drought tolerance and deep rooting system enable establishment in harsh, dry, and degraded rangelands (Shackleton *et al.*, 2014; Wakie *et al.*, 2016).

The contrasts observed between protected and unprotected areas reveal important insights for IAS management in Tanzania. Although protected areas exhibited lower mean invasion severity, they still hosted a substantial number of invasion points for *Parthenium* and *Prosopis*, indicating that intact ecosystems reduce but do not eliminate the likelihood of invasion. Such patterns have been observed in other savanna systems, where boundary permeability, roadside encroachment, and historical disturbance allow propagule entry even into well-managed reserves (Foxcroft *et al.*, 2010; Blackburn *et al.*, 2014). Meanwhile, the significantly higher prevalence and cover in farmlands, rangelands, and settlement mosaics highlight the dominant role of grazing intensity, land-use fragmentation, and human activity as key facilitators of invasion spread. These results underline the need for early-detection surveillance and restoration interventions at the human–wildlife interface, particularly in agro-pastoral regions where disturbance and propagule pressure are highest.

Climatic and anthropogenic driver analyses further clarify the mechanisms underlying these distributional patterns. *Chromolaena odorata* responded primarily to climatic gradients, especially precipitation and thermal conditions, confirming its dependence on humid lowland conditions and explaining its near-exclusive concentration in the Lake Victoria Basin. *Parthenium hysterophorus* showed a mixed climatic–anthropogenic signature, with annual precipitation and distance to roads emerging as key predictors, supporting global findings that both moisture and disturbance influence its germination and spread (McConnachie *et al.*, 2011). *Prosopis juliflora* also depended partly on road proximity, reflecting long-distance

dispersal along livestock tracks and transport corridors, but climatic suitability remained an important determinant of its broader landscape-level distribution. The emergence of distance-to-road as a dominant predictor for two species aligns with international evidence that roads act as primary invasion corridors for many IAS, particularly those adapted to disturbed soils and long-distance seed dispersal (von der Lippe & Kowarik, 2007; Lockwood *et al.*, 2005). These results indicate that strategic surveillance along road networks and livestock routes could substantially strengthen early-warning and rapid-response capacity in Tanzania.

Habitat suitability modelling revealed strong concordance between predicted climatic suitability and observed field prevalence and cover, particularly for *Chromolaena* and *Prosopis*, confirming the validity of Hypotheses 3 and 4. Suitability hotspots corresponded with humid zones in the north-west for *Chromolaena*, highland–basin mosaics for *Parthenium*, and semi-arid northern and central rangelands for *Prosopis*, reflecting species-specific physiological tolerances and disturbance adaptations. The presence of large climatically suitable areas that are not yet invaded, especially for *Prosopis*—indicates an “invasion debt,” suggesting that without intervention these species are likely to expand into new ecological zones in coming years, mirroring patterns observed in Kenya, Ethiopia, and southern Africa (Wakie *et al.*, 2016; Shackleton *et al.*, 2014). Thus, proactive containment efforts are critical to prevent future ecological and socio-economic impacts.

Collectively, these findings confirm all major hypotheses posed in the Introduction: invasion severity and prevalence differ significantly among ecological zones; the three species show distinct ecological preferences; climatic and anthropogenic variables jointly predict their distributions; suitability outputs align with field-observed severity; and well-defined invasion hotspots cluster in regions with predictable climatic and disturbance regimes. These results have strong implications for management.

Targeted interventions should prioritize the Lake Victoria Basin for *Chromolaena* control, northern and central rangelands for *Prosopis* containment, and highland–basin mosaics for *Parthenium* surveillance. Roadside monitoring, biological control, fire management integration, and coordinated multi-agency action, aligned with Tanzania’s emerging invasive species strategies and the Kunming–Montreal Global Biodiversity Framework—can significantly reduce the spread and impacts of these invasive alien plants. The integration of field surveys, spatial analysis, and predictive modelling in this study provides a robust evidence base for designing national IAS management frameworks, advancing early-warning capacity, and improving long-term conservation outcomes.

These results directly align with global IAS governance frameworks that emphasize pathway-based prevention and spatial prioritization for surveillance. The climatic and anthropogenic drivers identified here provide the empirical basis required to operationalize Tanzania’s National Invasive Species Strategy and Action Plan, while also supporting the Kunming–Montreal Global Biodiversity Framework Targets 6 and 7, which call for strengthened early warning systems, reduced propagule pressure along introduction pathways, and restoration of invaded ecosystems. By linking climatic suitability to propagule corridors, our findings offer a scalable template for national monitoring systems increasingly required by multilateral environmental agreements governing biological invasions.

Management implications

Our findings identify ecologically discrete hotspots where management interventions will yield disproportionate conservation benefit. *Chromolaena* control should prioritize the Lake Victoria Basin, where high cover and climatic suitability overlap create self-reinforcing invasion dynamics. *Prosopis* containment is most urgent in the semi-arid northern

rangelands, where high suitability and road-proximity effects suggest ongoing spread along livestock and transport corridors. Targeted surveillance of *Parthenium* is warranted in highland–basin mosaics, particularly where grazing disturbance interacts with intermediate precipitation regimes. These priority areas correspond directly with the National Invasive Species Strategy and Action Plan and provide spatially explicit units for early-warning deployment, biological control trials, and restoration investments. Embedding these spatial priorities into district land-use plans would significantly reduce future invasion debt.

Limitations

Although this study integrates the most comprehensive invasive plant dataset available for Tanzania, several limitations remain. Field sampling intensity varied among ecological zones, which may under-represent invasion pressure in sparsely surveyed regions. Disturbance indicators such as grazing intensity, proximity to farms, and water sources were recorded at plot level but lacked continuous national spatial layers suitable for modelling. Consequently, species distribution models may underestimate localized propagule pressure along riparian corridors and informal livestock pathways. Finally, climatic predictors represent current conditions and do not incorporate projected warming or rainfall shifts that may alter suitability profiles for *Prosopis* and *Parthenium*. Future work should integrate dynamic climate scenarios, livestock-density proxies and remote sensing disturbance metrics to refine predictive accuracy and better anticipate invasion trajectories.

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PLANT SPECIES COMPOSITION AND DIVERSITY OF THE AGA KHAN UNIVERSITY, ARUSHA CAMPUS, NORTHERN TANZANIA

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ABSTRACT

Plant biodiversity data are prerequisites for the sustainable management of ecosystems.

Studies of the composition and diversity of plant species in most ecosystems provide a scientific basis for optimal allocation and regulation of community structure. However, the extent to which human activities alter plant species composition and diversity within ecosystems remains insufficiently understood. The present study investigated the composition and diversity of the current plant species composition in the Aga Khan University Campuses over a period of a decade. We employed line transects and the nested plot technique for data collection. The study findings revealed that a total of 831 individual plants, belonging to 49 families and 237 species, were recorded in the area. This accounted for approximately 95% of the previously recorded plant species in the area. The plant species diversity was not significantly different ($p > 0.05$) among the study sites, except at the Two Rivers site, specifically in the riverine forest. High plant species richness was found in the Two Rivers site compared to the Manyara site. In addition, plant species diversity was not statistically significantly different across all habitats in the area. Analysis of the species co-occurrence matrix revealed that the herbaceous plant *Themeda triandra* exhibited negatively associations with other plant species, probably due to a combination of competitive exclusion, niche modification, and possibly allelopathy interactions. *Ficus sur*, a woody plant species, showed negative associations with most plants in the area due to its aggressive growth behaviour. A range of plant growth forms was recorded including climbers, grass, herbs, sedges, shrubs, and trees, with herbs representing the most species-rich plant growth form. Despite this diversity, the biodiversity of the area might be threatened by the presence of invasive plant species. We therefore recommend the urgent implementation of effective measures to control invasive plant species within the area.

Keywords: biodiversity, conservation, ecosystem, habitat, invasive

INTRODUCTION

Native vegetation plays a crucial role in maintaining ecological balance and supporting biodiversity within an ecosystem (Zamin *et al.* 2025). The composition of plant species within a vegetation community defines its species identity and is recognized as a key

indicator of both ecological processes and site management practices (Behera *et al.*, 2023). According to Behera *et al.* (2023), species diversity encompassing both species richness (number of species in a local community) and species evenness (relative abundance of a species in a community) is a measurable indicator of a community structure that

reflects the fundamental characteristics of an ecosystem. In ecosystems, species are interdependent, either directly or indirectly, with each species exploiting different portion of available resources for survival (Behera *et al.* 2023). A system that is more species-diverse is considered to be more efficient, productive, and sustainable (Behera *et al.* 2023). Therefore, it is crucial to document plant species composition across ecosystems to better understand ecological patterns for informing evidence-based conservation priorities (Behera *et al.* 2023).

Previous studies at the Aga Khan University land site revealed that it is characterized by a broad spectrum of habitat types, including woodlands, riverine forests, grasslands, wooded grasslands, and shrublands, which support diverse flora and fauna (Moshi & Mbago, 2013; Werema *et al.*, 2013). The study by Moshi & Mbago (2013) showed that the Agha Khan University land Site harbors a diverse range of plant species. Plant species, which are predominantly native to the study area, comprise subtropical dry scrub vegetation, which is well-adapted to semi-arid and rocky terrain conditions, supporting thorny shrubs, grasses, and other drought-resistant species (Zamin *et al.* 2025). Plant species are fundamentally important to the environment, as they produce oxygen necessary for aerobic life, which all other living things require for survival and reproduction. Also, plants are the base of most food chains, as they are capable of producing their own food through photosynthesis process, on which other animals depend. They play a critical role in regulating the climate by sequestering greenhouse gases, particularly carbon dioxide, thereby mitigating the adverse effects of climate change. Moreover, plants contribute to soil stability and reduce erosion by maintaining soil cover while providing also essential habitats for other living organisms,

However, anthropogenic activities substantially affect plant abundance, diversity, and distribution, underscoring the necessity

of conservation efforts to preserve native vegetation (Zamin *et al.*, 2025). According to Zamin *et al.* (2025), despite extensive deforestation and habitat loss which have led to reduction in forest cover and the extinction of economically valuable plant species, conservation measures can help mitigate these effects. Previous studies conducted in other regions have demonstrated that ecosystem disturbances can create unfavorable conditions for the successful regeneration of many existing plant species (Behera *et al.*, 2023). However, there is very limited information on how human-induced disturbances influence the composition, diversity, and distribution pattern of the plant species in the Aga Khan University land sites. Therefore, this study aimed first to document the current status of plant species in the area, second, to determine the presence of alien and invasive plant species in the area, and finally to identify and list endangered and vulnerable plant species that need priority conservation to improve biodiversity and ecosystem services of the area.

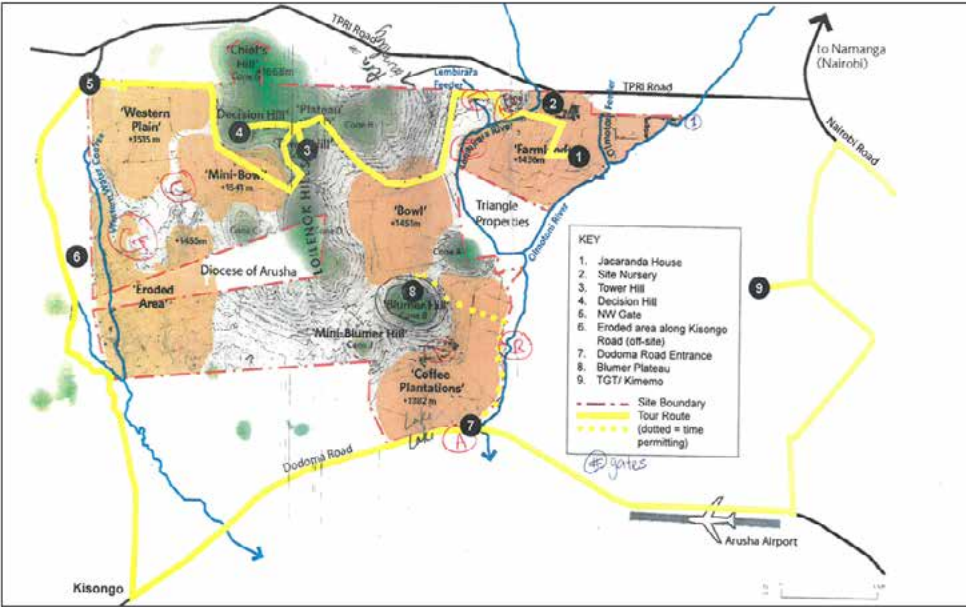
MATERIALS AND METHODS

Study area

The study was conducted at the Agha Khan University Campuses, specifically at the Manyara site and Two Rivers site (Map 1 and 2, respectively). Manyara site is located at approximately 37M 0230000, UTM 9631715 and 37M 0236000, UTM 9631715 at the Kisongo area in Arusha District, about 20 kilometres west of Arusha city centre (Moshi & Mbago, 2013; Map 1). Floristically, the site falls under the phytocoria of the Somali-Maasai regional centre of endemism, whose vegetation is predominantly edaphic grassland and deciduous bushland (Moshi & Mbago, 2013). The natural vegetation of the study site is currently highly disturbed due to soil erosion and anthropogenic pressure, mainly cultivation, over-grazing, and settlements (Moshi & Mbago, 2013).

The study was also conducted in the other Aga Khan University Campus (Two Rivers site), which is 2 km by 1 km in size, about 500 acres, and located between two tributaries of the Usa River (Ntalwila *et al.*, 2024). It is located 14 km east of Arusha city centre (Ntalwila *et al.*, 2024; Map 2). about 80% of land is under cultivation, and the remaining 20% consists of scattered

trees, shrubland, and open grass areas (Ntalwila *et al.*, 2024). The general characteristics of the area include high precipitation, fertile soil, and diverse vegetation. The contrasting environmental conditions of being farmland together with geographic location provide a strong basis and rationale for biodiversity assessment (Ntalwila *et al.*, 2024).



Map 34: AKU Arusha Manyara Site - dotted redline is the study area boundary



Map 35: AKU Arusha Two Rivers Site – left top-down frames are total cultivated and uncultivated areas, respectively. Right top-down frames are grassland and shrubland.

Data collection

The data were collected using a stratified random sampling technique to determine plot positions and directions within each transect. Plot direction was selected by randomly choosing one of two tossed written papers inscribed either “right” or “left”. Following this, sampling quadrats were systematically placed to collect vegetation data in the area. Within each transect, a nested plot technique was employed to collect detailed data. Each transect measured 1km in length and was subdivided into five sampling plots, with plots measuring 20m x 20m size which were placed at 200m intervals (i.e. m, 200m, 400m, 600m, 800m), whereby each plot had three allocated sampling plots from the right corner of the plot (1m x 1m) for herbaceous plants, 5m x 5m for shrubs, and 20m x 20m for trees (Figure 1). Two observers worked together to sample all of the quadrat plots in each sampling site during the survey period. Within each 1m x 1m sub-plot, herbaceous species cover (%) and height (cm) were measured, and species were identified whenever possible. The programme “*PictureThis App*” assisted in identifying all the unknown plants found during the survey. Other information recorded included the date, transect name, GPS location, and plot identification number, site code, plot code, GPS location, herbaceous sp., and shrub sp. Trees sp., their heights (cm), their % cover, average total height (cm), average % total cover, altitude(m), and level of disturbance.

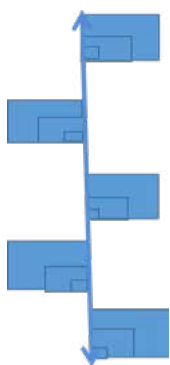


Figure 1: Sketch showing the vegetation transect sampling design.

Data Analysis

Data analyses was performed using R software version 4.3.2. The Shapiro test was used to test the normality of the data. PCA was used to ordinate the plant species composition across land use types according to Legendre and Legendre (2012). An ordination diagram was subsequently generated to visualise species composition in relation to land use type. The plant species diversity among different land use types was determined in terms of the Shannon-Wiener diversity index according to the following formula:

$$\text{Diversity Index (H)} = -\sum p_i \ln p_i$$

Where:

$p_i = n_i/N$ is the proportion of the total number of all species in a quadrat, and $\ln$ = natural logarithm to base e.

Herbaceous plant species associations were analysed using a null model, as described by Griffith *et al.* (2016). Difference in plant diversity amongst land types and habitats were analysed using pooled data for respective land-use types and habitats, employing type III sums of squares in the analysis of variance (ANOVA). The distribution of plant species in different habitats was analysed according to Heberle *et al.* (2015).

RESULTS

A total of 831 individuals belonging to 49 families and 237 plant species were recorded in the area (Appendix 1). These plant species represented 95% of the previously recorded plant species in the area (Moshi & Mbago, 2013). A previous study conducted during the dry season documented 248 plant species within the area (Moshi & Mbago, 2013). The dendrogram (Fig. 2) grouped composition of plant species into 2 clusters. From this study, results indicated that composition of plant species in the Decision Hill and Tower Hill Sites in Manyara was different from the rest of the surveyed sites. This signified the presence of microhabitats along the altitudinal gradient in the two hills.

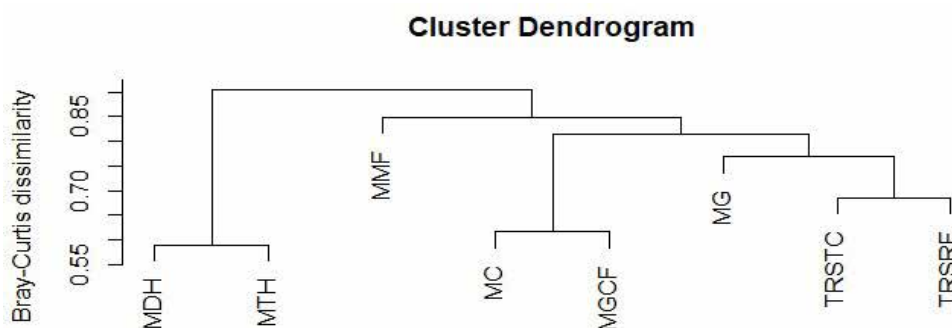


Figure 2: Cluster analysis of plant species composition for different land use types surveyed

The results also showed that diversity of plant species was not significantly ($p > 0.05$) different in the two land use sites, except in the riverine forest in the Two Rivers site (Fig. 3).

The distribution of plant species in the area

High species richness was recorded in transects, namely riverine forest ($n = 297$) and cultivated area ($n = 103$) at the Two Rivers site and cultivated area ($n = 99$) and mixed forest ($n = 92$) at the Manyara site (Fig. 4).

Eight habitat types were identified, including bushed grassland, bushland, cultivation, forest, grassland, riverine, wooded grassland, and woodland. Amongst them, more plant species (35.8%) were found in the riverine habitat than in other habitats (Fig. 5), and their differences were statistically significant, Kruskal-Wallis test ($X^2 = 16.02$, $p = 0.02458$).

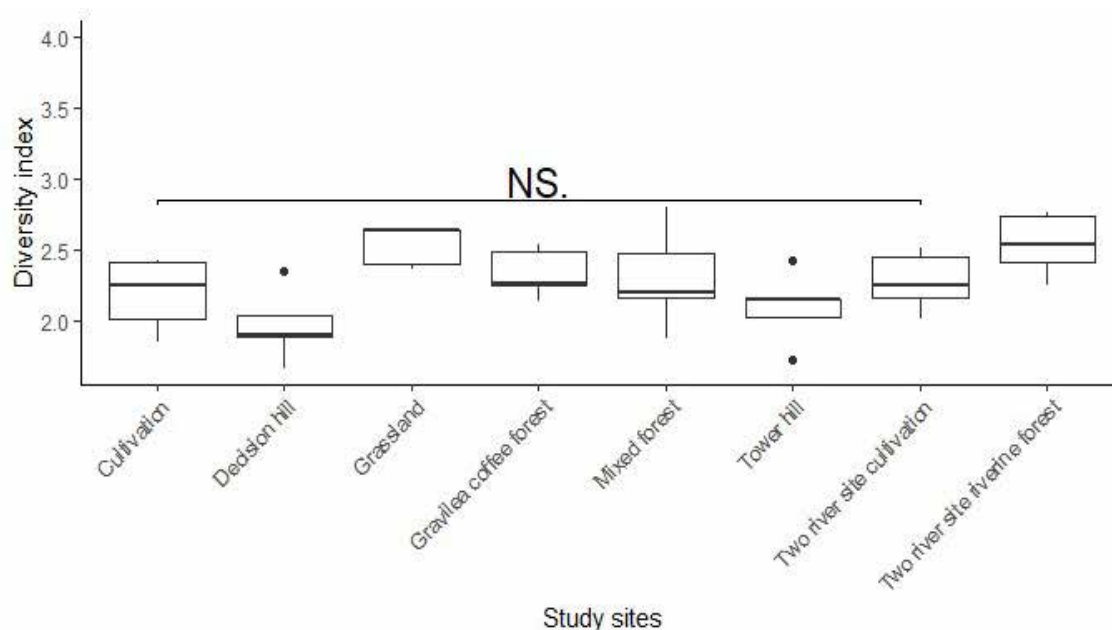


Figure 3: Plant species diversity among different surveyed sites

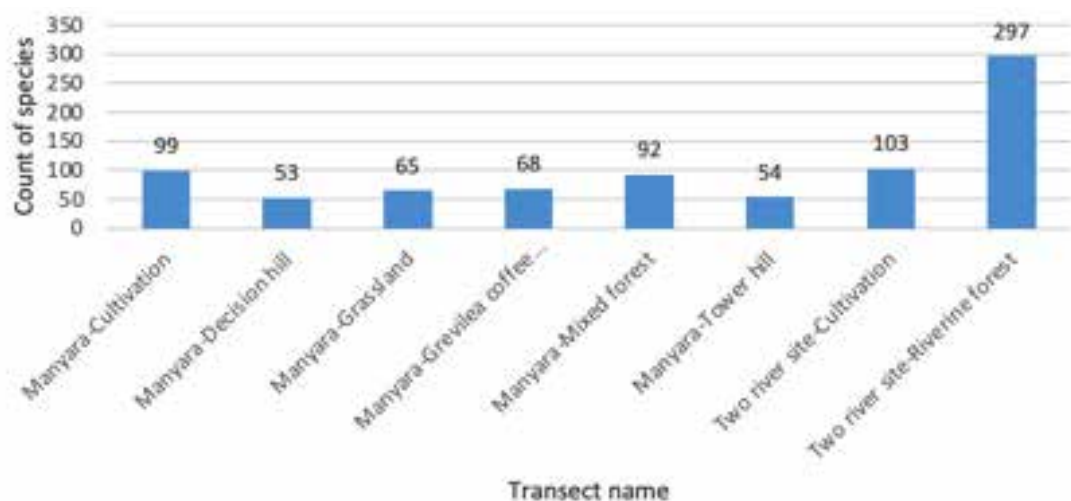


Figure 4: The distribution of the plant species richness across different transects in the areas.

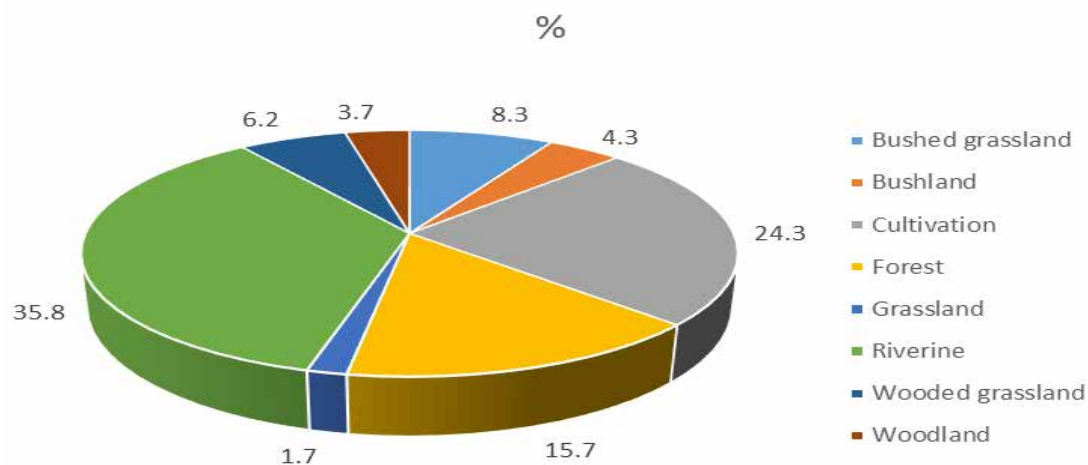


Figure 5: The distribution of plant species between different habitat types in the area

The results show that plant species diversity did not differ significantly different ($p > 0.05$) across the surveyed habitats (Fig. 6). However,

the Riverine habitat showed the relatively highest plant diversity while the grassland had the lowest plant diversity index.

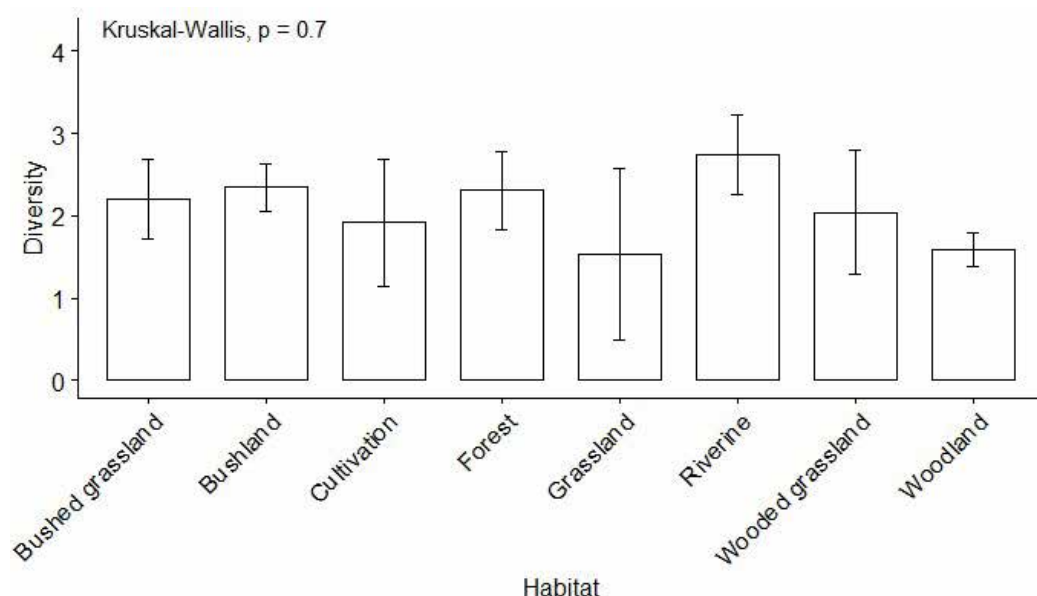


Figure 6: Diversity of plant species in different surveyed habitats

Plant species association

The results indicated that *Themeda triandra*, which is a highly palatable plant species for herbivores, is negatively associated with *Bidens pilosa*, *Commelina benghalensis*, *Cyperus rotundus*, and *Malvastrum coromandelianum* (Fig. 7). The results also revealed that *Gutenburghia cordifolia*, which is an invasive plant species, was negatively associated with *Bidens pilosa*, *Commelina benghalensis*, *Cyperus rotundus*, and *Malvastrum coromandelianum*.

Species Co-occurrence Matrix

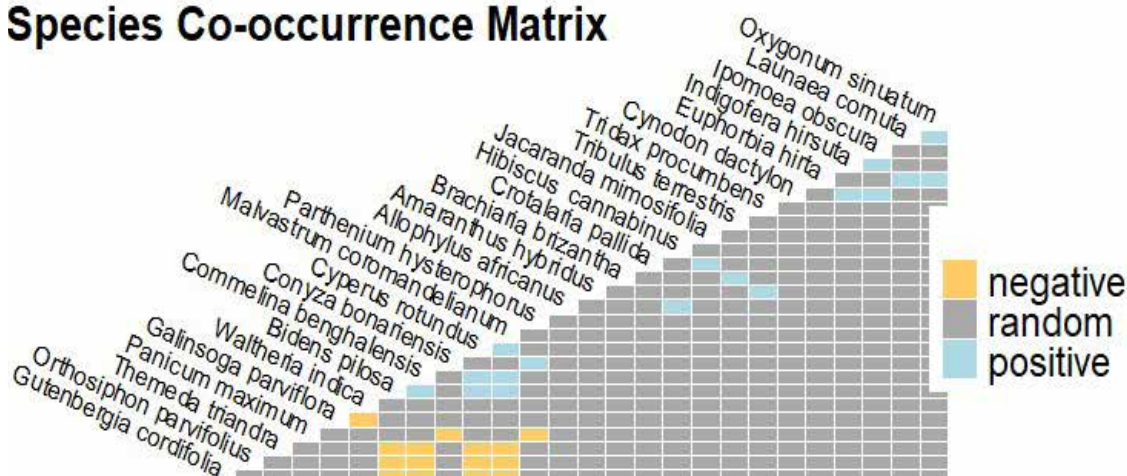


Figure 7: Herbaceous plant species association in different sites surveyed at the AKU sites

The results further indicated that *Ficus sur* is negatively associated with most plants in the area (Fig. 8).

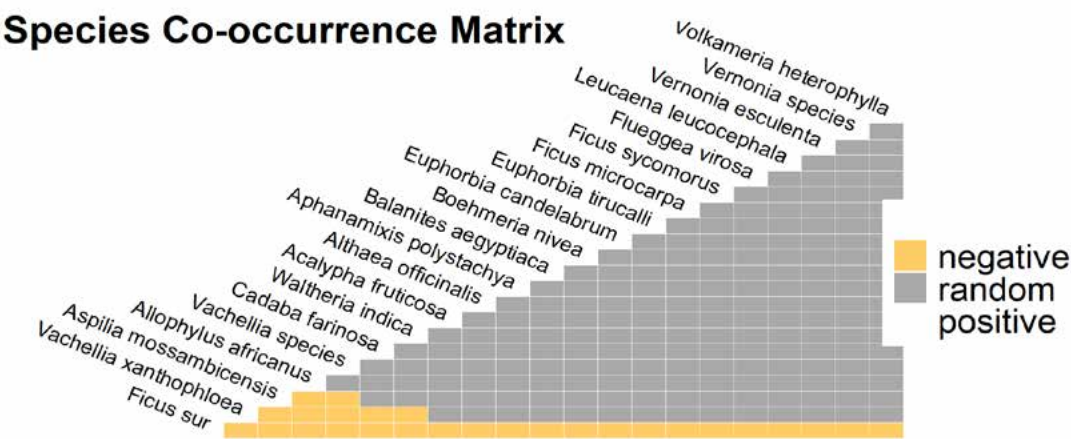


Figure 8: Woody plant species association in different sites surveyed at the AKU sites

Plants' life forms composition

In terms of plant life forms, the study revealed that both sites had different plant species with different growth forms, including, climbers, grass, herbs, sedges, shrubs, and trees. All these plant forms showed different compositions in terms of richness and diversity. Herbs were the richest plant form in the surveyed area with 129 species, followed by shrubs with 74 species,

grasses with 38 species, trees with 26 species, climbers with 7 species, and sedge form was the least with only 3 species. Furthermore, the diversity of plant species was different across these plant forms as herbs were the most diverse species with ($H' = 4.328$), followed by shrubs ($H' = 3.861$), grasses ($H' = 3.268$), tree ($H' = 2.798$), a climber ($H' = 1.631$), and least at sedge with Shannon ($H' = 0.38$; Figure 9).

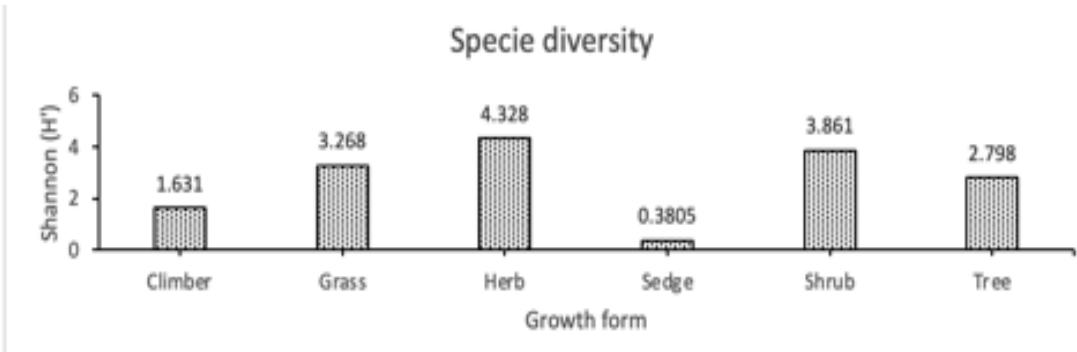


Figure 9: Showing plant species diversity in different plant growth forms at Aga Khan University campus areas, Northern Tanzania.

DISCUSSION

Our findings of 237 plant species from 49 families highlight substantial flora information of the area, underscoring the importance of enhancing comprehensive and proactive

conservation initiatives (Zamin *et al.* 2025). This is because preserving and conserving natural areas and reducing biodiversity loss are core principles of most protected area management programmes (Masubelele *et al.*, 2013). In this study, about 11 plant species

were missed compared to the previous survey (Moshi & Mbago, 2013), probably because the surveys were taken in different seasons or the transects were placed differently within the area. Plant species such as *Kleinia petrae*, *Orbea denboefii*, and *Delonix regia*. *Mangifera indica*, *Kyaya anthotheca*, and *Olea welwitschii* were not recorded by this study but were recorded in a previous study by Moshi & Mbago, (2013). However, our findings still support the idea that the Aga Khan University (AKU) land sites are crucial for plant biodiversity conservation. It has also been reported that biodiversity conservation is not only for protected area management programmes but also in unprotected areas (Masubelele *et al.*, 2013).

Moreover, our findings revealed that plant species diversity did not differ significantly across the study area, though values ranged from $H' = 0.38$ to $H' = 4.328$. By comparison, a previous study by Moshi & Mbago, (2013) reported diversity values ranging from $H' = 1.94$ to $H' = 4.40$. However, the diversity of plant species was different across these plant growth forms, as herbs were the most diverse species, followed by shrubs, grasses, trees, climbers, and sedges. According to Moshi & Mbago (2013), this could be attributed to the fact that these life forms, in most cases, are the primary colonisers of disturbed areas and have short growth cycles. The same pattern was observed during the previous study (Moshi & Mbago, 2013). Moreover, a study of Moshi and Mbago (2013) documented diverse plant growth forms at the AKU land site. In the current study, herb species were the most dominant in the area, with shrubs, grasses, and trees following in diversity. This trend suggests that shrubs, grasses, and trees may exert less competitive pressure on herbs compared to other plant growth forms (Zamin *et al.* 2025). The high species diversity observed in the Riverine forest is vital in maintaining ecosystem sustainability and supporting various life forms. Since it was found that the riparian ecosystems are amongst the most ecologically important ecosystems on Earth, constituting

the crucial link between terrestrial and aquatic habitats (Pastina *et al.*, 2024). In addition, plant species diversity enhance ecosystem stability and resilient, thereby improving the system's capacity to withstand environmental changes and disturbances. Furthermore, it has been found that the decline in plant species diversity may lead to changes in nutrient cycling and associated ecosystem functions in biogeochemical processes, which then result in the reduction and decline of ecosystem services (Mi *et al.*, 2022). The lower species diversity may indicate ecological stresses, habitat degradation, or the encroachment of invasive species. According to Mi *et al.* (2022), the plant species diversity of the community, like that of the AKU, usually affects grassland productivity, belowground carbon distribution, and nutrient transport, which might be influenced by the soil microbial composition. Our results supported the point that diversity of plant species is essential for the stability of ecosystems, as they showed diversity of plant species was not significantly different in all habitats surveyed in the area. Again, our finding supports the point that efforts to conserve plant diversity should extend beyond the protected area boundaries to achieve long-term conservation of nature and its native biodiversity (Zisadza-Gandiwa *et al.*, 2013). Additionally, the dendrogram grouped the AKU plant species composition into 2 clusters and indicated that plant species composition in the Decision Hill and Tower Hill sites of Manyara was different from the rest of the sites surveyed. This might be caused by the difference in elevations experienced in the Decision and Tower hills. Since elevation has a significant impact on plant species composition, as it influences various environmental factors such as temperature, precipitation, and sunlight availability, which in turn affect the types of plants that can grow in a particular area (Masoud *et al.*, 2025). According to Masoud *et al.* (2025), as elevation increases, temperature generally decreases, and precipitation patterns change, leading to distinct vegetation zones at different elevations. Elevation variation encompasses meteorological and physical factors that directly influence community

structure (Masoud *et al.*, 2025). Additionally, elevation gradients are widely recognised as valuable tools for studying the relationship between climate and vegetation (Masoud *et al.*, 2025). According to Masoud *et al.* (2025), this approach is effective because temperature consistently decreases with elevation, making elevation a reliable proxy for understanding temperature variations and their impacts on vegetation. Changes in ecosystem properties, such as plant diversity, productivity, species traits, and physiology, occur along gradients (Masoud *et al.*, 2025).

The survey found three plant species of IUCN Red List Concerns, which are alien but not yet invasive in the area, such as *Brugmansia suaveolens* (Humb. & Bonpl. ex Willd.) Bercht. & J. Presl Solanaceae, which was identified as Extinct in the Wild. This finding differs from the previous one, which found only two plant species, *Aloe ballyi* (Rat Aloe) and *Khaya anthotheca* (African mahogany), falling under the two IUCN categories of Endangered and Vulnerable, respectively, were identified (Moshi & Mbago, 2013). According to Pundir *et al.*, (2022) *B. suaveolens* (Solanaceae), also popularly known as “Angel’s trumpet, Brazil’s white angel trumpet, also known as angel’s tears and snowy angel’s trumpet,” is a flowering shrub, indigenous to South Africa. The plant is cultivated globally as an ornamental plant for its appealing trumpet-shaped white flowers, and there is only one known species of Brugmansia, *Brugmansia suaveolens* (Pundir *et al.*, 2022). However, other associated species have been reported elsewhere (Pundir *et al.*, 2022).

Other plant species found in the area were *Coffea arabica* L. Rubiaceae, identified as endangered, and *Jacaranda mimosifolia* D. Don in Bignoniaceae family as vulnerable by the IUCN Red List. The study by Davis *et al.* (2019) has shown that *C. arabica* has the most thorough extinction risk assessment of any coffee species, due to plentiful high-quality ground point data, rigorous ground truthing, and inclusion of climate change projections. Therefore, according to Davis *et al.* (2019), when

climate change projections are incorporated into the extinction risk assessment, *C. arabica* moves into three categories, from LC to EN.

On the herbaceous plants, it was found that *Themeda triandra* exhibits a negative association with *Bidens pilosa*, *Commelina benghalensis*, *Cyperus rotundus*, and *Malvastrum coromandelianum* in the study area. This finding prompts several ecological interpretations regarding the mechanisms driving species coexistence and competitive exclusion in this ecosystem one explanation centers on competitive interactions. *Themeda triandra* is known for its robust clonal growth and dense tussock formation, characteristics that can monopolise space and resources such as light, water, and nutrients. These resource-based competitive advantages may reduce the establishment and growth of neighboring forbs and smaller herbaceous weeds such as *Bidens pilosa* and *Commelina benghalensis*. In environments where *Themeda triandra* is dominant, its extensive root system tends to limit belowground resource availability for other species like *Cyperus rotundus* and *Malvastrum coromandelianum*, thereby suppressing their proliferation through competitive exclusion (Connell, 1983).

A second perspective involves niche differentiation and environmental modification. *Themeda triandra*, by establishing a dense canopy and altering soil microclimates (e.g., moisture retention and temperature), may create conditions that are suboptimal for certain coexisting species. For example, *Bidens pilosa* and *Commelina benghalensis* often require high-light or disturbed conditions for successful germination and growth. They may be disadvantaged in the relatively stable, shaded conditions under *Themeda triandra* canopies (Grime, 2001). Such micro-environmental modifications can drive ecological filtering process, whereby only species possessing compatible or complementary functional traits are able to coexist within the same habitat (Keddy, 1992).

Furthermore, allelopathy is another potential mechanism. Some studies have suggested that certain grasses can release biochemicals into the soil that inhibit the germination or growth of neighboring plants (Callaway and Aschehoug, 2000). Although allelopathic interactions have been more frequently reported in some invasive species, similar processes could conceivably occur with native species like *Themeda triandra*, contributing to its negative associations with the aforementioned species.

It is also important to consider that the observed spatial patterns might partly reflect differences in ecological preferences or tolerance to soil conditions and disturbance regimes. For example, *Cyperus rotundus* is a species adapted to disturbed conditions with shallow root systems, and its distribution might be limited in areas where *Themeda triandra* has improved soil structure and moisture retention through its dense root system. Similarly, *Malvastrum coromandelianum*, often associated with ruderal or edge habitats, might find such habitats less prevalent in areas occupied by the perennial and competitive *Themeda triandra* (Myster, 2002). The negative association of *Themeda triandra* with *Bidens pilosa*, *Commelina benghalensis*, *Cyperus rotundus*, and *Malvastrum coromandelianum* appears to result from a combination of competitive exclusion, niche modification, and possibly allelopathic interactions. These findings add to the growing body of evidence that spatial associations in plant communities are often the cumulative result of multiple interacting ecological processes.

The woody plant species *Ficus sur* exhibited negatively associated with most co-occurring plants in the area, likely due to its aggressive growth behavior of displacing other plant species. Moreover, the species is a large, fast-growing, evergreen tree, reaching up to 35 m high, aggressive to other woody plants in a riverine habitat due to its robust and invasive root system, which can physically damage or overwhelm other plants. Despite its high spatial requirements for growth and persistence,

the species provides important ecological benefits, including serving as a food source and habitat for urban fauna (birds, bats, and small mammals), contributing to microclimate regulation, and supporting water conservation (Siswo *et al.* 2019).

Biodiversity of the area was found to face considerable threats, including biological invasion caused by invasive plant species such as *Guntebergia cordifolia*, *Lantana camara*, *Lippia javanica*, *Parthenium hysterophorus* L., *Ricinus communis*, *Argemone mexicana*, and *Tagetes minuta*, more specifically in the Manyara site. It has been previously revealed that invasive plant species are currently considered to be one of the main drivers of biodiversity loss worldwide (Nininahazwe *et al.*, 2023). According to Kumar & Singh (2020), invasive plant species are often more competitive for resources than native species, causing a reduction or extinction of native vegetation, a reduction of primary productivity, and an alteration of ecosystem functions. Thus, plant species invasion may have pronounced effects on agricultural productivity, which constitutes a significant component of the economy of our country.

CONCLUSION AND RECOMMENDATIONS

This study demonstrates that the Aga Khan University (AKU) land sites are of considerable important for biodiversity conservation, as they support 237 plant species across 49 families. These results emphasize the need to increase efforts to conserve diversity of plant species beyond the AKU boundaries to achieve long-term conservation of nature and its native biodiversity. Despite this, the biodiversity conservation of the area may be threatened by the presence of invasive plant species. This may be due to the intense land use that has impacted and altered the natural environment and threatened the vegetation communities of the area. Therefore, we recommend the urgent implementation of effective management strategies to control invasive plant species

within the area, alongside systematic evaluation of their effectiveness.

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Appendix 1. Showing the plant species checklist of the Aga khan University campus areas, Northern Tanzania.

IUCN Key: **EW**= Extinct in the Wild (EW) when it is known only to survive in cultivation, **EN**= Endangered; **LC**=Less Concern; **NA**= Not Applicable; **VU**=Vulnerable;

SN	Scientific name	Family	IUCN	Climber	Grass	Herb	Sedge	Shrub	Tree	Total
1	<i>Abutilon grandifolium</i> (Willd.) Sweet	Malvaceae	NA			2				2
2	<i>Abutilon</i> sp	Malvaceae	NA			1				1
3	<i>Acacia</i> sp	Fabaceae	NA						1	1
4	<i>Acalypha fruticosa</i> Forssk.	Euphorbiaceae	LC					2		2
5	<i>Acalypha ornata</i> Hochst. ex A.Rich.	Euphorbiaceae	LC					1		1
6	<i>Acanthospermum hispidum</i> DC.	Asteraceae	NA			3				3
7	<i>Achyranthes aspera</i> L.	Amaranthaceae	NA			3				3
8	<i>Acrocarpus fraxinifolius</i> Wight & Arn.	Fabaceae	NA						2	2
9	<i>Ageratum conyzoides</i> L.	Asteraceae	NA		1	4				5
10	<i>Ajuga genevensis</i> L.	Lamiaceae	NA			1				1
11	<i>Albizia gummifera</i> (J.F.Gmel.) C.A.Sm.	Fabaceae	LC						2	2
12	<i>Albizia</i> sp.	Fabaceae	NA			4		6	4	14
13	<i>Allium</i> sp	Amaryllidaceae	NA			1				1
14	<i>Allophylus africanus</i> P. Beauv.	Sapindaceae	LC					3		3
15	<i>Alternanthera pungens</i> Kunth	Amaranthaceae	NA			6				6
16	<i>Althaea officinalis</i> L.	Malvaceae	NA					2		2
17	<i>Alyscarpus</i> sp	Fabaceae	NA			3				3
18	<i>Amaranthus hybridus</i> L.	Amaranthaceae	NA			3				3
19	<i>Aphanamixis polystachya</i> (Wall.) R. Parker	Meliaceae	LC					1		1
20	<i>Arabidopsis</i> sp	Brassicaceae	NA			1				1
21	<i>Argemone mexicana</i> L.	Papaveraceae	NA			1				1
22	<i>Aristida adscensionis</i> L.	Gramineae	NA		1					1

SN	Scientific name	Family	IUCN	Climber	Grass	Herb	Sedge	Shrub	Tree	Total
23	<i>Artemisia californica</i> Less.	Asteraceae	NA			1				1
24	<i>Asparagus setaceus</i> (Kunth) Jessop	Asparagaceae	NA			1				1
25	<i>Aspilia mossambicensis</i> (Oliv.) Wild	Asteraceae	NA			2		5		7
26	<i>Balanites aegyptiaca</i> (L.) Delile	Zygophyllaceae	LC					1		1
27	<i>Bidens pilosa</i> L.	Asteraceae	NA			18				18
28	<i>Blumea balsamifera</i> (L.) DC.	Asteraceae	LC					1		1
29	<i>Boehmeria nivea</i> (L.) Gaudich.	Urticaceae	NA					1		1
30	<i>Boerhavia erecta</i> L.	Nyctaginaceae	NA			1				1
31	<i>Bothriochloa ischaemum</i> (L.) Keng	Gramineae	NA		5					5
32	<i>Brachiaria brizantha</i> (A.Rich.) Stapf	Gramineae	NA		9					9
33	<i>Brachiaria ruziziensis</i> R.Germ. & C.M.Evrard	Gramineae	NA		1					1
34	<i>Brachiaria sp</i>	Gramineae	NA		3					3
35	<i>Brassica tournefortii</i> Gouan	Brassicaceae	NA			1				1
36	<i>Brugmansia suaveolens</i> (Humb. & Bonpl. ex Willd.) Bercht. & J. Presl	Solanaceae	EW					1		1
37	<i>Cadaba farinosa</i> Forssk.	Capparaceae	LC					4		4
38	<i>Cadaba sp</i>	Capparaceae	NA					1		1
39	<i>Caesalpinia decapetala</i> (Roth) Alston	Fabaceae	LC					4		4
40	<i>Calliandra haematocephala</i> Hassk.	Fabaceae	NA					1		1
41	<i>Camassia scilloides</i> (Raf.) Cory	Asparagaceae	NA			1				1
42	<i>Cananga odorata</i> (Lam.) Hook. f. & Thomson	Annonaceae	LC			1			1	2
43	<i>Capparis tomentosa</i> Lam.	Capparaceae	NA	2				1		3
44	<i>Carissa spinarum</i> L.	Apocynaceae	LC					2		2
45	<i>Celtis sinensis</i> Pers.	Cannabaceae	LC					2		2
46	<i>Cenchrus ciliaris</i> L.	<i>Cenchrus ciliaris</i> L.	LC		1					1
47	<i>Cenchrus polystachios</i> (L.) Morrone	Gramineae	LC		1					1

SN	Scientific name	Family	IUCN	Climber	Grass	Herb	Sedge	Shrub	Tree	Total
48	<i>Centella asiatica</i> (L.) Urb.	Umbelliferae	LC			1				1
49	<i>Chamaecrista nictitans</i> (L.) Moench	Fabaceae	LC			1				1
50	<i>Chiococca alba</i> (L.) Hitchc.	Rubiaceae	LC			1				1
51	<i>Chloris gayana</i> Kunth	Gramineae	NA		1					1
52	<i>Cichorium intybus</i> L.	Asteraceae	NA			1				1
53	<i>Citharexylum spinosum</i> L.	Verbenaceae	LC					1		1
54	<i>Coffea arabica</i> L.	Rubiaceae	EN					12		12
55	<i>Coleataenia anceps</i> (Michx.) Soreng	Gramineae	NA		2					2
56	<i>Combretum molle</i> R. Br. ex G. Don	Combretaceae	LC					2		2
57	<i>Commelina africana</i> L.	Commelinaceae	LC			5				5
58	<i>Commelina benghalensis</i> L.	Commelinaceae	LC			15				15
59	<i>Convolvulus arvensis</i> L.	Convolvulaceae	NA	1						1
60	<i>Conyza bonariensis</i> (L.) Cronquist	Asteraceae	NA			9				9
61	<i>Conyza floribunda</i> Kunth	Asteraceae	NA			2				2
62	<i>Corchorus tridens</i> L.	Malvaceae	NA			6				6
63	<i>Cordia africana</i> Lam.	Boraginaceae	LC						4	4
64	<i>Crepis</i> sp	Asteraceae	NA			5				5
65	<i>Crotalaria juncea</i> L.	Fabaceae	NA			3				3
66	<i>Crotalaria pallida</i> Aiton	Fabaceae	NA			3		2		5
67	<i>Crotalaria pumila</i> Ortega	Fabaceae	LC			3				3
68	<i>Crotalaria</i> sp	Fabaceae	NA			2				2
69	<i>Croton macrostachyus</i> Hochst. ex Delile	Euphorbiaceae	LC					2	4	6
70	<i>Croton megalocarpus</i> Hutch.	Euphorbiaceae	LC					2	3	5
71	<i>Cucumis melo</i> L.	Cucurbitaceae	NA	1						1
72	<i>Cucurbita moschata</i> (Duchesne) Duchesne ex Poir.	Cucurbitaceae	NA	1						1
73	<i>Cyclospermum leptophyllum</i> (Pers.) Sprague	Apiaceae	NA			1				1

SN	Scientific name	Family	IUCN	Climber	Grass	Herb	Sedge	Shrub	Tree	Total
74	<i>Cynodon dactylon</i> (L.) Pers.	Gramineae	NA		12					12
75	<i>Cynodon plectostachyus</i> (K.Schum.) Pilg.	Gramineae	NA		7					7
76	<i>Cyperus rotundus</i> L.	Cyperaceae	LC			1	19			20
77	<i>Dactyloctenium aegyptium</i> (L.) Willd.	Gramineae	NA		1					1
78	<i>Datura stramonium</i> L.	Solanaceae	NA			2		1		3
79	<i>Desmodium tortuosum</i> (Sw.) DC.	Fabaceae	NA			1				1
80	<i>Dichanthium annulatum</i> (Forssk.) Stapf	Gramineae	NA		2					2
81	<i>Dichrostachys cinerea</i> (L.) Wight & Arn.	Fabaceae	LC					1		1
82	<i>Digitaria abyssinica</i> (Hochst. ex A.Rich.) Stapf	Gramineae	NA		1					1
83	<i>Digitaria sanguinalis</i> (L.) Scop.	Gramineae	LC		7					7
84	<i>Dodonaea viscosa</i> (L.) Jacq.	Sapindaceae	LC					1		1
85	<i>Duranta erecta</i> L.	Verbenaceae	LC			1				1
86	<i>Echinochloa colonum</i> (L.) Link	Gramineae	LC		7					7
87	<i>Eclipta prostrata</i> (L.) L.	Asteraceae	LC			1				1
88	<i>Eleusine indica</i> (L.) Gaertn.	Gramineae	LC		4					4
89	<i>Eragrostis superba</i> Peyr.	Gramineae	NA		1					1
90	<i>Eragrostis tenuifolia</i> (A.Rich.) Hochst. ex Steud.	Gramineae	NA		2					2
91	<i>Eranthemum pulchellum</i> Andrews	Acanthaceae	NA			4				4
92	<i>Erythrina abyssinica</i> Lam.	Fabaceae	LC						1	1
93	<i>Eucalyptus globulus</i> Labill.	Myrtaceae	LC						18	18
94	<i>Euonymus</i> sp.	Celastraceae	NA			1				1
95	<i>Eupatorium capillifolium</i> (Lam.) Small	Asteraceae	NA			1				1
96	<i>Euphorbia candelabrum</i> Welw.	Euphorbiaceae	LC						1	1
97	<i>Euphorbia heterophylla</i> L.	Euphorbiaceae	LC			1				1
98	<i>Euphorbia hirta</i> L.	Euphorbiaceae	NA		1	12				13
99	<i>Euphorbia tirucalli</i> L.	Euphorbiaceae	LC					1		1

SN	Scientific name	Family	IUCN	Climber	Grass	Herb	Sedge	Shrub	Tree	Total
100	<i>Evolvulus alsinoides</i> (L.) L.	Convolvulaceae	NA			1				1
101	<i>Ficus microcarpa</i> L. fil.	Moraceae	LC						1	1
102	<i>Ficus sur</i> Forssk.	Moraceae	LC					1	7	8
103	<i>Ficus sycomorus</i> L.	Moraceae	LC						1	1
104	<i>Flueggea virosa</i> (Roxb. ex Willd.) Royle	Phyllanthaceae	LC					2		2
105	<i>Galinsoga parviflora</i> Cav.	Asteraceae	NA			11				11
106	<i>Gentiana</i> sp	Gentianaceae	NA			2				2
107	<i>Gomphrena serrata</i> L.	Amaranthaceae	NA			1				1
108	<i>Grevillea robusta</i> A. Cunn. ex R. Br.	Proteaceae	LC						17	17
109	<i>Grewia bicolor</i> Juss.	Malvaceae	NA					1		1
110	<i>Grewia biloba</i> G. Don	Malvaceae	LC			1				1
111	<i>Grewia</i> sp	Malvaceae	NA					1		1
112	<i>Gutenbergia cordifolia</i> Benth. ex Oliv.	Asteraceae	NA			2				2
113	<i>Gutenbergia</i> sp	Asteraceae	NA			1				1
114	<i>Gymnocladus</i> sp	Fabaceae	NA					1		1
115	<i>Hedychium</i> sp	Zingiberaceae	NA					1		1
116	<i>Heliotropium steudneri</i> Vatke	Boraginaceae	NA			1				1
117	<i>Herissantia crispa</i> (L.) Brizicky	Malvaceae	NA			2				2
118	<i>Heteropogon contortus</i> (L.) P.Beauv. ex Roem. & Schult.	Gramineae	NA		1					1
119	<i>Hibiscus denudatus</i> Benth.	Malvaceae	NA					1		1
120	<i>Hibiscus rosa-sinensis</i> L.	Malvaceae	NA					1		1
121	<i>Hibiscus</i> sp	Malvaceae	NA			3				3
122	<i>Hoslundia opposita</i> Vahl	Lamiaceae	NA					2		2
123	<i>Hypoestes</i> sp	Acanthaceae	NA			1				1
124	<i>Indigofera astragalina</i> DC.	Fabaceae	NA			1				1

SN	Scientific name	Family	IUCN	Climber	Grass	Herb	Sedge	Shrub	Tree	Total
125	<i>Indigofera erecta</i> Thunb.	Fabaceae	NA			1				1
126	<i>Indigofera hirsuta</i> L.	Fabaceae	NA			14				14
127	<i>Indigofera miniata</i> Ortega	Fabaceae	LC			1				1
128	<i>Indigofera pallida</i> Craib	Fabaceae	NA			1				1
129	<i>Indigofera</i> sp	Fabaceae	NA					1		1
130	<i>Indigofera suffruticosa</i> Mill.	Fabaceae	NA			3		1		4
131	<i>Indigofera tinctoria</i> L.	Fabaceae	NA			2				2
132	<i>Indigofera volkensii</i> Taub.	Fabaceae	NA			2				2
133	<i>Ipomoea</i> sp	Convolvulaceae	NA	6						6
134	<i>Jacaranda mimosifolia</i> D. Don	Bignoniaceae	VU					1	10	11
135	<i>Jacquemontia tamnifolia</i> (L.) Griseb.	Convolvulaceae	NA	1						1
136	<i>Justicia exigua</i> S. Moore	Acanthaceae	NA			1				1
137	<i>Justicia</i> sp	Acanthaceae	NA			3				3
138	<i>Lactuca sativa</i> L.	Asteraceae	NA			2				2
139	<i>Lantana camara</i> L.	Verbenaceae	NA			1		16		17
140	<i>Launaea cornuta</i> (Hochst. ex Oliv. & Hiern) C. Jeffrey	Asteraceae	NA			10				10
141	<i>Leonotis nepetifolia</i> (L.) R.Br.	Lamiaceae	NA			2				2
142	<i>Lepidagathis incurva</i> Buch.-Ham. ex D. Don	Acanthaceae	NA			1				1
143	<i>Leucaena leucocephala</i> (Lam.) de Wit	Fabaceae	NA			1		1	1	3
144	<i>Leucas martinicensis</i> (Jacq.) R.Br.	Lamiaceae	NA			1				1
145	<i>Lippia javanica</i> (Burm.f.) Spreng.	Verbenaceae	NA			7		13		20
146	<i>Malva parviflora</i> L.	Malvaceae	NA			1				1
147	<i>Malvastrum coromandelianum</i> (L.) Garcke	Malvaceae	NA			15				15
148	<i>Malvaviscus arboreus</i> Cav.	Malvaceae	LC			4		4		8
149	<i>Melicoccus</i> sp	Sapindaceae	NA			1				1
150	<i>Melinis repens</i> (Willd.) Zizka	Gramineae	NA		2					2

SN	Scientific name	Family	IUCN	Climber	Grass	Herb	Sedge	Shrub	Tree	Total
151	<i>Millettia pachycarpa</i> Benth.	Fabaceae	LC						2	2
152	<i>Mimosa quadrivalvis</i> L.	Fabaceae	LC			1				1
153	<i>Momordica charantia</i> L.	Cucurbitaceae	NA			1				1
154	<i>Montanoa hibiscifolia</i> (Benth.) K. Koch	Asteraceae	NA			1		3		4
155	<i>Neonotonia wightii</i> (Wight & Arn.) J.A.Lackey	Fabaceae	LC			13				13
156	<i>Newtonia buchananii</i> (Baker) G.C.C.Gilbert & Boutique	Fabaceae	LC			1		2	2	5
157	<i>Ocimum basilicum</i> L.	Lamiaceae	NA			2				2
158	<i>Ocimum suave</i> Willd.	Lamiaceae	NA		1	5				6
159	<i>Ocimum tenuiflorum</i> L.	Lamiaceae	NA			1				1
160	<i>Oplismenus hirtellus</i> (L.) P.Beauv.	Gramineae	NA		4					4
161	<i>Orthosiphon</i> sp	Lamiaceae	NA			4				4
162	<i>Oxalis corniculata</i> L.	Oxalidaceae	NA			6				6
163	<i>Oxygonum sinuatum</i> (Hochst. & Steud. ex Meisn.) Dammer	Polygonaceae	NA			15				15
164	<i>Panicum dichotomiflorum</i> Michx.	Gramineae	NA		1					1
165	<i>Panicum maximum</i> Jacq.	Gramineae	NA		4					4
166	<i>Pappea capensis</i> Eckl. & Zeyh.	Sapindaceae	LC						1	1
167	<i>Parthenium hysterophorus</i> L.	Asteraceae	NA			8				8
168	<i>Pennisetum mezianum</i> Leeke	Gramineae	LC		2					2
169	<i>Pennisetum</i> sp	Gramineae	NA		1					1
170	<i>Pentanisia ouranogyne</i> S.Moore	Rubiaceae	NA			2				2
171	<i>Phyllanthus</i> sp	Phyllanthaceae	NA			3				3
172	<i>Phyllanthus tenellus</i> Roxb.	Phyllanthaceae	NA			1		1		2
173	<i>Phyllanthus urinaria</i> L.	Phyllanthaceae	NA			2				2
174	<i>Piper</i> sp	Fabaceae	NA					1		1
175	<i>Pluchea odorata</i> (L.) Cass.	Asteraceae	NA			2		3		5

SN	Scientific name	Family	IUCN	Climber	Grass	Herb	Sedge	Shrub	Tree	Total
176	<i>Portulaca quadrifida</i> L.	Portulacaceae	NA			1				1
177	<i>Priva lappulacea</i> (L.) Pers.	Verbenaceae	NA			1				1
178	<i>Pueraria montana</i> (Lour.) Merr.	Fabaceae	NA			1				1
179	<i>Rauvolfia caffra</i> Sond.	Apocynaceae	LC			2		1	1	4
180	<i>Rhus aromatica</i> Aiton	Anacardiaceae	NA					1		1
181	<i>Rhynchosia minima</i> (L.) DC.	Fabaceae	LC			26				26
182	<i>Rhynchospora colorata</i> (L.) H.Pfeiff.	Cyperaceae	LC				1			1
183	<i>Richardia scabra</i> L.	Rubiaceae	NA			3				3
184	<i>Ricinus communis</i> L.	Euphorbiaceae	NA					1		1
185	<i>Ruellia caroliniensis</i> (J. F. Gmel.) Steud.	Acanthaceae	NA			1				1
186	<i>Ruellia patula</i> Jacq.	Acanthaceae	LC			2				2
187	<i>Schkuhria pinnata</i> (Lam.) Kuntze	Asteraceae	NA			3				3
188	<i>Scutia myrtina</i> (Burm. fil.) Kurz	Rhamnaceae	LC					1		1
189	<i>Senegalia mellifera</i> (Vahl) Seigler & Ebinger	Fabaceae	NA					1		1
190	<i>Senegalia polyacantha</i> (Willd.) Seigler & Ebinger	Fabaceae	NA						5	5
191	<i>Senna bicapsularis</i> (L.) Roxb.	Fabaceae	LC					1		1
192	<i>Senna spectabilis</i> (DC.) H.S.Irwin & Barneby	Fabaceae	LC						5	5
193	<i>Sesamum</i> sp	Pedaliaceae	NA					1		1
194	<i>Setaria palmifolia</i> (J.Koenig) Stapf	Gramineae	NA		1					1
195	<i>Setaria pumila</i> (Poir.) Roem. & Schult.	Gramineae	NA		1					1
196	<i>Setaria sphacelata</i> (Schumach.) Stapf & C.E.Hubb. ex Moss	Gramineae	LC		2					2
197	<i>Setaria verticillata</i> (L.) P.Beauv.	Gramineae	NA		2					2
198	<i>Setaria viridis</i> (L.) P.Beauv.	Gramineae	NA		1					1
199	<i>Sida acuta</i> Burm. fil.	Malvaceae	NA			6				6
200	<i>Sida ovata</i> Forssk.	Malvaceae	NA			1				1

SN	Scientific name	Family	IUCN	Climber	Grass	Herb	Sedge	Shrub	Tree	Total
201	<i>Sida sp</i>	Malvaceae	NA			1				1
202	<i>Solanum incanum L.</i>	Solanaceae	LC			1		5		6
203	<i>Sorghum halepense (L.) Pers.</i>	Gramineae	NA		1					1
204	<i>Sphaeranthus suaveolens (Forssk.) DC.</i>	Asteraceae	NA			8				8
205	<i>Spinacia oleracea L.</i>	Amaranthaceae	NA			1				1
206	<i>Sporobolus africanus (Poir.) Robyns & Tournay</i>	Gramineae	NA		4					4
207	<i>Syzygium guineense (Willd.) DC.</i>	Myrtaceae	LC					1		1
208	<i>Tabernaemontana ventricosa Hochst. ex A. DC.</i>	Apocynaceae	LC			1		6	1	8
209	<i>Tagetes minuta L.</i>	Asteraceae	NA			15		2		17
210	<i>Tanacetum balsamita L.</i>	Asteraceae	NA			1				1
211	<i>Tarenna mollissima (Hook. & Arn.) B.L.Rob.</i>	Rubiaceae	LC					1		1
212	<i>Themeda triandra Forssk.</i>	Gramineae	NA		8					8
213	<i>Thladiantha dubia Bunge</i>	Cucurbitaceae	NA	1						1
214	<i>Toddalia asiatica (L.) Lam.</i>	Rutaceae	NA					1		1
215	<i>Tragus berteronianus Schult.</i>	Gramineae	NA		1					1
216	<i>Tribulus terrestris L.</i>	Zygophyllaceae	LC			5				5
217	<i>Trichilia emetica (Forssk.) Vahl</i>	Meliaceae	LC					1		1
218	<i>Trichodesma zeylanicum (Burm. fil.) R. Br.</i>	Boraginaceae	NA			1				1
219	<i>Tridax procumbens L.</i>	Asteraceae	NA			5	1			6
220	<i>Triumfetta rhomboidea Jacq.</i>	Malvaceae	NA			2		1		3
221	<i>Tropaeolum majus L.</i>	Tropaeolaceae	NA			1				1
222	<i>Vachellia nilotica (L.) P.J.H.Hurter & Mabb.</i>	Fabaceae	LC					7	5	12
223	<i>Vachellia sp</i>	Fabaceae	NA			1		3		4
224	<i>Vachellia xanthophloea (Benth.) Banfi & Galasso</i>	Fabaceae	LC					1	4	5
225	<i>Verbascum blattaria L.</i>	Scrophulariaceae	NA			1				1
226	<i>Verbena brasiliensis Vell.</i>	Verbenaceae	NA			1				1

SN	Scientific name	Family	IUCN	Climber	Grass	Herb	Sedge	Shrub	Tree	Total
227	<i>Vernonia esculenta</i> Hemsl. ex F. B. Forbes & Hemsl.	Asteraceae	LC					1		1
228	<i>Vernonia sp</i>	Asteraceae	NA					1		1
229	<i>Vinca major</i> L.	Apocynaceae	NA			1				1
230	<i>Vinca minor</i> L.	Apocynaceae	NA			1				1
231	<i>Volkameria heterophylla</i> Poir.	Lamiaceae	NA					1		1
232	<i>Waltheria indica</i> L.	Malvaceae	LC			8		4		12
233	<i>Withannia sp</i>	Solanaceae	NA			1				1
234	<i>Xanthium orientale</i> L.	Asteraceae	NA			2				2
235	<i>Zanthoxylum fagara</i> (L.) Sarg.	Rutaceae	LC			1		2		3
236	<i>Zinnia elegans</i> Jacq.	Asteraceae	NA			1				1
237	<i>Ziziphus mucronata</i> Willd.	Rhamnaceae	LC					1		1
	Grand Total			13	107	418	21	168	104	831

BIODIVERSITY CONSERVATION IN WETLANDS THROUGH INTEGRATED WATER RESOURCES MANAGEMENT: KEY CHALLENGES, OPPORTUNITIES, AND LESSONS FROM THE KATUMA CATCHMENT, TANZANIA

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ABSTRACT

Wetlands in the Katuma Catchment of western Tanzania are facing accelerated ecological degradation due to hydrological changes, rapid land-use expansion, and inadequate institutional coordination. This study employed a mixed-methods approach, combining hydrological records (2016–2024), satellite data on wetland trends, ecological field assessments, and 12 stakeholder interviews, to analyse the challenges and opportunities for integrating biodiversity conservation into the Integrated Water Resources Management (IWRM) framework. Findings indicate a 23–31% reduction in wetland area over the past two decades, along with significant decreases in rainfall and river discharge, including four years with markedly below-average rainfall (2016, 2017, 2018, and 2021). Dry-season flows now end 1–2 months earlier than a decade ago, and long-term data show a statistically significant decline ($p < 0.05$) in both groundwater recharge and base flow conditions. Ecological assessments reveal increased sedimentation, vegetation loss, and habitat fragmentation, which are crucial for breeding and foraging, with some hippo and elephant use areas reduced by over 40%. Livestock numbers rose by nearly 180% between 2012 and 2024, worsening trampling, erosion, and water quality issues. Over 70% of surveyed villages reported agricultural expansion into riparian zones, while unregulated artisanal mining continues to pose a new threat. Institutional analysis identified ongoing coordination gaps among LRBWB, TANAPA, district councils, and Water User Associations, with less than 35% of planned joint monitoring activities implemented due to overlapping mandates and limited resources. Despite these obstacles, increased community awareness, active basin structures, and ongoing restoration projects offer strong opportunities for integrated management. The study recommends a practical framework that connects adaptive water governance, ecosystem-based restoration, and community participation as complementary pathways to enhance wetland resilience and support national commitments to SDGs 6, 13, 14, and 15.

Keywords: Biodiversity, Wetland conservation, Integrated Water Resources Management, Katuma catchment, Climate variability, and Water governance

INTRODUCTION

Wetlands are among the world's most productive ecosystems, supporting biodiversity, regulating water cycles, and sustaining local communities (Mitsch *et al.*, 2015). Globally, they host nearly 40% of all species but continue

to decline quickly; more than 35% have been lost since 1970 due to land-use change, water extraction, pollution, and weak governance (Davidson, 2014; Gardner *et al.*, 2020). This loss reduces ecological resilience and harms the livelihoods of communities that depend on wetland services. Across sub-Saharan Africa,

wetlands provide vital water, grazing resources, fisheries, and support for agriculture. However, these ecosystems face increasing pressure from farming, deforestation, urban expansion, and irrigation development (Rebelo *et al.*, 2010). These environmental threats are worsened by fragmented institutional systems that often separate water management from biodiversity conservation (Emerton *et al.*, 2021). Integrated Water Resources Management (IWRM) is promoted as a comprehensive approach to addressing these issues; however, its implementation remains weak in many developing countries, including Tanzania, due to capacity limitations and lack of stakeholder coordination (Tortajada & Biswas, 2016).

Tanzania's wetlands, which cover about 10% of the country, are vital for biodiversity conservation and supporting local economies (URT, 2014). The Katuma Catchment (KC) in western Tanzania highlights the pressures these ecosystems face. It sustains wildlife in the Katavi–Rukwa region and provides essential water for farming, livestock, and household needs. However, increasing water extraction, agricultural development, and upstream irrigation have disrupted flow patterns, decreased wetland size, and intensified conflicts between humans and wildlife, as well as among different user groups. While earlier studies have looked at hydrological trends and water-use conflicts in the area (Kashaigili *et al.*, 2015; Mwakalila, 2018), little attention has been given to how biodiversity priorities can be effectively integrated into IWRM. This study, regards the Katuma Catchment as a socio-ecological system whose sustainability depends on aligning wetland conservation with water governance. It assumes that weak enforcement of land-use regulations, limited institutional coordination, and unregulated water abstraction are causing ecological decline. Conversely, improved collaborative governance, ecological restoration, and greater community participation could enhance wetland resilience. This study aims to identify the main challenges affecting wetland biodiversity within current water management practices, examine hydrological and ecological trends, and explore

ways to incorporate biodiversity conservation into IWRM. Specifically, it addresses three key questions: (1) What ecological and governance issues hinder wetland conservation? (2) How have hydrological changes impacted wetland health and biodiversity? (3) What strategies can improve the integration of biodiversity into catchment-level water management? The underlying hypothesis is that participatory, coordinated, and adaptive water governance enhances biodiversity conservation outcomes by aligning ecological needs with human development priorities.

MATERIALS AND METHODS

Study Area Description

The study was conducted in the Sitalike and Nsaginya sub-catchments, situated within the Katuma Catchment of the Katavi–Rukwa ecosystem in western Tanzania (Fig. 1). These catchment covers Nsimbo District Council, Tanganyika District Council, and part of Mpanda Municipal Council, with its southern boundary stretching into Katavi National Park. Geographically, it is located between longitudes 30°40'E and 31°10'E and latitudes 6°10'S and 6°50'S. Elevation in the area varies from approximately 956 to 2,018 meters above sea level.

Two major rivers drain the catchment: the Katuma River, flowing from the northwestern highlands toward Lake Rukwa, and the Mpanda River, originating in the northern interior. These rivers nourish a network of wetlands that support diverse plant and animal life, including hippopotamuses, crocodiles, various fish species, and many migratory birds. The catchment also supplies vital water resources for local livelihoods, especially agriculture, livestock farming, and fishing. The region has a unimodal rainfall pattern, with rain falling from November to April and a clear dry season from May to October. This seasonal change significantly affects wetland hydrology, groundwater recharge, and overall biodiversity within the catchment.

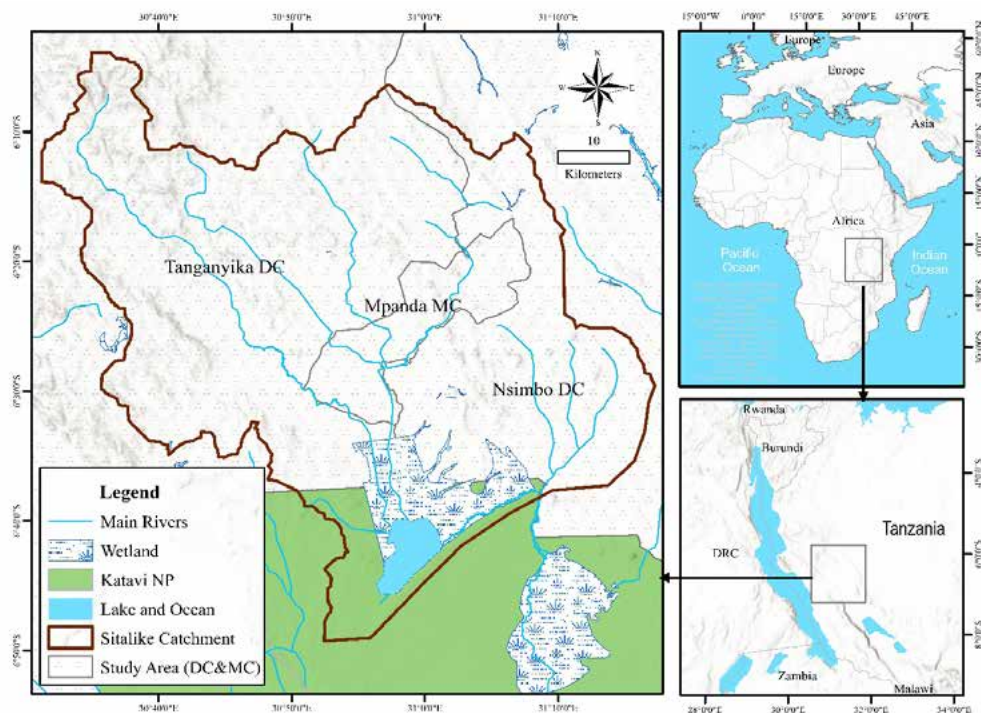


Figure 1: Study area location Map

Study Design Overview

This study employed a mixed-methods approach that combined biophysical, hydrological, and socio-institutional analyses to examine how biodiversity conservation relates to IWRM in the Katuma Catchment. Quantitative data such as rainfall, land cover, hydrology, human-induced factors (like livestock), and ecological observations were supplemented by qualitative insights from key informants and institutional documents. This triangulated method enabled a comprehensive understanding of ecological changes, human pressures, and governance dynamics impacting wetland management.

Data Collection Procedures

This study used both primary and secondary data sources to explore the ecological, hydrological, institutional, and community aspects of wetland management in the Katuma Catchment. Primary data were collected through Key Informant Interviews (KIIs) with representatives from TANAPA, the Lake Rukwa Basin Water Board (LRBWB), district authorities, NGOs,

Water User Associations (WUAs), and village leaders. These interviews examined governance structures, wetland-use practices, institutional cooperation, and community perceptions of environmental changes. Complementary ecological observations were conducted across key wetland sites within protected particularly Katavi National Park (3 assessment stations) and unprotected areas (3 assessment stations) to record vegetation structure, signs of ecological degradation, sedimentation patterns, and evidence of human and livestock disturbance. These direct observations provided essential ground-truthing for spatial analyses and helped interpret stakeholder narratives. Secondary data included CHIRPS rainfall datasets (2016–2026), which were utilized to analyze historical rainfall patterns and hydrological variability, as well as Sentinel-2 satellite imagery for detecting spatial changes in wetland extent and surrounding land cover. Additionally, institutional and policy documents relevant to IWRM and wetland governance were reviewed to understand the regulatory frameworks and management mechanisms shaping resource use. Stakeholder engagement was enhanced

through consultative meetings and validation workshops involving government agencies, NGOs, and community groups. These interactions provided site-specific insights, validated field findings, and contributed to local ecological knowledge, thereby enhancing the interpretation and contextual relevance of the results.

Data Processing and Analysis Procedures

Qualitative, quantitative, and spatial data were analyzed using standard procedures. KIIs were transcribed, coded, and thematically analyzed in NVivo 12 to assess governance, ecological conditions, human pressures, and community participation. Policy documents were reviewed to find gaps in IWRM and wetland governance. Hydrological data were processed in Excel and SPSS employing the Mann–Kendall and Sen’s slope methods to examine long-term trends in rainfall, flow, and groundwater, which were interpreted alongside ecological field observations. Spatial datasets were analyzed in ArcGIS 10.8 using supervised classification and accuracy assessment to create land cover maps for 2016 and 2024, and to evaluate changes in wetland, forest, agricultural, and settlement classes. Ecological data were summarized with descriptive statistics and the Shannon–Wiener Index. Combined hydrological, ecological, spatial, and qualitative evidence informed a conceptual model linking biodiversity conservation, water governance, and community participation, which was refined through stakeholder validation.

RESULTS AND DISCUSSION

Drivers Undermining Wetland Biodiversity Conservation under Existing Water Resource Management Regimes

Status and Hydrological Dynamics of Wetland Systems

Wetland ecosystems in the Katuma Catchment show clear signs of ecological stress, as consistently reported by key informants from LRBWB, TANAPA, district authorities, NGOs, and local communities. Respondents described observable reductions in wetland size, declining groundwater recharge, and rising sedimentation, patterns that closely match the hydrological trends derived from CHIRPS rainfall data for 2016–2024. Analyses of annual rainfall totals showed significant variation from year to year during the study period (Fig. 2). Although the Mann–Kendall test identified a statistically significant upward trend in annual rainfall ($Z = 1.98$, $p < 0.05$), Sen’s slope suggested a moderate increase of about +45.2 mm per year. Despite this long-term upward trend, the rainfall pattern has become more unpredictable, experiencing delayed starts, short seasons, and extended dry spells within the season. Years like 2016, 2017, 2018, and 2021 had notably lower totals, leading to shallow flooding, shorter wet periods, and fast drying of wetlands condition frequently mentioned in interviews.

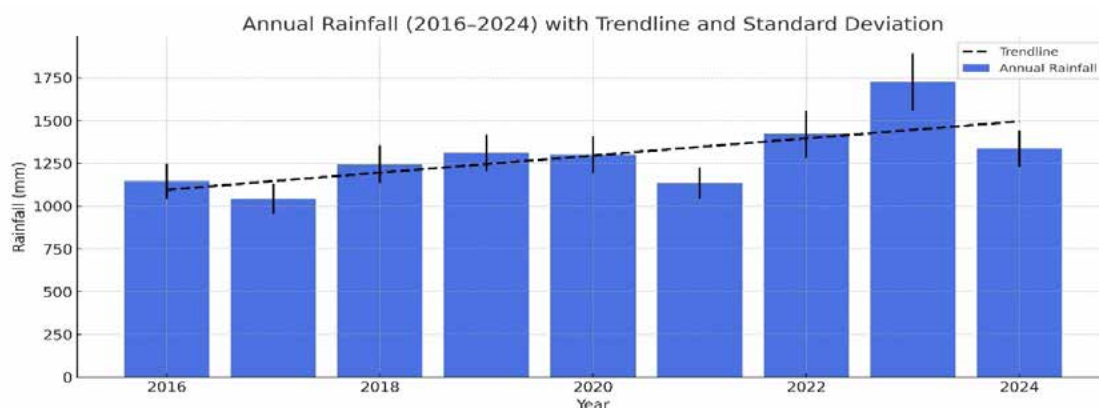


Figure 2: Annual rainfall trends in the Sitalike Catchment (2016–2024).



Figure 3. Rainfall anomalies relative to the long-term mean (2016–2024).

Further insight is provided by the rainfall anomaly analysis (Fig. 3), which shows strong negative anomalies in 2016, 2017, 2018, and 2021, followed by pronounced positive anomalies in 2022 and 2023. Stakeholders reported that negative-anomaly years were marked by early wetland desiccation, limited overbank flooding, and reduced groundwater recharge. Meanwhile, the brief hydrological recovery observed in 2022–2023 did not reverse long-term ecological degradation, as increased human pressures continued to limit wetland resilience.

These hydrological changes were verified by field-based ecological observations. According to one LRBWB hydrologist, *“Wetlands here are significantly smaller than they were a decade ago; the hydrological balance is clearly shifting.”* TANAPA ecologists similarly noted that wetlands within the protected area remain comparatively stable, whereas those outside park boundaries experience compounded stress from both altered rainfall patterns and expanding human activities.

Anthropogenic Pressures and Landscape Transformation

Land-Use and Land Cover Transitions across the Catchment

Human activities have become a major driver of wetland degradation in the Katuma Catchment, with land cover change emerging as one of the

clearest indicators of increasing anthropogenic pressure. Key informants consistently highlighted rapid agricultural expansion, encroachment into grassland and wetland areas, and the conversion of natural habitats as primary contributors to ecological decline. A representative from LRBWB remarked, *“People are expanding their farms into the wetlands, and the irrigation pumps upstream are reducing downstream flows much earlier.”* Village leaders echoed this observation, describing the widespread clearing of areas that were previously intact wetland systems. As the leader from Kabage stated, *“Wetlands used to be open and flowing; now farms are everywhere.”*

The LULC maps for the years 1984 - 2024 (Fig. 4) demonstrate a clear and progressive spatial pattern of land cover transformation within the catchment. The analysis of land use and land cover change (Table 1) shows substantial transformations across the landscape, reflecting increasing human pressure and progressive ecological degradation. The results demonstrate a marked decline in natural vegetation, particularly forest and woodland ecosystems, accompanied by rapid expansion of agricultural land and a significant reduction in wetland and water resources. Forest cover experienced the most drastic change, decreasing from 162,604 ha in 1984 to only 45,636 ha in 2024, representing a 71.95% loss over four decades. Woodland cover also declined considerably, with a reduction of more

than 160,000 ha, equivalent to a 37% decrease. These losses indicate persistent deforestation and conversion of natural vegetation, likely driven by agricultural expansion, fuelwood harvesting, charcoal production, and settlement growth. The decline in dense vegetation suggests increasing landscape fragmentation, which undermines ecological connectivity, reduces habitat availability for wildlife, and weakens the resilience of natural systems to climate stress. During the same period, Shrubland expanded substantially, increasing by more than 113,000 ha, which represents a 113% rise. This expansion is often associated with degradation of formerly dense vegetation into more open systems due to overgrazing, frequent fires, selective tree removal, and other anthropogenic pressures. The shift from woodland and forest to shrubland implies a downward transition in ecosystem quality, with reduced biomass, lower carbon storage, and diminished biodiversity. Shrub-dominated areas often reflect intermediate degradation stages and can be early indicators of long-term environmental decline if no interventions are implemented.

Wetland ecosystems recorded one of the most alarming reductions, declining from 97,474 ha in 1984 to just 22,650 ha in 2024—a 76.8% loss. Water bodies similarly decreased by 78%. These declines are likely the result of encroachment for crop cultivation, sedimentation caused by upland soil erosion, increased water abstraction, and prolonged dry periods linked to climate variability. The loss of wetlands has serious implications for hydrological regulation, groundwater recharge, biodiversity support, and dry-season flow maintenance. These ecosystems are critical for sustaining fisheries, offering flood mitigation, and supporting local livelihoods, and their degradation places communities at greater risk of water scarcity and reduced agricultural productivity.

In contrast, cultivated land expanded rapidly, increasing from 32,180 ha in 1984 to 226,650 ha in 2024, a growth of more than 600%. This trend reflects the growing demand for agricultural land driven by population increase and livelihood dependence on smallholder farming. The expansion of cultivation is strongly linked to the decline of forest, woodland, and wetland resources, indicating ongoing conversion of natural ecosystems into farmland. Built-up areas also expanded, increasing by more than 200%, although they remain relatively small in absolute terms. Their growth nonetheless marks permanent conversion of natural land into human settlements and infrastructure. Bare soil increased substantially, rising from 66 ha to 566 ha over the study period, which likely reflects increased soil erosion, overgrazing, abandoned fields, or recurring fires. The increase in bare soil signals declining land productivity and heightened land degradation risks, which may further contribute to sedimentation of rivers and wetlands. Overall, the LULC changes from 1984 to 2024 highlight a landscape undergoing rapid transformation from natural to human-modified systems. The combined effects of vegetation loss, wetland degradation, increased soil exposure, and agricultural encroachment reveal growing environmental pressures that threaten ecological integrity, water security, and long-term sustainability of natural resources. These changes have direct implications for biodiversity conservation, hydrological regulation, climate resilience, and community livelihoods. Understanding these patterns provides a critical foundation for planning restoration efforts, guiding sustainable land use, and informing policy interventions aimed at protecting and rehabilitating degraded ecosystems.

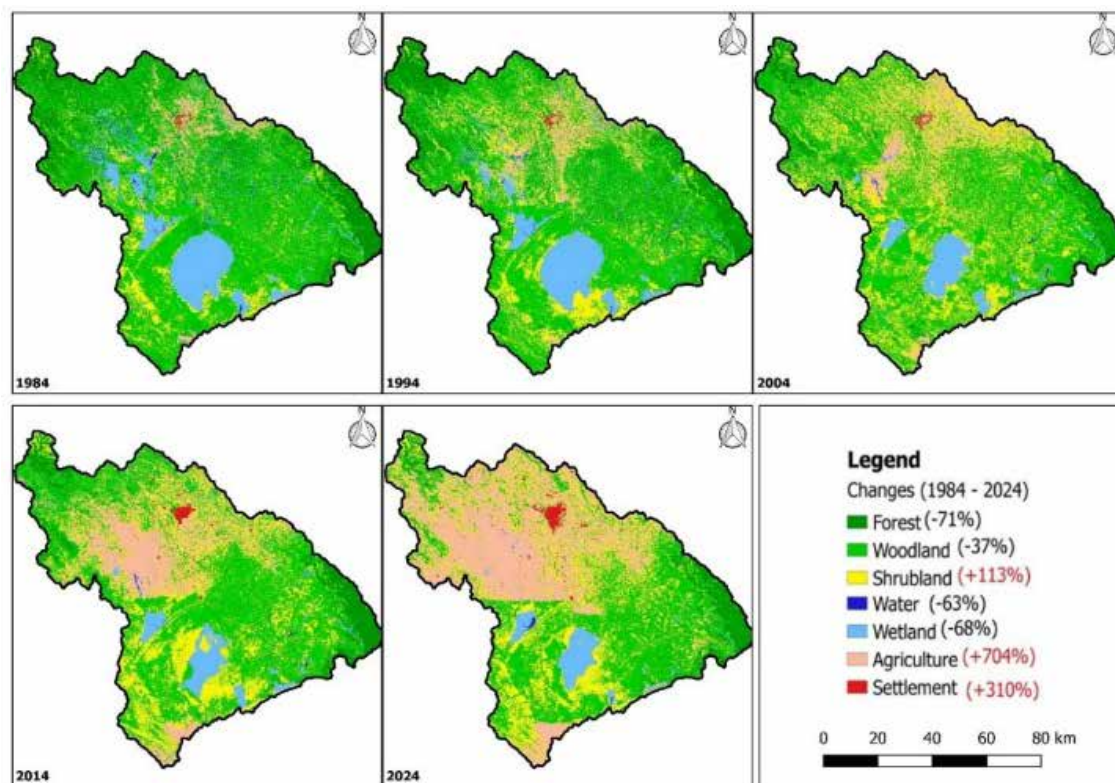


Figure 4: Spatial and quantitative illustration of land cover changes in the Katuma Catchment between 1984 and 2024.

Table 1. Summary of land cover changes in the Katuma Catchment between 1984 and 2024.

	1984 – 1994			1994 – 2004			2004 – 2014					2014 – 2024				
LULC	Area change (Ha)	Percentage change (%)	Annual Rate of Change (Ha/year)	Area change (Ha)	Percentage change (%)	Annual Rate of Change (Ha/year)	Area change (Ha)	Percentage change (%)	Annual Rate of Change (Ha/year)	Area change (Ha)	Percentage change (%)	Annual Rate of Change (Ha/year)	Area change (Ha)	Percentage change (%)	Annual Rate of Change (Ha/year)	
Forest	-28834	-18	-2883	-69531	-51.98	-6953	-1934	-3.01	-193	-16668	-26.75	-1667	116968	-72	-3899	
Woodland	-22224	-5	-2222	-5635	-1.37	-563	-4478	-8.52	-3448	-97943	-26.46	-9794	160280	-37	-5343	
Shrubland	44476	45	4448	97485	67.51	9749	-7830	-11.51	-2783	-764	-0.36	-76	113367	113	3779	
Water	-650	-25	-65	-679	-35.04	-68	-173	-13.71	-17	-135	-12.46	-14	-1637	-63	-55	
Wetland	-11408	-12	-1141	-42242	-49.08	-4224	-6194	-14.13	-619	-7261	-19.3	-726	-67104	-69	-2237	
Cultivated land	18735	58	1874	20373	40.01	2037	67246	94.33	6725	1E+05	86.83	12030	226651	704	7555	
Built up area	-69	-4	-7	194	11.6	19	3058	163.79	306	2223	45.14	222	5406	310	180	
Bare soil	-27	-41	-3	35	88.71	3	304	412.94	30	254	67.05	25	566	861	19	

Intensification of Livestock Grazing Pressure

Livestock grazing emerged as another major human impact contributing to wetland degradation in the Katuma Catchment. Key informants consistently emphasized the intensity of livestock movement across wetland areas. As one village leader in Katuma observed, *“Every day, herds of cattle walk straight through the wetlands. They break the riverbanks and stir mud into the water.”* Such accounts directly align with the livestock data for the Sitalike Catchment, which show a clear and sustained increase in herd numbers over recent years.

District-level records show that livestock populations have increased significantly from 2012–2014 to 2020–2024 (Fig. 5). Tanganyika

District Council, which holds the largest share of the basin’s herds, grew from 142,856 animals in 2012–2014 to 392,000 in 2020–2024, an almost 174% increase. Similar upward trends are seen in Nsimbo District Council, where livestock numbers rose from 26,373 to 83,827, while Mpanda Municipal Council modestly increased from 10,989 to 22,000 during the same period.

These ongoing increases indicate rising grazing pressure throughout the catchment. Larger herd sizes and more frequent seasonal movements lead to continued trampling of riparian zones, damage to wetland soils, and faster erosion and sediment buildup. Field observations showed eroded riverbanks, wider livestock trails, and higher turbidity in shallow wetland areas—conditions directly connected to unmanaged livestock access.

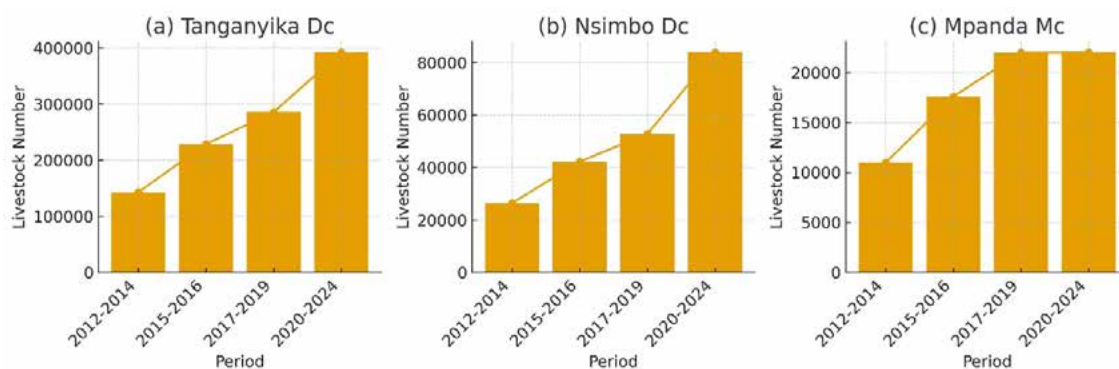


Figure 5. Livestock population trends in the Katuma Catchment (2012–2024).

The livestock trends strongly reinforce stakeholder concerns. As the number of animals grows, pressure on wetlands intensifies, especially during the dry season when herds concentrate around the few remaining water points. The convergence of expanding herd sizes, unregulated movement, and limited grazing alternatives collectively positions livestock encroachment as a significant and escalating threat to the integrity of wetlands within the Katuma Catchment.

Ecological and Socioeconomic Consequences of Wetland Degradation

Wetland degradation has far-reaching implications for biodiversity. TANAPA ecologists reported reduced feeding and breeding sites for large mammals: *“Hippos and elephants now walk much longer distances for water, and some die before reaching reliable sources.”* Fish breeding zones are also declining due to siltation. An IUCN officer noted, *“Species that depend on shallow wetlands are*

the first to disappear. The fish breeding zones are filling with mud.” Livelihoods are similarly affected. Farmers reported reduced yields due to declining soil moisture. A district natural resource officer observed, *“Harvests are failing because the land no longer holds water the way it used to.”* Fishing communities also experience reduced catches: *“We used to catch fish even in dry months. Now some wetlands are empty.”* Diminishing resources intensify farmer–pastoralist conflicts. A WUA member stated, *“Conflicts arise because everyone is using the same shrinking spaces.”*

Institutional Arrangements, Cross-Sector Coordination, and Governance Gaps

Respondents acknowledged that institutions possess clearly defined mandates, yet coordination remains weak and inconsistent. LRBWB oversees water regulation, TANAPA manages wildlife and wetland ecosystems, district councils enforce bylaws, NGOs provide technical support, and village governments handle local enforcement. Yet these roles often overlap, causing institutional friction. A LRBWB officer explained, *“We collaborate, but the coordination is ad hoc and mostly dependent on donor projects.”* TANAPA officer noted that coordination weakens outside protected areas: *“Inside the park, coordination is strong... once we step outside, it becomes difficult because responsibilities are not shared in a structured way.”* NGOs expressed concerns over limited human and financial capacity. SNV officer stated, *“Everyone is doing something, but without proper integration, we are losing impact.”* Weak data-sharing mechanisms and inconsistent stakeholder meeting attendance further undermine coordinated action.

Governance Framework, Policy Coherence, and Enforcement Limitation

Respondents acknowledged the robustness of Tanzania’s environmental policy framework, which includes the Water Resources Management Act, the Environmental Management Act, and the Protected Areas Act. However, enforcement challenges persist.

A district environmental officer emphasized, *“The policies exist on paper, but implementing them is the real challenge due to staff shortages and limited resources.”* Policy inconsistencies were reported, particularly where agricultural expansion conflicts with wetland protection goals. WASIMA officer commented, *“Agricultural policies encourage expansion, but this directly contradicts wetland protection measures.”* Village leaders highlighted that land allocations often disregard environmental guidelines: *“Sometimes land is given out without considering ecological impacts.”* Enforcement of local bylaws varies widely depending on the strength of leadership and community commitment. The Village Chairperson noted, *“We have bylaws, but enforcing them is difficult when people depend on wetlands for survival.”*

Community Engagement, Local Knowledge Systems, and Participation Barriers

Community participation in wetland conservation is limited, although it exists. Participation channels include Water User Associations (WUAs), village environmental committees, and NGO-supported initiatives. However, dependency on wetlands constrains full participation: *“People want to conserve wetlands, but at the same time they depend on them every day for farming and water,”* explained by WUA member. Key barriers include poverty, low awareness, insufficient incentives, and limited technical skills. IUCN officer stated, *“Communities need alternatives before they can reduce pressure on wetlands.”* Cultural norms provide some protection in areas with sacred springs or taboo sites, which communities traditionally avoid disturbing. A village elder reflected, *“We do not touch the sacred springs. Our ancestors protected them, and we continue.”* Despite these constraints, respondents agreed that communities are willing to participate if provided with adequate support and incentives.

Opportunities, Lessons Learned, and Strategic Pathways for Strengthening Wetland Governance

Respondents identified several promising opportunities that can enhance the integration of biodiversity conservation within the IWRM framework in the Katuma Catchment. These opportunities emerge from ongoing conservation efforts, increasing community awareness, and the presence of established institutional mechanisms. Together, they demonstrate the practical pathways through which coordinated and inclusive management approaches can reconcile water resource use with ecological integrity.

Strengthening Institutional Mechanisms for Integrated Water and Biodiversity Management

The strong presence of conservation and development partners, including GIZ, IUCN, SNV, Rikolto, and WASIMA, was repeatedly cited as a key opportunity. These organizations play central roles in facilitating multi-stakeholder collaboration, providing technical expertise, and implementing targeted conservation projects. Basin planning structures and active Water User Associations (WUAs) offer additional institutional entry points for embedding biodiversity priorities into catchment-wide water management plans. These structures provide platforms for participatory decision-making and create a governance environment conducive to integrated resource management.

Growing Community Awareness and Engagement

An encouraging finding was the increasingly widespread recognition among community members of the ongoing decline of wetland ecosystems. As one NGO officer noted, *“People now understand that wetlands are disappearing. This awareness is a major opportunity for change.”* Rising awareness serves as a foundation for behavioral change,

community participation, and local stewardship, elements that are essential for long-term sustainability. Community-driven initiatives, such as participatory water monitoring, local wetland patrols, and awareness campaigns, have already demonstrated positive outcomes.

Successful Local Restoration and Conservation Interventions

Several field-based interventions were highlighted as successful models for strengthening biodiversity conservation within IWRM. These include riverbank restoration, establishment of community tree nurseries, ecosystem rehabilitation activities, and protection of ecologically sensitive sites. Such ecosystem-based restoration efforts not only enhance wetland resilience but also act as practical demonstration sites for integrating hydrological restoration with biodiversity recovery.

Lessons on Co-Management, Community Ownership, and Multi-Stakeholder Collaboration

A dominant lesson was that conservation initiatives are more successful when local communities feel a sense of ownership. As respondents emphasized, *“Projects succeed when communities feel responsible.”* This reinforces the importance of co-management approaches in which communities participate meaningfully in planning, implementation, and monitoring activities. Additionally, collaboration among government agencies, NGOs, basin authorities, and local leaders enhances collective problem solving and reduces institutional fragmentation.

Priority Actions for Integrated Wetland and Water Governance

Based on the opportunities and lessons identified, respondents proposed several priority actions to strengthen the linkages between IWRM and biodiversity conservation. These include strengthening the enforcement

of environmental laws and village bylaws to safeguard riparian zones and regulate water abstraction, as well as improving ecological and hydrological monitoring systems, particularly through community-led participatory approaches, to enhance adaptive management. Respondents also emphasized the need to harmonize sectoral policies, especially those governing agriculture, water, and conservation, to address contradictions that currently undermine wetland protection. Scaling up restoration initiatives such as wetland rehabilitation, riverbank stabilization, and reforestation of degraded catchments was highlighted as essential for restoring ecological functionality. Additionally, promoting climate-resilient livelihood alternatives, including sustainable agriculture, beekeeping, and eco-tourism, was identified as crucial for reducing direct community dependence on wetlands and enhancing long-term conservation outcomes.

Integrated Framework for Biodiversity Conservation within Adaptive IWRM Systems

The proposed integrated framework builds on identified opportunities, lessons learned, and priority actions to create a coordinated model for enhancing biodiversity conservation within the IWRM system of the Katuma Catchment. At its core, the framework places adaptive water governance as the main mechanism connecting wetland ecological processes, institutional collaboration, and active community involvement. Strengthening the enforcement of environmental laws and village bylaws forms the foundation of the governance structure, ensuring that riparian zones, water abstraction points, and key wetland habitats are effectively protected. Additionally, the framework emphasizes the importance of enhancing ecological and hydrological monitoring systems, particularly through participatory community-based monitoring, to establish real-time feedback loops that inform decision-making and support adaptive management.

Institutional harmonization is another critical pillar of the framework. Aligning agricultural,

water, and conservation sector policies reduces contradictions that currently undermine wetland protection, enabling coordinated planning and integrated responses across institutions such as LRBWB, TANAPA, district councils, WUAs, and NGOs. Ecological restoration forms the ecological pillar of the framework, with interventions such as wetland rehabilitation, riverbank stabilization, and catchment reforestation that strengthen ecosystem resilience and ensure sustained hydrological functioning. Finally, the framework incorporates climate-resilient livelihood diversification, such as sustainable agriculture, beekeeping, and eco-tourism, to reduce direct human pressure on wetlands and enhance community support for conservation initiatives.

Overall, the framework functions through three interconnected pathways: (i) strengthening governance and enhancing policy coherence, (ii) restoring ecosystems and conducting ecological monitoring, and (iii) empowering communities and promoting sustainable livelihoods. By linking these pathways through adaptive management, the proposed framework offers a practical and scalable model for integrating biodiversity conservation into IWRM and promoting long-term ecological and social resilience in the Katuma Catchment.

Summary of Key Findings and Policy Implications for Wetland Management

This study provides clear evidence that wetland ecosystems in the Katuma Catchment are experiencing significant ecological decline, caused by a combination of hydrological, land-use, and governance issues. Hydrological analyses show increasingly unpredictable rainfall patterns, including several years of strong negative rainfall anomalies, leading to decreased groundwater recharge, shorter hydro-periods, and reduced dry-season flows. These hydrological trends align with satellite data indicating a reduction in wetland size and widespread conversion of grasslands into farmland. Field observations also show speeding sediment buildup, vegetation loss,

and fragmentation of key wetland habitats used by hippos, elephants, fish, and other species that depend on wetlands.

These ecological disruptions directly undermine community livelihoods. Farmers face declining yields as soil moisture diminishes, fishers report reduced spawning grounds and lower catches, and pastoralists experience increasing competition over limited water points. The rapid expansion of livestock populations across multiple districts, coupled with agricultural encroachment into riparian zones, intensifies pressure on already stressed wetland ecosystems. These combined pressures heighten farmer–herder conflicts and reveal the socio-ecological vulnerabilities of the catchment.

Despite these challenges, several opportunities emerge from the findings. Community awareness of wetland degradation is increasing, and basin-level institutions, including WUAs and district environmental committees, provide platforms for coordinated action. Ongoing NGO-led restoration efforts, such as riverbank rehabilitation and community tree nurseries, demonstrate the feasibility of ecosystem-based interventions. Insights from stakeholders underscore the importance of inclusive governance structures, harmonized cross-sectoral policies, and long-term ecological and hydrological monitoring.

Together, these findings inform the development of an integrated framework that links adaptive water governance, ecosystem restoration, and community participation. This framework offers a practical and transferable model for strengthening wetland management in the Katuma Catchment and other socio-ecological landscapes confronting similar pressures.

Policy Implications

The results of this study generate several policy and management recommendations critical for enhancing the resilience of wetlands in the Katuma Catchment:

Strengthened Environmental Enforcement

Effective wetland protection requires stronger enforcement of existing regulations, particularly those governing riparian buffer integrity, water abstraction, and wetland encroachment. Clarifying institutional mandates and enhancing enforcement capacity at district and village levels will improve compliance.

Harmonized Sectoral Policies and Institutional Coordination

Agriculture, water, land-use, and conservation policies must be aligned to minimize contradictions. Establishing formal coordination mechanisms, such as joint planning committees, basin-level data-sharing systems, and synchronized monitoring programs, will reduce duplication and enhance cooperation among LRBWB, TANAPA, district councils, NGOs, and WUAs.

Investment in Long-Term Hydrological and Ecological Monitoring

Adaptive management depends on consistent, high-quality data. Strengthening hydrological stations, groundwater monitoring, water-quality assessments, and biodiversity surveys supported by participatory community monitoring will enable early detection of ecological change and evidence-based decision-making.

Scaling Up Ecosystem Restoration and Nature-Based Solutions

Wetland rehabilitation, catchment reforestation, riverbank stabilization, and invasive species control should be integrated into district and basin planning frameworks. Long-term technical and financial support is essential to sustain restoration efforts and improve ecological functionality.

Livelihood Diversification and Community Empowerment

Reducing pressure on wetland ecosystems requires expanding climate-resilient and low-impact livelihoods. Policies supporting sustainable agriculture, beekeeping, eco-tourism, aquaculture, and nature-based enterprises can improve household incomes while reducing dependence on wetland resources.

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SUB-THEME: INFRASTRUCTURE DEVELOPMENT AND WILDLIFE CONSERVATION

STATUS OF WILDLIFE ROADKILL IN THE NGORONGORO CONSERVATION AREA, NORTHERN, TANZANIA

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ABSTRACT

Globally, roads traversing protected areas facilitate the transportation of goods and services. Still, they usually create some negative impacts, such as wildlife roadkill, which has occurred within the Ngorongoro Conservation Area (NCA). An 82 km stretch of gravel main road from Lodoare to Golini was used as the study transect to survey wildlife roadkill in NCA from October 2021 to October 2022 for four consecutive days, twice a day per month for ten months during the wet and dry seasons, to document wildlife roadkill species and their spatial-temporal patterns in the area. The surveys started early in the morning, from 0700 to 1800 hours and employed direct and opportunistic encounter observation methods for data collection. A vehicle with a speed limit of 20km/hr was used during the survey. The results revealed that 85 individual wild animals with encounter rates of 0.016 animals/kilometer belonging to 26 species, including 10 mammalian, 5 reptilian, and 11 bird species, were killed in the area. More birds (69.4%) than mammals (18.8%) and reptiles (11.8%) were found killed in the area. Additionally, the nightjar (*Caprimulgus europaeus*; 30.6%), followed by black rat (*Rattus rattus*; 7.1%) and chameleon (*Chamaeleo chamaeleon*; 3.5%), were the most frequently recorded killed species in the area.

Furthermore, wildlife roadkills did not differ significantly between seasons ($p = 0.371$) and time of day (e.g., morning vs. afternoon; $p = 0.652$) but differed significantly between their body size (e.g., small, medium, and large; $p < 0.001$) and habitat types (e.g. grassland, woodland, shrubland, wooded grassland, forest; $p = 0.005$). The study recommends punishments and penalties for overspeeding drivers and installing cameras, speed limits, and signboards along the highway to alert drivers to reduce speed. Also, providing regular education to road users on the impacts of roadkill within the area is critical.

Keywords: conservation, habitats, roads, vehicles, wildlife

INTRODUCTION

Worldwide, the construction of new roads increases access to native environments, resulting to a significant landscape changes. Evidence shows approximately 85% of

deforestation occurs within fifty kilometers of the road boundary highlighting the strong link between (Santos *et al.*, 2022). These roads contribute to the mortality of native wildlife through vehicle collisions (e.g. roadkill incidents) across protected areas

worldwide (Pinto *et al.*, 2020b; Schwartz *et al.*, 2020; Lyamuya *et al.*, 2021; Santos *et al.*, 2022). Over the next decade, those roads would increase in number, size, and traffic (Schwartz *et al.*, 2020; Lyamuya *et al.*, 2021), and some might be upgraded from unpaved to paved roads shortly, particularly those on the main road of the NCA. According to Santos *et al.* (2022), Brazil has over 1.72 million kilometers of highways, as reported by the National Transport Confederation. According to Santos *et al.* (2022), Brazil has over 1.72 million kilometers of highways, as reported by the National Transport Confederation (CNT), which has directly destroyed more than 595,000 hectares of vegetation previously occupied by a diverse range of organisms. It was estimated that 25 million kilometers of new roads will be built worldwide by 2050, an increase of 60% in the total length of roads compared to 2010, with 90% of them to be built in developing countries such as Brazil (Santos *et al.*, 2022). Again, the study by Grilo *et al.* (2021)(b reported that globally, there are about 36 million km of road networks, and it was estimated that an additional 25 million km of roads will be needed by 2050 (Alamgir *et al.*, 2019; Grilo *et al.*, 2021). The expansion of road networks is also expected in developing countries containing exceptional biological diversity and the majority of hotspot areas (Ng *et al.*, 2020). The previous study also reported that the impacts on wildlife are manifold, including changes in animal behavior and habitat use, modified composition, structure, and dynamics of communities, and increased mortality rate (Behera *et al.*, 2021; Santos *et al.*, 2022). Due to forecasted impacts on biodiversity due to the expansion of road networks worldwide, evaluating their current and future impacts is critical for improving current management practices and biodiversity conservation (Collinson *et al.*, 2019; Lyamuya *et al.*, 2021). Wildlife roadkill is one of the most visible and studied impacts of roads and is considered the main cause of vertebrate death by human activity (Santos *et al.*, 2022). To determine the mortality of wildlife populations caused by roadkill, counts have been used

across seasons and years in protected areas (Lyamuya *et al.*, 2022). The study by Lyamuya *et al.* (2022) reported that several studies on roadkill in Tanzania have reported on the occurrence of dead animals on the roads in the Serengeti ecosystem; however, scant information is available for the Ngorongoro Conservation Area (NCA). Therefore, this study aimed to bridge this gap by determining the status and patterns of wildlife roadkill in the NCA. It had been explained that the body size and diet preference have been reported as important variables to explore patterns in interspecies variation in roadkill frequency and how it affects the probability of roadkill (Ford and Fahrig, 2007; Lyamuya *et al.*, 2022). The present study, therefore, aimed at determining the status of wildlife roadkill and their spatial-temporal patterns in the area. The study firstly hypothesized that small-bodied animals would be more hit by vehicles compared to medium to large animals in the area. Secondly, the frequency of wildlife roadkill would occur differently between habitat types within the NCA. The information accrued from this study would help in minimizing the impacts of wildlife roadkills and increase the safety of people and their properties within the NCA. Further, it will create awareness and issues to consider for different stakeholders in designing and expanding road networks, especially within protected areas.

MATERIALS AND METHODS

Study Area

The study was conducted in the Ngorongoro Conservation Area (NCA), which covers 8,292km² (URT, 2016; Fig. 1). NCA is in the North of Tanzania, 90 kilometres West of Arusha, adjoining the South-eastern edge of Serengeti National Park between 2°30' to 3°30'S and 34°50' to 35°55'E (Masao *et al.*, 2015). NCA was inscribed on the World Heritage List in 1979, hence a UNESCO World Heritage site. The main feature of the NCA is the volcanic Ngorongoro Crater, the largest unbroken caldera in the world, home to the

densest wildlife population of some 25,000 large animals, mainly ungulates, alongside the highest density of diverse mammalian predators in Africa. The area supports the Greater wildebeest migration from the Serengeti - Maasai Mara Ecosystem (Moehlman *et al.*, 2020). NCA is home to critically endangered Black Rhino (*Diceros bicornis*) and wild dog (*Lycaon pictus*) populations. The main wildlife species found include elephants (*Loxodonta africana*), wildebeests (*Connochaetes taurinus*), impala (*Aepyceros melampus*), eland (*Taurotragus oryx*), zebra (*Equus quagga*), black rhinoceros (*Diceros bicornis*) and hippopotamus (*Hippopotamus amphibius*), leopards (*Panthera pardus*), lions (*Panthera leo*), cheetahs (*Acinonyx jubatus*), spotted hyenas (*Crocuta crocuta*) and a variety of bird species (Mahulu *et al.*, 2015; Moehlman *et al.*, 2020; NCAA, 2020). This outstanding concentration of wildlife species constitutes one of the natural wonders of the world, attracting tourists from different parts of the world. The area also covers the Empakaai and Olmoti Crater, Laetoli, and Olduvai Gorge, famous for their geological attractions and associated first human beings to walk on earth (Masao *et al.*, 2015). The variations in climatic conditions, different landforms, and altitude have resulted in the occurrence of several ecosystems and diverse habitats, with

short grass plains acting as calving grounds of wildebeest, water-catching highland forests, scrub heath, montane long grasslands, and high open moorlands with evergreen trees and grasses. The vegetation of the area varies from Montane Forest, grasslands, Acacia woodlands, and thorn scrub (URT, 2019). The pastoralist Maasai inhabited the NCA before the protected areas were created. The Maasai were allowed to live with wildlife after the NCA was formed in 1959, as a multiple land use. The NCA is the home of biodiversity (flora and fauna), where forest cover plays a great role in maintaining the existence of wildlife. The NCA is located in the highlands with moist and misty where temperatures in the semi-arid plains can fall as low as 2°C, and often rise to 35°C (Masao *et al.*, 2015). The annual precipitation falls between November and April and varies from 500 mm on the arid plains in the West to 1,700 mm on the forested slopes in the East (Masao *et al.*, 2015). The area has two wet seasons (the short rains and the long rains) with an average annual rainfall of 600-900mm (Bartzke *et al.*, 2018; URT, 2019) and average annual temperatures of 9°C-22°C in the wet season, and 18°C-31°C in the dry season (URT, 2019). The main road bisects the NCA and runs from Lodoare to Golini gate, covering 82 km and connecting northern Tanzania and Lake Zone regions (URT, 2019).

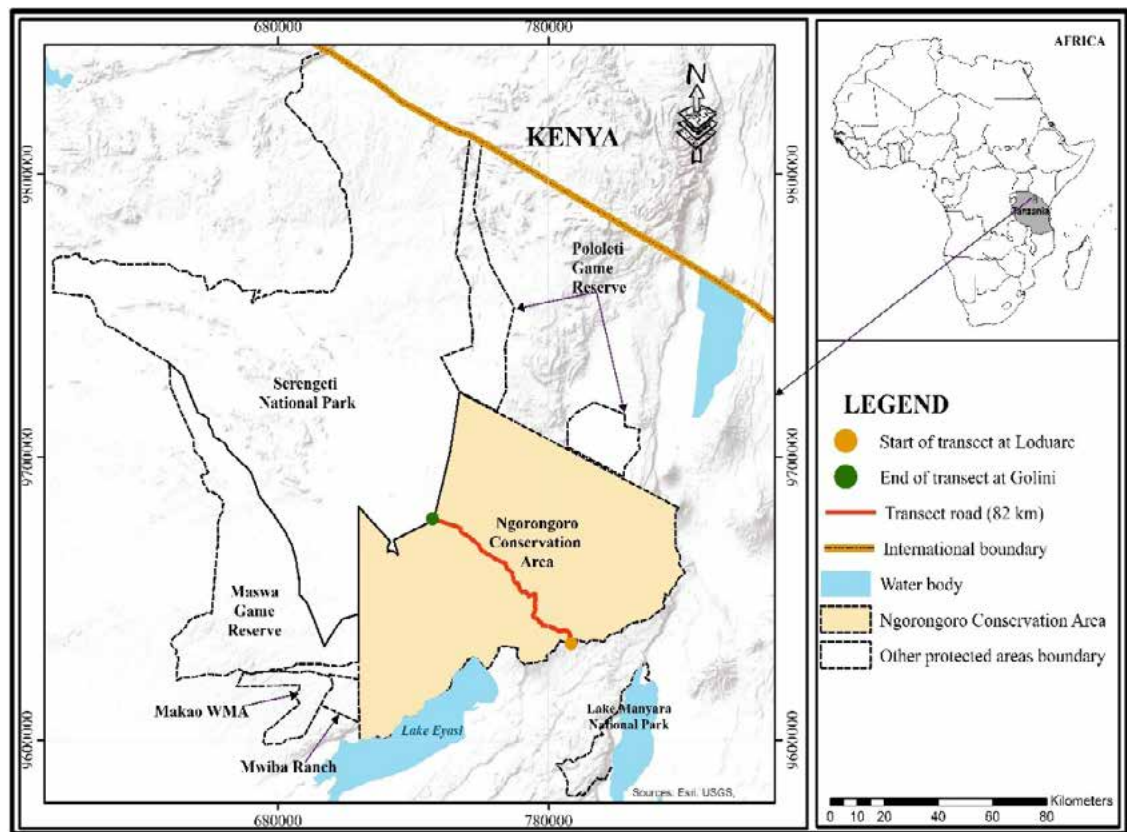


Figure 1: A map of the Ngorongoro Conservation Area displays the Lodoare-Golini Road Transect

Data collection

The collection of the wildlife roadkill data started early in the morning from 0700 to 1800 hours for eight consecutive days, twice a day per month during the wet and dry seasons from October 2021 to October 2022, to document wildlife roadkill. Before data collection, standardized procedures for data recording were conducted to ensure data collection consistency. Roadkill assessment was done in an 82 km stretch of unpaved main road that connects NCA and Serengeti National Park (SENAPA) from Lodoare gate - Golini for transportation services to the area (Hopcraft *et al.*, 2015; URT, 2019). Two sampling periods were conducted daily: the morning period (07h30–11h30) as the forward direction and the afternoon (14h00–18h00) as the backward direction along the same road, which acted as a transect. The surveys were performed along one transect, covering 82 km (Figure 1).

The surveyed transect was divided into 1 km segments ($n = 82$ segments) for sampling, and in total, there were 82 sampling segments along the 82 km transect. Each transect/road was driven twice a day (morning and afternoon) and therefore involved total segment sample sizes of 164 km for the study transect per day. In addition, the transect/road was driven four times per month, times 10 months (i.e., covering both wet and dry seasons), which resulted in a total effort of 6560 km of driven transects. The period also covered both the low and high tourism seasons to capture variations in the wildlife roadkills in both seasons, as the number of vehicles entering the NCA varies according to seasons (Kioko *et al.*, 2015b; Williams *et al.*, 2019). Moreover, a vehicle was used to conduct road surveys from Lodoare to Golini gate (transect) with a speed limit of 20 kilometres per hour for recording roadkill data. Furthermore, whenever opportunistic encounters of the wildlife roadkill outside the

survey period but within the transect were observed, they were also recorded (Kioko *et al.*, 2015a; Njovu *et al.*, 2019). The four observers in the car (two on the right, two on the left) conducted observations on either side of the road to locate any carcasses. At points where a carcass was sighted, the vehicle stopped. Each species killed was identified, number of dead individuals, the location (GPS coordinates) where the roadkill was found, the date (i.e. start and end dates), and time of the day (i.e. morning, afternoon, evening) when it was found, habitat type (i.e. grassland, woodland, wooded grassland, shrubland, forest) of the area, were recorded on a standard data sheet. After all of the required data had been collected, the carcasses were removed from the road to avoid double-counting.

Data analysis

The Statistical Package for Social Science (SPSS, version 16.0) and Palaeontological Statistics (PAST 4.03) program (Hammer *et al.*, 2001) software were used to analyze the data. The collected data were organized and cleaned using a Microsoft Excel spreadsheet, and a Shapiro-Wilk test was employed for testing normality. Since the results indicated that the data were not normally distributed, a nonparametric test was used to analyze the data. Differences in wildlife roadkills between the number of species killed, the IUCN status, seasons, and habitat types were tested using the Chi-square test. The differences between the wildlife roadkills between the morning (07:00 am to 11:59 am) and afternoon (12:00 pm to 06:00 pm) hours were assessed using a Mann-Whitney U test, and between their body sizes (i.e. small = < 30kg; medium = 31-50kg; large = > 50kg) were assessed using the Kruskal-Wallis test. Statistical significance was defined at $p < 0.05$.

RESULTS

Status of wildlife roadkill in NCA

The study results revealed that 85 individual animals (28 and 57 in the dry and wet seasons, respectively) belonging to 21 families, 5 mammalian, 3 reptilian, and 10 bird orders, were recorded as killed during 10 field trips in the area. These animals comprised 26 species, which included 10 mammalian, 5 reptilian, and 11 bird species, at an overall rate of 0.065 roadkill km per month. Moreover, more birds ($n = 59$; 69.4%) than mammals ($n = 16$; 18.8%) and reptiles ($n = 10$; 11.8%) were found killed in the area. Hence, birds were the most affected group by roadkill in the area. Amongst the bird species, the night jays (*Caprimulgus europaeus*; $n = 26$, 30.6%) were most frequently recorded killed, followed by black rat (*Rattus rattus*; $n = 6$, 7.1%) on the mammals' side and chameleon (*Chamaeleo chamaeleon*; $n = 3$, 3.5%) on the reptiles' side (Table 1). An average of 164 kilometres was surveyed per day, which resulted in a total effort of 6560 km of driven transects per year. In addition, the number of wildlife roadkill did not differ significantly in the area ($X^2 = 14.408$, $df = 26$, $p\text{-value} = 0.9672$). Moreover, regarding conservation status, the wildlife roadkill differed significantly based on their IUCN status ($X^2 = 3.771$, $df = 125$, $p < 0.001$), as one of them was found only endangered (Secretary bird, 1.2%) and the majority were least concerned (90.2%). While the remaining belonged to near-threatened (8.6%) IUCN status. Least concern species are plentiful in the wild, their chances of being affected by roadkill incidences is higher.

Table 1. Wild animal Species found killed along the Lodoare-Golini Road transect.

S/N	Common name	Scientific name	Class	Order	Family	No.killed	%
1	African buffalo	<i>Syncerus caffer</i>	Mammalia	Artiodactyla	Bovidae	1	1.2
2	Bat-eared fox	<i>Otocyon megalotis</i>	Mammalia	Carnivora	Canidae	1	1.2
3	Black rat	<i>Rattus rattus</i>	Mammalia	Rodentia	Muridae	6	7.1
4	Bushbuck	<i>Tragelaphus scriptus</i>	Mammalia	Artiodactyla	Bovidae	1	1.2
5	Dik-dik	<i>Madoqua kirkii</i>	Mammalia	Artiodactyla	Bovidae	1	1.2
6	Common Duiker	<i>Sylvicapra grimmia</i>	Mammalia	Artiodactyla	Bovidae	1	1.2
7	Olive baboon	<i>Papio anubis</i>	Mammalia	Primates	Cercopithecidae	1	1.2
8	Thomson's gazelle	<i>Eudorcas thomsonii</i>	Mammalia	Artiodactyla	Bovidae	2	2.4
9	Wildcat	<i>Felis silvestris</i>	Mammalia	Carnivora	Felidae	1	1.2
10	Zebra	<i>Equus quagga</i>	Mammalia	Perissodactyla	Equidae	1	1.2
11	Agama lizard	<i>Agama agama</i>	Reptilia	Squamata	Agamidae	2	2.4
12	Black mamba	<i>Dendroaspis polylepis</i>	Reptilia	Squamata	Elapidae	2	2.4
13	Chameleon	<i>Chamaeleo chamaeleon</i>	Reptilia	Squamata	Chamaeleonidae	3	3.5
14	Leopard tortoise	<i>Stigmochelys pardalis</i>	Reptilia	Testudines	Testudinidae	2	2.4
15	Puff adder	<i>Bitis arietans</i>	Reptilia	Squamata	Viperidae	1	1.2
16	Ash starling	<i>Lamprotornis unicolor</i>	Aves	Passeriformes	Sturnidae	1	1.2
17	Bee-eater	<i>Merops apiaster</i>	Aves	Coraciiformes	Meropidae	1	1.2
18	Francolin	<i>Francolinus vulgaris</i>	Aves	Galliformes	Phasianinae	7	8.2
19	Helmeted guineafowl	<i>Numida meleagris</i>	Aves	Galliformes	Numididae	9	10.6
20	Kori bustard	<i>Ardeotis kori</i>	Aves	Otidiformes	Otididae	2	2.4
21	Night jar	<i>Caprimulgus europaeus</i>	Aves	Caprimulgiformes	Caprimulgidae	26	30.6
22	Eagle Owl	<i>Bubo bubo</i>	Aves	Strigiformes	Strigidae	5	5.9
23	Pied crow	<i>Corvus albus</i>	Aves	Passeriformes	Corvidae	1	1.2
24	Secretary bird	<i>Sagittarius serpentarius</i>	Aves	Accipitriformes	Sagittariidae	1	1.2
25	Superb starling	<i>Lamprotornis superbus</i>	Aves	Passeriformes	Sturnidae	3	3.5
26	Weaver	<i>Ploceus cuculatus</i>	Aves	Passeriformes	Ploceidae	3	3.5
Total						85	100

Patterns of wildlife roadkill in NCA

In most of the wildlife, roadkill incidences occurred in the wet ($n = 57$, 67.1%) compared to the dry ($n = 28$, 32.9%) season, but their differences were not statistically significant ($\chi^2 = 29.831$, $df = 28$, $p = 0.371$). Despite that, in total, birds were mostly frequently recorded killed in both wet and dry seasons ($n = 45$, $n = 12$) compared to mammals ($n = 7$, $n = 9$) and reptiles ($n = 5$, $n = 7$), respectively, in the area. Moreover, the occurrences of wildlife roadkill differed significantly between the habitat types in the area ($X^2 = 204.73$, $df = 156$, $p = 0.005$) with more roadkill recorded in the woodland ($n = 35$, 41.2%), followed by grassland ($n = 25$, 30.6%), shrubland ($n = 19$, 22.3%), forest ($n = 3$, 3.5) and wooded grassland ($n = 1$, 2.4%). More wildlife roadkills were recorded in the morning ($n = 53$, 62.3 %) especially Night jars ($n = 15$) on the birds' side, black rats ($n = 4$) on the mammals' side, and chameleons ($n = 2$) on reptiles' side than at afternoon ($n = 32$, 37.7%) hours, no significant differences were observed between those hours ($z = 0.63$, $p = 0.652$). The wildlife roadkill differed significantly in their body sizes, Kruskal-Wallis test ($\chi^2 = 28.94$, $p < 0.001$), with more having small body size ($n = 79$, 92.9%), followed by medium body size ($n = 4$, 4.7%) and large body size ($n = 2$, 2.4%). Small-bodied animals are more plentiful in the wild than large-bodied animals, hence higher chances of being affected by roadkill.

DISCUSSIONS

The study findings show that multiple taxa were involved in roadkill incidents within NCA, as had been previously reported from other regions worldwide (Drews, 1995; Kioko *et al.*, 2015a; Grilo *et al.*, 2020; Lyamuya *et al.*, 2021). This is because, worldwide, wildlife species traverse through roads often viewed as modified landscapes and face the risk of collision with vehicles (Aquino & Nkomo, 2021). Therefore, these findings confirm previous findings that wildlife taxa in most protected areas around the world, and more specifically in NCA, are impacted by roadkill

accidents. Again, it was found that more bird species were killed compared to other taxa within NCA, similar to those reported previously (Husby, 2016; Grilo *et al.*, 2020; Pinto *et al.*, 2020a; Lyamuya *et al.*, 2021; Medrano-Vizcaíno *et al.*, 2022). This could be because birds are known to search for food on roads more likely than other taxa and, hence, become more prone to roadkill (Husby, 2016), and are scared away by the oncoming vehicle, or they may clash when flying across the road at a low elevation. (Drews, 1995; Kioko *et al.*, 2015a; Husby, 2016). Moreover, the study by Husby (2016) reported that many bird species are probably put in danger by traffic because of speeding drivers, as birds lack the cognitive ability to swiftly digest all the information about distance, direction, and speed from vehicles. Amongst the birds in our study, the nocturnal Nightjars (*C. europaeus*) were most frequently killed. A similar observation was found by Grilo *et al.* (2020), where higher road mortality was more likely to occur in birds with smaller bodies, higher population densities, longer lifespans, omnivorous diets, or diets predominantly made up of plants and seeds. Another explanation for this could be that the late drivers who want to get out of the NCA gate before 06:00 pm or late hours are over speeding their vehicles while at the same time, the vehicle's lights are on to an extent that they blind the birds' eyes and also do not give a chance for the species to take off and run away (Møller *et al.*, 2011; Lyamuya *et al.*, 2021). The study also recorded the mortality of Secretary Birds (*S. serpentarius*), a species listed by the International Union for Conservation of Nature (IUCN) as endangered (IUCN, 2016). Previous studies found that Secretary Birds prefer road surfaces as hunting grounds for capturing different prey species or for taking advantage of the presence of carcasses on roads, which consequently exposes them to vehicle collisions (Lyamuya *et al.*, 2021).

Furthermore, the study found that a few mammals ($n = 16$) were recorded killed by vehicles, likely because they spent a very short time around the roads. After all, food and

water are limited near roads (Lyamuya *et al.*, 2021). Moreover, they probably might have been scavenged or removed by other animals/people in the area (Lee *et al.*, 2021). On the other hand, the Black rat (*Rattus rattus*) was the one species frequently hit by vehicles in the area. This might have been caused by their strong preference for feeding along vegetated road verges or sometimes due to their high population density (Lyamuya *et al.*, 2021). Other contributing factors could be the fact that they are small in size, which makes it more difficult for drivers to detect their presence and avoid them (Lyamuya *et al.*, 2021).

Although reptiles are most likely to be victims of roadkill due to their distinguishing long body morphology and their behaviour of frequently using roads as thermoregulation sites (Braz and Franc, 2016), again, few numbers of them were recorded killed in NCA, and most represented the chameleons (*C. chamaeleon*). In a study that was done across Europe, reptile casualties were found to be more pronounced compared to other taxa, which is in agreement with the current study (Morelli *et al.*, 2024) gathering 1688 records of reptiles killed by vehicle traffic. A total of 125 taxa were found documented as roadkill (100 native and 25 exotic species. This could probably be because this species is a slow-moving reptile, and therefore commonly hit by vehicles (Braz and Franc, 2016).

Interestingly, the study recorded only two killed carnivore species, including the bat-eared fox (*Otocyon megalotis*) and the wildcat (*Felis silvestris*). These animals were probably killed while either foraging or crossing the road, or due to their inability to escape from over speeding vehicles (Lyamuya *et al.*, 2021). When water is scarce, carnivores normally expand their home ranges and often cross roads through their expanded territory (Aquino & Nkomo, 2021). This might lead them to roadkill cases.

Additionally, the wildlife roadkills with small body sizes (<30Kg) were most frequently recorded killed compared to medium body-sized (31 – 50Kg) and large body-sized

(>50Kg), probably because they are small in size, which makes them more difficult for drivers to see and avoid (Lyamuya *et al.*, 2021).

The study results again found that wildlife roadkill occurred mostly in woodland habitats in the area. This finding strongly agrees with that found in southern Portugal, where there was a strong relationship between vulnerability to traffic deaths and foraging behavior and habitat type, with the most vulnerable species being the small woodland species that prefer to feed in shrubs and trees in the area (Pinto *et al.*, 2020b; Rendall *et al.*, 2021). Furthermore, ecological and behavioral characteristics increase the vulnerability of some species (Pinto *et al.*, 2020a). In addition, in woodland habitats, there are probably more forage resources, shelter, and cover for most of the wildlife found in the area, and thus the increased movements encountered during searching for those resources (Nathan *et al.*, 2020) could have increased their chances of collision with vehicles. Additionally, the type of habitat close to the roads can attract different wildlife species, especially birds, to the roads and increase their risk of becoming roadkill victims (Braz and Franc, 2016). Furthermore, Grilo *et al.* (2020) reported that high species richness areas had more roadkill incidents compared to low species richness areas.

CONCLUSION AND RECOMMENDATIONS

The present study recorded a substantial number of reptiles, mammals, and birds as roadkill victims on a stretch of the 82-km main road located in the NCA. The loss of individual wildlife species by roadkill has negatively impacted the structure of the terrestrial wildlife community and caused a severe loss of biodiversity in the area. Therefore, the study recommends enforcing punishments and penalties for over speeding drivers, installation of cameras to monitor speed limit, especially within protected areas, providing regular awareness on wildlife roadkill to different road users, constructing underground tunnels in suitable areas, and placement of road signs for

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different road users. This needs to be included in strategies and plans to mitigate the loss of biodiversity in the area. Furthermore, it is critical to proceed with roadkill monitoring in the area to avoid further biodiversity loss.

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SUB-THEME: INNOVATION AND TECHNOLOGY IN WILDLIFE CONSERVATION

DYNAMICS OF AGRICULTURAL AND SETTLEMENT DEVELOPMENT AND THEIR IMPACT ON MOUNTAIN HANANG NATURE FOREST RESERVE USING REMOTE SENSING

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ABSTRACT

This study assessed land use and land cover (LULC) changes and their environmental implications in Mountain Hanang Nature Forest Reserve, Manyara Region, from 1994 to 2024. The study aimed to analyze trends in land cover change, quantify forest cover loss and examine the relationship between forest loss and the expansion of agricultural land and settlements. Landsat satellite imagery from 1994 (Landsat 5 TM), 2004 (Landsat 5 TM), 2014 (Landsat 8 OLI) and 2024 (Landsat 8 OLI) was used for classification, supported by field-based ground truthing and 90 training samples. Supervised classification techniques were applied to map five land cover classes: forest, cultivated land, settlement, bare land and water bodies. Data analysis was performed using Google Earth Engine, ArcGIS and Microsoft Excel and a regression model was used to assess the influence of agriculture and settlement development on forest loss. The findings show significant land cover changes over the 30-year period. Forest cover declined from 4,955.83 hectares in 1994 to 4,442.12 hectares in 2024, representing a net loss of 10.3%. However, a 25% decline occurred in the last decade alone (2014–2024). In contrast, cultivated land increased from 1,168.24 ha to 4,531.58 ha, while settlement areas expanded from 264.73 ha to 1,167.82 ha. Regression analysis results revealed that agricultural expansion is the main factor contributing to forest loss, with settlement growth showing a weaker relationship. The study further found that the continued conversion of forest to farmland and built-up areas has resulted in biodiversity loss, habitat fragmentation, increased soil erosion, disrupted water systems, and reduced ecosystem services.

The study concludes that human activities, especially small-scale farming driven by maize and sunflower cultivation, have intensified land pressure in and around the forest reserve. Without effective intervention, the ecological functions and biodiversity of Mountain Hanang may continue to decline. The study recommends enhanced enforcement of conservation regulations, participatory forest management, sustainable agricultural practices and awareness programs to restore and protect the remaining forest cover.

Keywords: Biodiversity loss, Deforestation, Ecosystem services, Regression analysis, Soil erosion, Satellite imagery.

INTRODUCTION

Forests cover about 31% of Earth's land and provide vital services such as climate regulation and biodiversity support, but they are increasingly threatened by farming,

urban growth, and illegal logging as the global population rises (FAO, 2020). Despite international conservation efforts such as the REDD+ mechanism, which provides financial incentives for forest conservation, these efforts often face challenges due to governance issues

and socio-economic pressures (Angelsen *et al.*, 2018).

Mangrove forests in Asia, home to over 50 true species across 112 countries, are highly sensitive to land-use changes, but agricultural and settlement expansion have caused major habitat loss and disrupted natural ecological processes, raising global concerns about long-term forest degradation (Gerona-Daga & Salmo, 2022; Giri *et al.*, 2015). Internationally, the loss of forests is largely driven by economic development, population growth, and the increasing demand for land (Kissinger *et al.*, 2016). However, this agricultural growth often leads to biodiversity loss, soil degradation, and reduced water quality, highlighting the need for better land management practices (Ntukey *et al.*, 2022) representing an important biodiversity hotspot. However, deforestation and forest degradation due to the high demand for fuelwood has reduced forest cover and diversity, with unknown consequences for associated terrestrial carbon stocks in this LCF system. Our study assessed spatio-temporal land use and land cover changes (LULC). Efforts like the Paris Agreement and the UNFCCC focus on reducing emissions from deforestation, but achieving sustainable outcomes requires more robust policy enforcement and support for local communities (Kissinger *et al.*, 2016). Africa loses approximately 3.9 million hectares of forest each year due to agricultural expansion, charcoal production, and population growth, posing significant environmental and socio-economic challenges (FAO, 2020). This trend threatens the continent's ability to maintain vital ecosystem services, including water regulation and soil fertility (Kissinger *et al.*, 2016). Countries such as the Democratic Republic of Congo, Nigeria, and Tanzania face severe deforestation driven by agricultural expansion and settlement growth, while initiatives like the African Forest Landscape Restoration Initiative aim to address this issue but their effectiveness remains limited due to weak governance and insufficient resources (FAO, 2020).

In East Africa, forests are under increasing pressure from agriculture, illegal logging and infrastructure development (Blomley *et al.*, 2008). In Uganda, subsistence farming has significantly altered land cover, particularly around Mount Elgon National Park, where agricultural activities have caused extensive deforestation between 1978 and 2020 (Opedes *et al.*, 2022). Also, the Cherangany Hills forest ecosystem in Kenya has been severely impacted by agriculture and settlement growth, resulting in reduced forest cover, weakened biodiversity, and declining water resources, highlighting the need for sustainable land-use practices (Rotich & Ojwang, 2021). Tanzania's forests are facing significant decline, mainly due to factors such as agricultural expansion and livestock grazing (Borges *et al.*, 2022). Also, a study of land use changes between 1995 and 2010 showed that while farmland increased, forests significantly declined, with an annual deforestation rate of 320,067 hectares (0.9%), demonstrating the agricultural growth as a primary cause of forest loss (Nzunda & Midtgaard, 2019) forest, bushland, grassland, cultivation and other land use and cover (built up areas, bare land, etc..

However, the National Forest Policy Implementation Strategy (2021–2031) identifies agriculture and settlement growth as key causes of forest loss, and to address this, it promotes better forest governance, sustainable land use, stronger conservation, and stakeholder cooperation for long-term ecological stability (URT, 2021). The Manyara Region is experiencing serious forest loss due to small-scale farming, overgrazing, and settlement growth, which are putting pressure on forest ecosystems and causing fragmentation and depletion of forest reserves (Godgift *et al.*, 2014).

MATERIALS AND METHODS

Study area

The Hanang Mountain Forest Reserve (Fig.1) is located in the Manyara region of Tanzania,

specifically situated between 4° 25' - 4° 35' S and 35° 20' - 35° 25' E (Godgift *et al.*, 2014). Manyara lies between latitudes 4° 25' and 5° 00' south of the equator and longitudes 34° 45' and 35° east of the Greenwich Meridian (Lulandala, 2023). The Hanang Mountain Forest Reserve

has a sub-humid climate with a wet season from November to April, bringing 600–1,200 mm of rainfall that supports vegetation, while the dry season from May to October brings heat and low rainfall, affecting ecosystems, wildlife, and nearby farming (Godgift *et al.*, 2014).

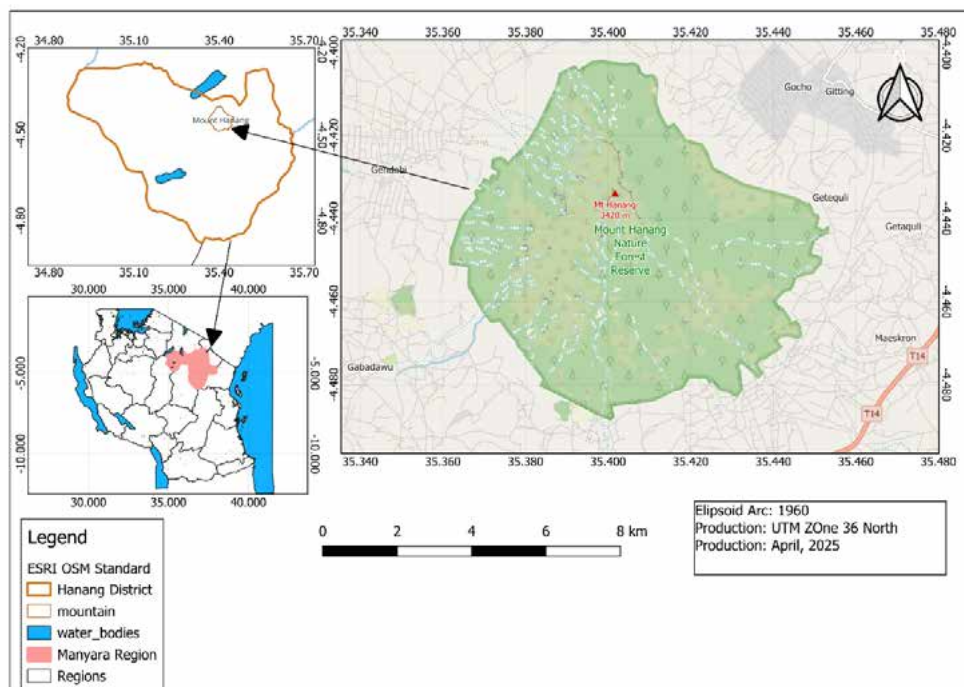


Figure 36. A map showing the Mountain Hanang Nature Forest Reserve in May 2025. Source: QGIS version 3.24.1

Data Collection

This study utilized satellite imagery to support the assessment of land cover changes within the Mountain Hanang Nature Forest Reserve. Specifically, Landsat imagery with 30-meter spatial resolution was acquired for 1994, 2004, 2014, and 2024. These years were selected based on the availability of cloud-free images and their relevance to major shifts in land use patterns within the study area. The imagery was accessed through the Google Earth Engine (GEE) platform, which provides direct access to the USGS Landsat archive without the need for downloading. The selected imagery came from Landsat 5 Thematic Mapper (TM), Landsat 7 Enhanced Thematic Mapper Plus (ETM+), and Landsat 8 Operational Land Imager (OLI) and Thermal Infrared Sensor (TIRS), depending

on the year of observation. The use of multi-temporal Landsat data provided a consistent and reliable foundation for detecting land cover transitions across a 30-year period. These satellite images served as the basis for further analysis of forest cover dynamics and the impacts of human activities such as agricultural expansion and settlement development (Phiri *et al.*, 2023). Ground truthing was employed as a primary data collection method to validate land cover classifications and support accurate image interpretation (Kanja *et al.*, 2019). This process involved direct field observation of selected locations within the study area, based on a stratified random sampling design (Dong *et al.*, 2020). A total of 60 ground truth points were randomly selected from three major land cover zones: forest, agriculture, and settlement with 20 points per zone(strata). At each point,

data were collected on land cover type, current land use, and any visible human activities, such as cultivation, grazing, or infrastructure.

Data Analysis

The analysis of land cover changes was carried out using a combination of Google Earth Engine (GEE) and supporting tools, including ArcGIS and Excel (Borges *et al.*, 2022). GEE, a cloud-based geospatial analysis platform, was used due to its capability to process large satellite datasets, implement advanced classification algorithms, and perform efficient time-series analysis (Opedes *et al.*, 2022). ArcGIS was used for spatial visualization and map layout design, while Excel facilitated statistical computations and graphical representations. The following analytical techniques were employed to assess land cover dynamics within the study area.

Change detection analysis

To assess land cover changes in the Mountain Hanang Nature Forest Reserve, a post-classification comparison technique was applied using classified satellite images from the years 1994, 2004, 2014, and 2024. This method involved analyzing differences between independently classified land cover maps to detect and quantify temporal transitions.

The analysis incorporated not only the forest reserve but also a 3.5 km buffer zone surrounding its boundary (Kija *et al.*, 2020). Including this buffer zone enabled the study to capture the influence of adjacent human activities, such as agricultural expansion and settlement development, on the forest ecosystem. This broader spatial scope provided a comprehensive understanding of land cover dynamics both inside and immediately outside the protected area.

The primary focus was on identifying forest loss and its conversion into agricultural and settlement areas, which constitute the major land cover transformations observed within the study region. By comparing land cover

categories across the selected time intervals, the study traced the progression of human-induced changes and evaluated the impact of land use activities on forest cover over the 30-year period (Nzunda & Midtgaard, 2019) forest, bushland, grassland, cultivation and other land use and cover (built up areas, bare land, etc.. This approach facilitated a clear understanding of the direction, magnitude, and spatial distribution of land cover changes, supporting the generation of statistical summaries that illustrate trends in land use transitions. The method is consistent with the approach applied by (Kanja *et al.*, 2019) in their analysis of agricultural expansion in Mwekera Forest Reserve using post-classification comparison to detect long-term landscape change through remote sensing and GIS techniques.

Supervised classification and Normalized Difference Vegetation Index analysis (NDVI)

A supervised classification technique was applied using the Random Forest algorithm to categorize satellite imagery into five primary land cover classes: Open Forest, Closed Forest, Cultivated Land (Agriculture), Developed Areas (Settlements) and Bare Land (Rotich & Ojwang, 2021). A total of 501 training samples were generated through stratified random sampling, ensuring that the training data adequately represented the landscape heterogeneity across the study area. These samples were used within the Google Earth Engine platform to extract spectral signatures specific to each land cover category, which were then used to train the classification model.

In addition to the training samples, to assess the reliability of the classification, a total of 60 ground truthing points were used for validation. These points were collected during fieldwork and were evenly distributed across the three main land cover zones: forest, agriculture, and settlement. Each validation point was compared against the corresponding land cover classification output derived from the supervised classification process. This comparison was used to perform an accuracy assessment,

which involved generating a confusion matrix to evaluate how well the classified results matched the actual ground observations (Islam *et al.*, 2023). The assessment produced an overall classification accuracy of 80% and a Kappa coefficient of 0.83, indicating a strong level of agreement between the classified land cover map and the actual ground conditions observed in the field.

Also, the Normalized Difference Vegetation Index (NDVI) was calculated to assess vegetation health and assist in distinguishing forest structure within the study area (Gandhi *et al.*, 2015). NDVI values were derived using the red and near-infrared (NIR) bands from the Landsat satellite imagery. In this study, areas

classified as Closed Forest, characterized by dense canopy cover which exhibited higher NDVI values and appeared as dark green on the map outputs. Conversely, Open Forest areas, which have more scattered tree cover, showed moderate NDVI values and appeared as light green. This variation in NDVI values provided an effective means of detecting changes in vegetation condition, identifying areas of forest degradation, and monitoring spatial and temporal patterns of land cover transformation. This technique is consistent with the methods used by (Gandhi *et al.*, 2015; Islam *et al.*, 2023) who applied Normalized Difference Vegetation Index (NDVI) and supervised classification to study vegetation dynamics and land use change

Table 10: The LULC classification scheme based on recommendations by the Intergovernmental Panel on Climate Change (IPCC).

LULC Class	Description
Closed Forest	Areas with dense tree cover, where canopy closure exceeds 70%. These zones represent primary or mature forests with minimal disturbance and high ecological integrity.
Open Forest	Areas with partial tree cover, typically ranging from 30% to 70% canopy closure. These represent secondary or degraded forests and regenerating woodlands.
Cultivated Land	Land actively used for agriculture, including seasonal and perennial crops. This corresponds to the “Cropland” category in IPCC definitions.
Developed Areas	Areas dominated by human structures and infrastructure such as villages, houses, and roads. These match IPCC’s “Settlements” classification.
Bare Land	Land with very sparse or no vegetation cover (<10%), including exposed soil, rocky areas, degraded lands, or areas cleared by human activity such as burning.

Visualization and Map Layout

After classification, the final land cover maps for the years 1994, 2004, 2014, and 2024 were exported to ArcGIS for visualization and map layout design (Phiri *et al.*, 2023). In ArcGIS, detailed map layouts were developed to effectively present the spatial distribution of land cover types and their changes over time. Moreover, essential cartographic elements such as a legend, scale bar, north arrow, map title, and appropriate symbology were added to

each map to improve clarity and interpretation (Borges *et al.*, 2022). These components helped communicate the results more clearly by visually displaying the extent and direction of land cover transformations, including forest loss, agricultural expansion, and settlement development. This visualization step enhanced the overall presentation of the study findings and supported spatial analysis for better understanding of land use dynamics. To quantify land cover changes between 1994 to 2024, this study used Microsoft Excel to perform

statistical analyses based on the classified land cover data by (San *et al.*, 2024). The focus was to evaluate the extent of forest cover loss and the expansion of other land use classes such as agriculture, settlements, and bare land over the 30-year period. Whereas, In Microsoft Excel, percentage and area values (in hectares) for each land cover class as Open Forest, Closed Forest, Agriculture, Settlements, and Bare Land, were summarized and used to generate graphs and charts. This visual representation clearly showed the extent, temporal trends in land cover changes and direction of forest cover loss, alongside the increase in agricultural and settlement areas. Furthermore, Microsoft Excel was used to perform regression analysis to examine the relationship between forest cover loss and the expansion of agricultural land and settlements. This statistical approach helped quantify the impact of human activities on forest decline within the study area. This integrated use of statistical tools provided clear insights into the spatial and temporal patterns of land cover transformation. The approach is similar to that applied by (San *et al.*, 2024), who combined remote sensing outputs with statistical techniques to analyze forest dynamics in Myanmar.

RESULTS

Land Cover Trends: 1994–2004

Between 1994 and 2004, the area under Closed Forest increased from 4,321.24 ha (21.04%) to 4,735.54 ha (23.07%), reflecting modest forest recovery likely influenced by conservation enforcement and natural regeneration. Similarly, Open Forest rose from 634.59 ha (3.09%) to 936.36 ha (4.56%), possibly indicating recovery in previously degraded patches. Also, Cultivated Land expanded notably from 8,300.26 ha (40.39%) to 9,194.66 ha (44.83%), reflecting growing agricultural demand, especially in the buffer zone. Developed Areas grew from 2,919.34 ha (14.22%) to 3,492.19 ha (17.02%), highlighting initial phases of settlement expansion. Meanwhile, Bare Land sharply declined from 4,367.20 ha (21.26%) to

2,161.26 ha (10.53%), suggesting intensified land use or partial vegetation recovery.

Land Cover Trends: 2004–2014

During this decade, Closed Forest peaked at 5,363.79 ha (26.14%), marking the highest coverage recorded throughout the study period. This upward trend likely reflects the impact of regeneration efforts, conservation interventions, and reduced anthropogenic disturbance, possibly through natural succession or afforestation programs. Simultaneously, Open Forest slightly declined to 598.19 ha (2.92%), which may be a result of transition into denser, closed-canopy forest, indicating ecological recovery in some areas. Notably, Cultivated Land increased significantly to 9,841.10 ha (47.95%), its highest level in the entire 30-year period, this substantial rise suggests intensified agricultural activities, likely driven by population pressures and expansion into previously less-utilized areas such as degraded forest or bare land. In contrast, Developed Areas declined slightly to 3,363.68 ha (16.39%), which could reflect shifts in rural-urban migration patterns or limited infrastructure development during the period. Additionally, Bare Land decreased further to 1,354.30 ha (6.60%), suggesting continued land use conversion, either through agricultural expansion or natural vegetation recovery.

Land Cover Trends: 2014–2024

The period between 2014 and 2024 showed major changes in land cover patterns, many of which indicate increased human impact on the landscape. The most significant change was the sharp decline in Closed Forest, which dropped from 5,363.79 ha (26.14%) to 1,680.53 ha (8.18%), this loss of about 3,683 ha suggests growing pressure from agricultural expansion, fuelwood collection and settlement development, especially around the forest buffer zones. At the same time, Open Forest grew from 598.19 ha (2.92%) in 2014 to 2,761.59 ha (13.44%) in 2024. This rise was likely caused by human activities that disturbed the canopy and thinned the forest. Areas that were once

dense became more open. NDVI results also showed lower vegetation health and cover in these areas. Cultivated Land decreased from 9,841.10 ha (47.95%) to 7,413.34 ha (36.09%), showing a reduction of more than 2,400 ha, this may be due to several reasons, including soil degradation, population movement, or a shift in land use toward construction and infrastructure. In fact, part of the land that was previously used for farming appears to have been converted into Developed Areas or left as Bare Land. During the same period, Developed Areas increased from 3,363.68 ha (16.39%) to 5,796.72 ha

(28.22%), reflecting the growth of human settlements and infrastructure such as houses, roads, and services, this change links directly with the decrease in forest and cultivated areas, as land was cleared for development purposes. In addition, Bare Land rose from 1,354.30 ha (6.60%) to 2,890.12 ha (14.07%), which may be explained by abandoned farms, land degradation, or deforested areas left unused, this suggests that some areas are no longer being used productively and may face risks like erosion or desertification if not restored.

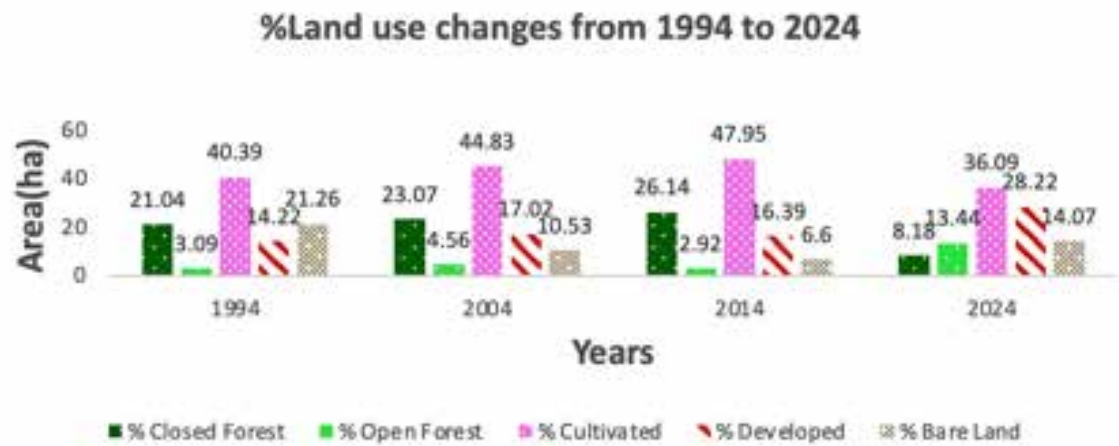
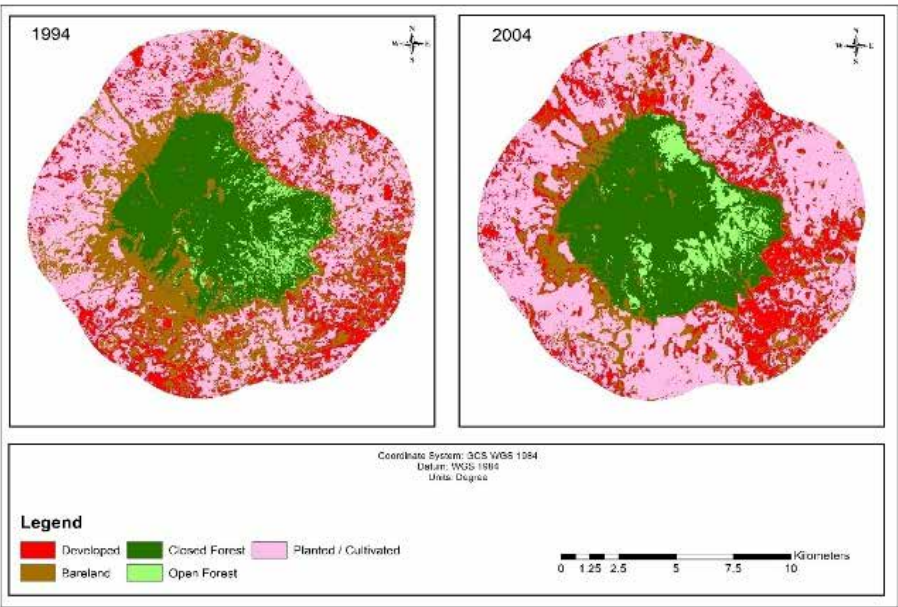


Figure 2: Percentage Distribution of Land Use/Land Cover (LULC) Classes in Mountain Hanang Nature Forest Reserve and Its Buffer Zone (1994–2024)



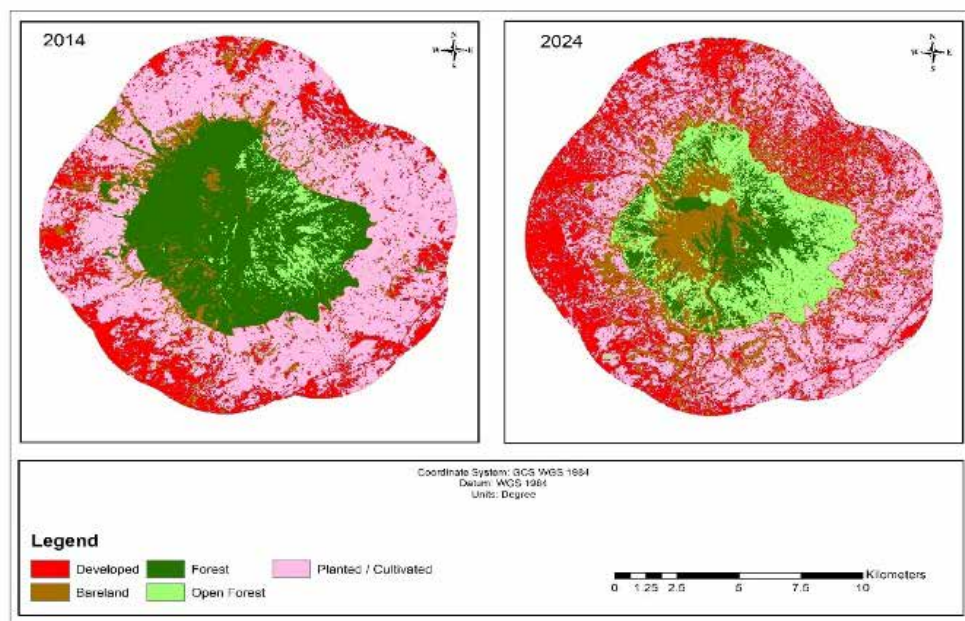


Figure 3: Extent of Land Use and Land Cover Changes from 1994 to 2024: source ARC GIS 10.8

Land Cover Conversion Trends in Mountain Hanang Nature Forest Reserve (1994–2024)

The post-classification comparison of land cover maps from 1994 to 2024 revealed significant land cover conversions within Mountain Hanang Nature Forest Reserve and its 3.5 km buffer zone. These conversions indicate dynamic land use changes influenced by agricultural expansion, forest disturbance, and settlement development. The Land conversions Trends are shown to the table 1 below

Table 2: Nature and Proportion of Land Use/Land Cover Conversions in Mountain Hanang Nature Forest Reserve from 1994 to 2024.

Class name	Area	% percentage
Bare land -> Developed	1091.703003	5.33
Bare Land->Forest	157.0900421	0.77
Bare Land->Open Forest	436.2113037	2.13
Bare Land->Planted / Cultivated	1537.256104	7.5
Developed->Bare Land	195.098877	0.95
Developed->Open Forest	5.06098938	0.02
Developed->Planted / Cultivated	1413.272461	6.9
Dense Forest->Bare Land	1101.28479	5.37
Dense Forest->Developed	3.781377077	0.02
Dense Forest->Open Forest	1784.321045	8.7
Dense Forest->Planted / Cultivated	2.489827156	0.01
No Change	8775.416016	42.82

Open Forest->Bare Land	43.01025391	0.21
Open Forest->Developed	0.658254087	0
Open Forest->Closed Forest	85.34195709	0.42
Open Forest->Planted / Cultivated	0.180817232	0
Planted / Cultivated->Bare Land	455.3872375	2.22
Planted / Cultivated->Developed	3404.923828	16.61
Planted / Cultivated->Forest	0.090413973	0
Planted / Cultivated->Open Forest	6.471028805	0.03
Total		100

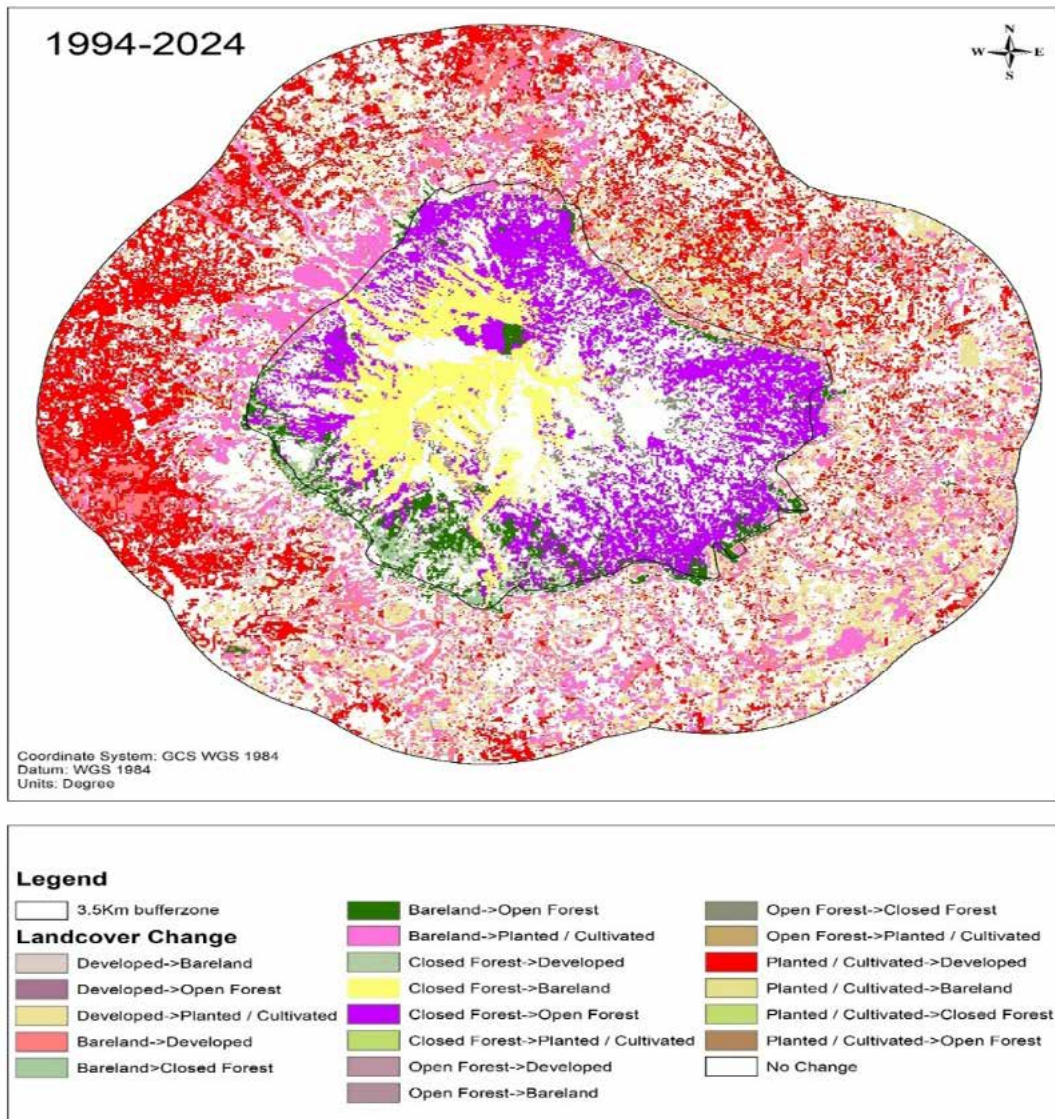


Figure 4: Land Cover Conversion Trends from 1994 to 2024: source ARC GIS 10.8

Forest cover loss over the study period.

The total forest area in the study area showed variation between 1994 and 2024. In 1994, the total forest cover was 4,955.83 hectares, with 4,321.24 hectares classified as closed forest and 634.59 hectares as open forest. By 2004, the total forest area had increased by 14% to 5,671.90 hectares, due to increases in both closed forest (4,735.54 ha) and open forest (936.36 ha). Between 2004 and 2014, total forest cover further increased by 5.1% to reach 5,961.98 hectares. During this period,

closed forest area grew to 5,363.79 hectares, while open forest decreased to 598.19 hectares. However, from 2014 to 2024, the total forest area decreased sharply by 25.5% to 4,442.12 hectares. This decline was mainly due to a significant reduction in closed forest cover to 1,680.53 hectares, although open forest increased to 2,761.59 hectares. Overall, these results indicate a net loss of forest cover of approximately 513.71 hectares (10.4%) from 1994 to 2024, with the most significant forest loss occurring in the last decade.

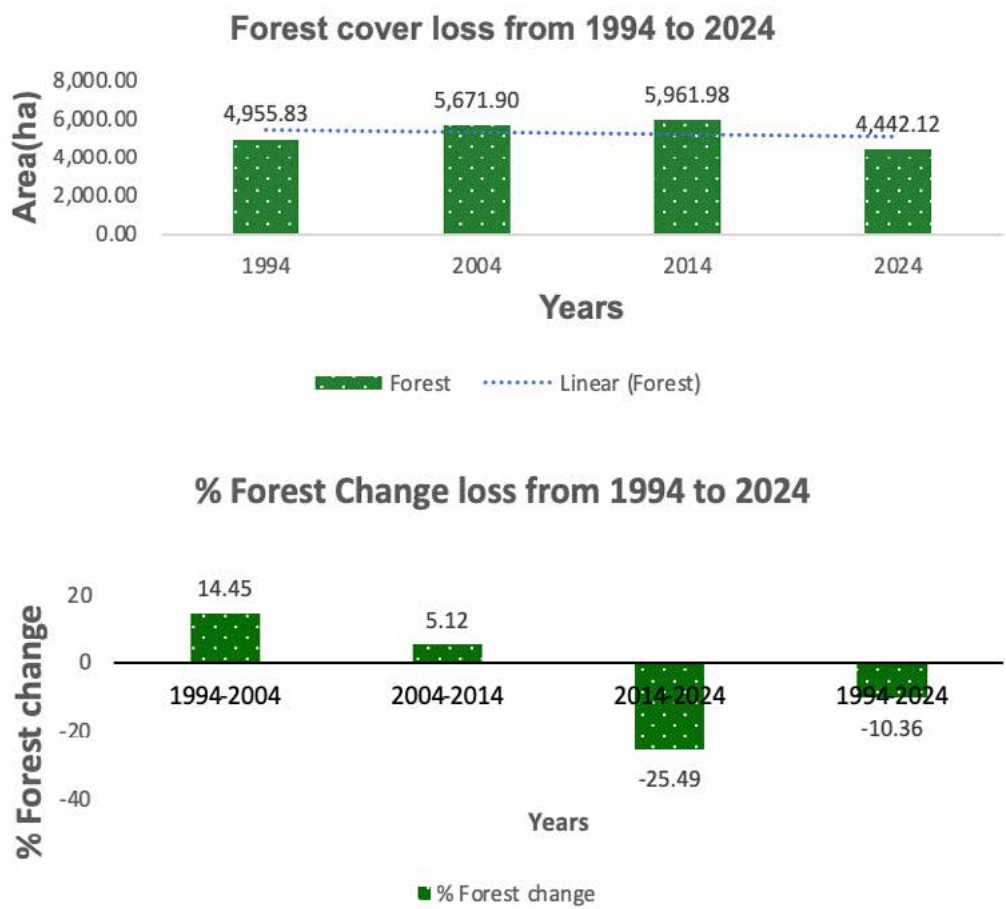


Figure 5: Trend in Total Forest Cover in Mountain Hanang Nature Forest Reserve (1994–2024)

The relationship between forest cover loss and the expansion of agricultural land and settlement development.

A multiple linear regression was performed to assess how the increase in cultivated

(agricultural) land and developed (settlement) areas relates to forest cover loss in the study area. The model produced a Multiple R of 0.996 and an R Square of 0.993, indicating that about 99.3% of the variation in forest

cover loss is explained by the combined changes in cultivated and developed land. The F-statistic was 66.70 with a Significance F of 0.086, which shows the model is statistically significant at the 10% level. The coefficient for cultivated land was 0.667, with a p-value of

0.074, indicating a strong positive relationship with forest loss. In contrast, the coefficient for developed land was 0.0224, with a higher p-value (0.786), suggesting a very weak and statistically insignificant effect.

Table 3: Relationship Between Forest Cover Loss and the Expansion of Agricultural and Developed Land From 2014 to 2024

Source	df	SS	MS	F	Significance F
Regression	2	1,400,596	700,298.1	66.95	0.0861
Residual	1	10,460.7	10,460.7		
Total	3	1,411,057			

Regression Coefficients

Variable	Coefficient	Std. Error	t Stat	P-value	Lower 95%	Upper 95%
Intercept	-625.798	869.0739	-0.720	0.6027	-11,668.43	10,416.83
Cultivated	0.6663	0.07783	8.5615	0.0740	-0.3226	1.6553
Developed	0.0249	0.06373	0.3909	0.7628	-0.7849	0.8347

DISCUSSION

Environmental Implications of Land Use and Land Cover Changes in Mountain Hanang Nature Forest Reserve (1994–2024)

Land Use and Land Cover Changes (LUCCs) over the past 30 years in Mountain Hanang Nature Forest Reserve have caused significant environmental effects. Satellite imagery and classification results show that forest cover fluctuated, representing approximately 24.1% of the study area in 1994 and declining slightly to 21.6% in 2024. Meanwhile, agricultural land expanded from about 15.4% to 39.5%, and settlements grew from 3.5% to 10.2% of the total area. These shifts reflect increasing human pressure on the natural environment, with important consequences for biodiversity, soil stability, and water resources as in the sections below.

Forest Cover Decline and Biodiversity Loss

The consistent decline in forest cover, shrinking from 24.1 in 1994 to 21.6% in 2024, reflects intensive land cover changes resulting from agricultural expansion, settlement encroachment, and resource extraction. This trend mirrors patterns in other East African mountain ecosystems, where anthropogenic pressures drive habitat loss and fragmentation (Yadeta *et al.*, 2022). Forest loss directly threatens the reserve’s rich biodiversity. Mountain Hanang hosts a wide range of flora, including *Podocarpus spp.*, *Ficus spp.*, *Albizia schimperiana*, *Calodendrum capense*, *Olea europaea*, *Prunus africana*, *Hagenia abyssinica*, *Juniperus procera*, and ericaceous shrubs adapted to high-altitude zones (Godgift *et al.*, 2014). The fauna includes species such as klipspringers (*Oreotragus oreotragus*), bush pigs (*Potamochoerus larvatus*), vervet monkeys (*Chlorocebus pygerythrus*), olive baboons (*Papio anubis*), and rare endemic species such

as the *Trioceros bitaeniatus* chameleon and mole-rats like *Fukomys hanangensis* (Faulkes *et al.*, 2017). As forest patches shrink and become isolated, wildlife habitats become fragmented, reducing species movement, breeding success, and food access. This increases the risk of local extinctions, especially for species with small ranges or specific habitat needs. Similar trends were reported in the Greater Serengeti Ecosystem, where biodiversity loss was linked to agricultural and settlement encroachment (Msofe *et al.*, 2019).

Agricultural Expansion and Livelihood Conflicts

The sharp rise in agricultural land from 15.4% to 39.5% shows growing pressure to meet food and income needs. This trend is similar to what has been observed in northern Tanzania and Zambia, where land inside or near forest reserves has been cleared extensively for farming (John Mwita, 2013; Zhunusova *et al.*, 2025). In Hanang, maize and sunflower are the main crops grown. These crops are vital for household incomes but require large areas of land, often leading to forest clearing on steep slopes and sensitive catchment zones. Practices like slash-and-burn and shifting cultivation speed up deforestation and cause long-term damage to the land. While agriculture helps meet immediate needs, the loss of important forest services such as pollination, water retention, and nutrient cycling threatens the land's long-term health. As soil productivity declines, farmers are forced to clear more forest, creating a harmful cycle of land degradation and increased dependence on farming.

Soil Erosion and Land Degradation

Deforestation in mountainous areas like Hanang removes the plants that naturally hold the soil in place. When agriculture expands onto steep slopes without proper soil conservation, surface runoff increases and causes faster soil erosion. The data show that much of the forest loss happened in steep areas, which has led to rapid land degradation. This matches findings from the Ethiopian Highlands, where clearing

forests for farming caused severe soil fertility loss and the formation of gullies (Wubie *et al.*, 2020). In Mountain Hanang, soil erosion lowers farm productivity, making agriculture less sustainable and pushing farmers to clear even more forest. Without plants to protect the soil, the land becomes more at risk of degradation and desertification.

Disruption of Hydrological Functions

The land use and land cover (LULC) changes within the reserve have also affected the natural water cycle. Forests help recharge groundwater, keep streams flowing, and reduce surface runoff. The drop in forest cover to only 25.5%, along with more hard surfaces from settlements and farms, has likely lowered water infiltration and caused more frequent and stronger flash floods. Similar problems have been seen in East African watersheds, where changes in land cover caused water shortages and changes in seasonal water flow (Bewket & Abebe, 2013). In Hanang, the drying of springs, less soil moisture, and seasonal water shortages in nearby communities may be connected to these changes in land cover.

Increased Carbon Emissions

The loss of forest in the Mountain Hanang Nature Forest Reserve likely caused a large release of carbon stored in trees and soil. (Godgift *et al.*, 2014), estimated that trees in the reserve store about 48.4 tons of carbon per hectare, and the soil stores between 31.0 and 64.2 tons of carbon per hectare depending on the altitude. Even though this study did not measure the exact amount of carbon lost, the 25.5% decrease in forest cover from 1994 to 2024 suggests a significant release of carbon caused by changes in land use and forest cover. This loss of forest reduces Tanzania's ability to meet climate goals like those under the REDD+ program, which aims to keep carbon stored in forests.

Decline of Ecosystem Services and Rural Livelihoods

The LULC trends observed pose direct and indirect risks to local livelihoods and ecosystem services. The expansion of agriculture and settlements may bring short-term food and income benefits, but the long-term effects include reduced water quality, soil degradation, crop failure, and increased vulnerability to climate change. Communities depending on the forest for firewood, herbal medicine, honey, and wild fruits face declining access (Gashaw *et al.*, 2018). The rise in settlement area from 3.5% to 10.2% over 30 years reflects growing population pressure, which, if unmanaged, will continue to strain the natural capital that rural communities rely on for resilience and survival.

Implications for Conservation Goals

The observed land cover changes within Mountain Hanang Nature Forest Reserve have serious implications for long-term conservation and sustainable forest management. The consistent conversion of closed forest into open forest and developed land reflects not only increased human pressure but also a decline in the effectiveness of forest protection and law enforcement mechanisms (Ntukey *et al.*, 2022). This trend suggests that existing conservation frameworks may not be adequately addressing the drivers of deforestation, particularly agricultural expansion and settlement encroachment. If such transformations continue unchecked, they threaten to undermine national and global conservation objectives, including the protection of critical water catchment areas, preservation of biodiversity, and the role of forests in climate change mitigation (URT, 2021). The degradation of forest ecosystems in Mountain Hanang could lead to loss of endemic flora and fauna, soil erosion, reduced carbon sequestration capacity, and disruption of ecological functions that support local livelihoods. Moreover, the findings underscore the need to strengthen community engagement in conservation. The limited participation of surrounding communities in forest management initiatives has likely contributed to the lack of

ownership and non-compliance with forest use regulations (Msofe *et al.*, 2019). Therefore, enhancing participatory forest management, coupled with alternative livelihood options, could be key in addressing the underlying socioeconomic drivers of forest loss.

To reverse the current trends and ensure the ecological integrity of the reserve, a combination of strategic interventions is urgently required. These should include targeted reforestation programs, zoning, sustainable land-use planning, reinforcement of protection policies, and capacity building for local enforcement units (Lupala *et al.*, 2015). Without such coordinated efforts, the ability of Mountain Hanang Nature Forest Reserve to function as a biodiversity hotspot and climate buffer will be significantly compromised.

To quantify forest cover loss over the study period

The analysis of land cover data for Mountain Hanang Nature Forest Reserve shows that forest cover did not remain constant between 1994 and 2024. Overall, the forest area slightly increased from 4,955.83 ha in 1994 to 5,961.98 ha in 2014, mainly due to better conservation measures and less human pressure during that time. However, a sharp decline was seen afterwards, dropping to 4,442.12 ha by 2024, which means a net loss of about 10% over the entire study period, with a major drop of about 25% in the last decade. This severe reduction after 2014 can be explained by two main factors. First, human activities increased around the reserve, with more land being cleared for small-scale farming and new settlements. This agrees with findings by (Kanja *et al.*, 2019) in Zambia, who reported large forest losses mainly due to expansion of cropland and settlements in forest reserves. The field visit and ground truthing confirmed that parts of Hanang were converted to agriculture and settlement matching what is seen in the classified images.

However, a major natural disaster worsened the situation. On 2023, a strong landslide hit Mount

Hanang following unusually heavy rains. This landslide destroyed forest trees and topsoil on the slopes, contributing to the significant drop in forest cover recorded between 2014 and 2024. Studies have shown that heavy rainfall and land use changes increase the risk of landslides in mountainous areas. (Mulugeta *et al.*, 2024), used geospatial and statistical methods to model landslide risk in the Kulfo River catchment in Ethiopia and found that factors such as rainfall, land cover, and slope steepness strongly influence landslide occurrence.

This loss of forest does not only affect trees. It threatens the local biodiversity, especially plant and animal species that are endemic to Hanang, as well as common animals that depend on forest cover for shelter and food. In addition, damage to forest cover can lead to more soil erosion, reduced water quality, and lower river flow in streams that communities use. These implications are similar to those observed in Mwekera Forest Reserve, where unsustainable land use and encroachment caused biodiversity loss and water stress (Kanja *et al.*, 2019).

The classification results, supported by accurate ground truthing, confirm that urgent actions are needed. The forest loss shows that current conservation measures are not enough to control both human encroachment and sudden disasters like landslides. Therefore, new interventions should include better community awareness, strict land use controls, early warning systems for landslides, and restoration of degraded slopes to reduce erosion and further forest loss

The relationship between forest cover loss and the expansion of agricultural land and settlement development

The regression results clearly show that agricultural land expansion is the main factor contributing to forest cover loss in the study area. The strong positive coefficient for cultivated land means that as communities expand farmland to meet subsistence and market needs, forested areas are directly converted into agricultural plots. This pattern is consistent

with findings from other parts of Tanzania. For instance, (Ntukey *et al.*, 2022), reported that agricultural encroachment remains the leading driver of deforestation in montane and nature forest reserves across northern Tanzania.

The weak relationship between settlement development and forest loss implies that, compared to farmland expansion, the clearing of forests for housing and infrastructure has had a smaller impact in this specific area. This aligns with observations by (Msofe *et al.*, 2019), who noted that although urbanization is increasing in Tanzania, its spatial footprint is still smaller than agricultural land use change, which remains the dominant force altering forest cover.

The statistical model's outcome reveals that forest cover loss in Mountain Hanang Nature Forest Reserve is strongly driven by agricultural expansion, while settlement growth plays a minimal direct role. This finding implies that deforestation in the area is primarily the result of livelihood activities particularly small-scale and possibly shifting cultivation rather than urban development or infrastructural spread. This trend is consistent with land use studies such as (Jahanifar *et al.*, 2018), who found that agricultural drivers had a more significant impact on forest land conversion than built-up expansion, especially in rural catchments with low industrialization. While the model was only significant at the 10% level, this still provides a useful basis for management decisions, particularly considering the limited number of time points analyzed. The implication is that forest conservation efforts should prioritize regulating and supporting more sustainable agricultural practices, possibly through community-based land use zoning, agroforestry incentives, or reforestation programs targeting former farmland. Furthermore, the weak statistical link between forest loss and developed areas suggests that settlement growth may be more of an indirect driver facilitating agricultural demand rather than a direct agent of deforestation. Overall, these results highlight the urgent need for

sustainable land use planning. If agricultural expansion is not properly controlled through policies such as land use zoning, community-based forest management, and agroforestry promotion, forest reserves will continue to shrink. Effective conservation should therefore address the root causes of farmland expansion while accommodating local livelihood needs.

CONCLUSION

This study found that forest cover in Mountain Hanang Nature Forest Reserve has declined over the past 30 years due to farming and settlement expansion, leading to serious environmental problems like loss of biodiversity, soil erosion, and reduced water supply, and showing that stronger actions are needed to stop further forest destruction and protect the reserve's ecological value.

RECOMMENDATION

To protect the forest and support local communities, it is recommended to strengthen community participation, enforce land use rules, promote sustainable farming, improve land use planning, build early warning systems, and support alternative livelihoods in line with the National Forest Policy Implementation Strategy (2021–2031).

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SPATIAL PATTERNS OF MAMMAL DIVERSITY AND RICHNESS FROM YEAR-LONG CAMERA TRAPPING IN THE KITENDENI WILDLIFE CORRIDOR, NORTHERN TANZANIA

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ABSTRACT

Understanding how mammal communities are structured across human-modified wildlife corridors is crucial for maintaining ecological connectivity and informing effective land-use planning. We conducted a year-long camera-trap survey in the Kitendeni Wildlife Corridor (northern Tanzania), a transboundary linkage between Kilimanjaro National Park (Tanzania) and Amboseli National Park (Kenya). Cameras were deployed across grassland, shrubland, and tree-cover habitats, and images were processed using an automated Wildlife Insights workflow followed by expert verification. We quantified habitat-level species richness and diversity, characterised encounter structure using effort-standardised relative abundance indices (RAI), estimated site use using naïve occupancy, and mapped spatial clustering of the number of species detected per camera station using kernel density estimation and inverse distance weighted (IDW) interpolation. Shrubland supported the broadest assemblage and the highest diversity, highlighting the importance of structurally heterogeneous habitats within the corridor mosaic. Encounter rates were dominated by a limited set of widely distributed, large-bodied herbivores and primates. In contrast, most species occurred at low encounter rates and at only a few stations, consistent with spatially patchy habitat use and/or reduced detectability for rare and cryptic species. Spatial analyses identified discrete multi-hotspots (KDE very high: 62.47–78.01; IDW very high: 14.60–17.99), and the congruence between KDE and IDW supported the robust identification of high-use nodes. Overall, the findings indicate that Kitendeni retains substantial mammal diversity and corridor value within a multi-use landscape. Still, effective management will require maintaining habitat heterogeneity and permeability within hotspot nodes and along connecting movement pathways.

Keywords: *Camera traps, Connectivity, Diversity, Spatial patterns, and Wildlife Corridor*

INTRODUCTION

Wildlife corridors are increasingly recognized as essential ecological linkages that preserve connectivity, promote gene flow, and support the long-term persistence of wildlife populations in fragmented landscapes (Beier & Noss, 1998; Hilty *et al.*, 2020; Hilty *et al.*, 2019a; Hilty *et al.*, 2019b). Evidence syntheses indicate that corridors can measurably increase animal movement between habitat patches, reinforcing their functional role in maintaining connectivity (Gilbert-Norton *et al.*, 2010). In

East African savanna ecosystems, where human activities continue to expand and alter natural habitats, corridors provide crucial pathways for the seasonal and long-distance movements of large mammals (Hilty *et al.*, 2019b).

The Kitendeni Wildlife Corridor, which links Amboseli National Park in Kenya with Kilimanjaro National Park in Tanzania, is a critical ecological corridor within the Greater Kilimanjaro Ecosystem (Kikoti *et al.*, 2010). It supports migration routes used by elephants, ungulates, and predators, while

also maintaining ecological processes crucial for regional biodiversity conservation (Beier & Noss, 1998; Hilty *et al.*, 2019b). Despite its ecological importance, Kitendeni has experienced substantial land-use change in recent decades. Increased livestock grazing, expansion of human settlements, fuelwood collection, and rising pressure on rangelands have contributed to a multifunctional landscape in which habitat conversion and competition for resources can constrain wildlife movement and habitat quality (Noe, 2003; Odadi *et al.*, 2011). Such pressures are widely recognized as risks to corridor functionality and may alter the composition and space use of mammal communities over time (Hilty *et al.*, 2019a; Hilty *et al.*, 2019b). However, locally specific ecological data remain limited, which constrains conservation organizations and planners' ability to make defensible decisions about corridor management and land-use planning (Noe, 2003).

To address this knowledge gap, we deployed camera traps within Kitendeni Wildlife Corridor. Camera trapping has become a valuable tool for monitoring terrestrial mammals in dynamic socio-ecological systems. Its non-invasive design and continuous operation make it well-suited for detecting wide-ranging, nocturnal, and cryptic, as well as for producing standardized datasets across heterogeneous landscapes (Burton *et al.*, 2015; O'Connell *et al.*, 2011). Nevertheless, robust inference from camera-trap detections requires careful attention to sampling design and analytical assumptions, particularly when using detection indices such as the relative abundance index (RAI) that do not explicitly model imperfect detection (Burton *et al.*, 2015; Sollmann *et al.*, 2013). We conducted a 12-month camera-trap survey using 36 passive-infrared cameras deployed across a range of habitat types, including woodlands, grasslands, forests, and shrublands. Our goal was to examine the spatial and ecological patterns of medium- and large-bodied mammals within the corridor and to determine how land-use intensity affects wildlife assemblages. Specifically, the study: (1) measured abundance using the

Relative Abundance Index (RAI); (2) assessed richness across different habitat types and land-use categories; (3) evaluated ecological complexity using the Shannon-Wiener (H') and Simpson's ($1-D$) diversity indices; (4) compared mammalian community structures across areas habitat types; and (5) assess existing spatial clustering. By combining long-term monitoring with established analytical methods, this study provides evidence to assess mammalian community structure within the Kitendeni Wildlife Corridor and to inform connectivity-focused land-use planning and adaptive corridor management.

MATERIAL AND METHODS

Study area

This study was conducted in the Kitendeni Wildlife Corridor (Fig.1), a crucial connectivity zone within the Greater Kilimanjaro Ecosystem of northern Tanzania (Kikoti *et al.*, 2010). The corridor spans a Maasai rangeland situated between Kitendeni village on the western side ($2^{\circ}50'53.50''S$, $37^{\circ}14'38.43''E$) and Irkaswa village on the eastern side ($2^{\circ}51'52.65''S$, $37^{\circ}19'26.75''E$) (Kikoti *et al.*, 2010). Functionally, it provides a transboundary linkage between Amboseli National Park in Kenya and Kilimanjaro National Park in Tanzania (Kikoti & Griffin, 2011), enabling movements of wide-ranging mammals and sustaining connectivity across a landscape increasingly shaped by human land use (Kikoti & Griffin, 2011; Noe, 2003).

The corridor landscape comprises a diverse savanna mosaic of woodland, open grassland, shrubland, riparian strips, and scattered forest patches, structured by local topography, seasonal rainfall patterns, and intensive livestock use (Kikoti *et al.*, 2010; Noe, 2003). Climatic conditions follow a bimodal rainfall pattern typical of northern Tanzania, with long rains commonly occurring from March to May and short rains from October to December, separated by a pronounced dry season during the mid-year months (Røhr & Killingtveit,

2003); these seasonal shifts strongly influence vegetation productivity, forage availability, and wildlife space use.

Pastoralism remains the dominant land use within and surrounding Kitendeni, while smallholder cultivation continues to expand along corridor margins, creating a multi-use setting where livestock and wildlife frequently overlap (Noe, 2003). The corridor spans an elevational range of approximately 1,600–1,750 m above sea level and is characterized by a savanna climate with an estimated mean annual rainfall of about 220 cm (Røhr & Killingtveit, 2003).

Vegetation composition varies along this gradient: upper-elevation woodlands are dominated by *Acacia seyal*, *A. nilotica*, *A. drepanolobium*, and *Balanites aegyptiaca*, whereas lower-elevation areas support shrub communities dominated by *Acacia nubica* and *Commiphora africana* (Kikoti *et al.*, 2010; Røhr & Killingtveit, 2003). Extensive grassland patches are present, particularly on the eastern side of the upper corridor and throughout the lower corridor, where *Themeda triandra*, *Cynodon plectostachyus*, *C. dactylon*, and *Pennisetum stramineum* are prominent. *Lantana* spp. are also widespread in open habitats across the corridor (Kikoti *et al.*, 2010; Røhr & Killingtveit, 2003).

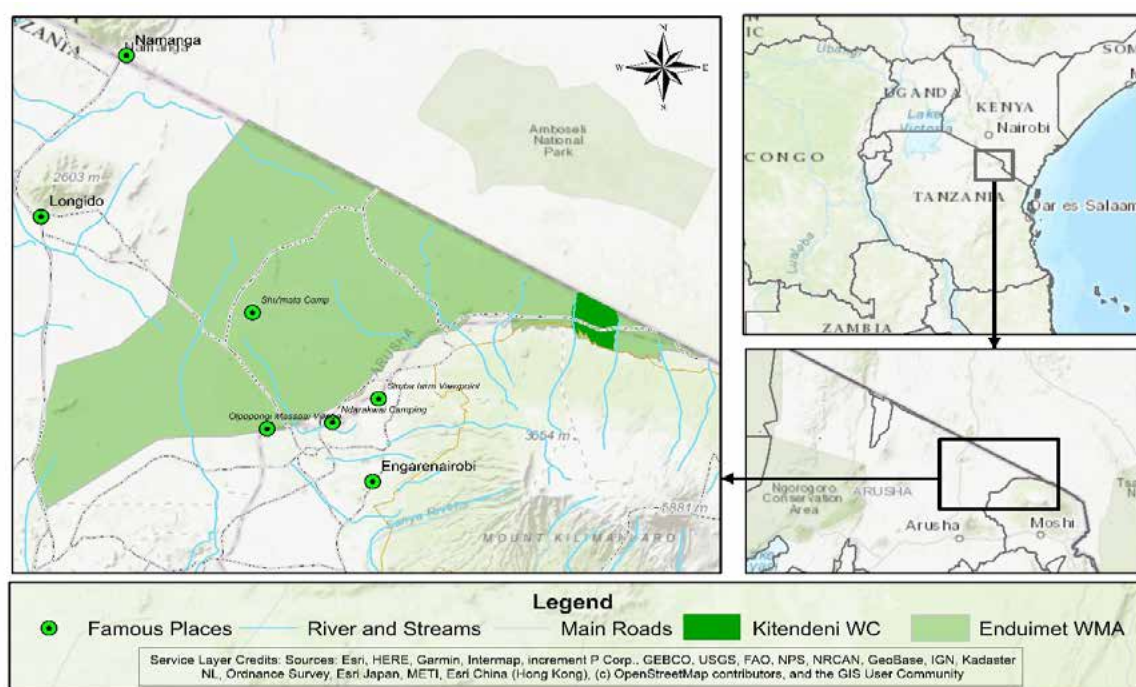


Figure 1: Location of the study area (Kitendeni Wildlife Corridor, northern Tanzania).

Data collection

Camera-trap survey design

A systematic and random sampling approach was adopted to establish camera-trap station locations across the corridor. Systematic spatial coverage is recommended for community-level inference in heterogeneous landscapes,

while the random component reduces placement bias that arises from preferentially selecting trails or high-use features (Kays *et al.*, 2020; Kolowski & Forrester, 2017). This combined design aimed to enhance detection probability for a broad set of medium- and large-bodied mammals while maintaining defensible spatial representativeness (Burton *et al.*, 2015; Wearn & Glover-Kapfer, 2017). A total of 36 passive infrared camera traps

were deployed continuously for 12 months, spanning multiple habitat types (woodlands, grasslands, shrublands, and forest patches). Continuous, long-duration deployments are widely recommended to enhance detectability, mitigate temporal bias, and facilitate seasonal comparisons (Burton *et al.*, 2015; Kays *et al.*, 2020).

Generation of systematic camera-trap station points

Candidate camera-trap stations were generated in a GIS environment using the corridor boundary polygon as the basis. The corridor polygon was imported into ArcGIS (v10.8) to define the sampling frame. A spatially structured distribution of candidate stations was produced to support systematic coverage, and station candidates were generated within the boundary using ArcGIS point-generation utilities (ESRI, 2026). A nominal 1-km station spacing was applied to distribute sampling effort across the corridor, consistent with widely used camera-trapping inventory approaches that seek to balance coverage, independence, and logistical feasibility (Kays *et al.*, 2020; Tobler *et al.*, 2008).

Camera maintenance and field protocol

Cameras were checked every 4-6 weeks to ensure functionality and data continuity. During each maintenance cycle, SD cards were replaced, batteries were checked/replaced as needed, and obstructing vegetation was cleared. Standardized field protocols were employed to ensure consistent camera positioning and minimize disturbance, as recommended to reduce avoidable variation in detection rates associated with deployment differences (Burton *et al.*, 2015; Wearn & Glover-Kapfer, 2017). Camera downtime (e.g., battery failure, theft, malfunction) was documented to support transparent effort correction and interpretation (Burton *et al.*, 2015). Sampling effort was expressed as trap nights, calculated as the number of operational nights between deployment and retrieval for

each unit, excluding documented downtime (Burton *et al.*, 2015).

Image screening, annotation, and verification

After retrieval, the images were downloaded and uploaded to Wildlife Insights, a platform designed to support the processing of camera-trap data and AI-assisted annotation at scale (Ahumada *et al.*, 2020). Automated outputs were treated as preliminary, and all labels were manually verified, reflecting best practice to reduce misclassification risk when integrating machine learning in wildlife monitoring workflows (Burton *et al.*, 2015; Fennell *et al.*, 2022). Where identification uncertainty persisted, taxonomic confirmation was supported by cross-checking profiles against the IUCN Red List. Images that could not be reliably identified due to limitations such as motion blur, poor lighting, or excessive distance were flagged and excluded from -level analyses, consistent with typical constraints reported for passive camera trapping (Burton *et al.*, 2015; Tobler *et al.*, 2008).

Data analysis

Standardization of detections and sampling effort

To reduce pseudo-replication, detections at the same camera station were considered independent events only if separated by at least 30 minutes. This interval-based approach is commonly applied in camera-trap studies of unmarked populations where repeat detections may otherwise inflate encounter rates (Burton *et al.*, 2015; Kays *et al.*, 2020). Sampling effort was expressed as trap nights (TN), calculated as the number of nights each camera operated, excluding documented downtime.

Relative Abundance Index (RAI)

Abundance was indexed using the Relative Abundance Index (RAI), defined as the

number of independent detections per unit of sampling effort. RAI is widely applied in multi-camera-trap studies, particularly when density estimation is not feasible for unmarked populations; however, interpretation requires caution because detectability may vary among habitats and levels of human disturbance (O'Brien, 2011; O'Brien *et al.*, 2003; Sollmann *et al.*, 2013). RAI was computed as:

$$RAI_i = \left(\frac{D_i}{TN}\right) \times 100$$

Where D_i is the number of independent detections for i , and TN is the total number of trap nights. The multiplier (100) represents detections per 100 trap nights, enabling comparisons of relative activity/encounter rates across different habitat types and land-use categories (O'Brien, 2011).

Richness and sampling completeness

Richness was computed as the total number of distinct mammal species detected in each habitat type and land-use category. Sampling completeness was evaluated using accumulation curves and rarefaction, implemented in the vegan package in R (Oksanen *et al.*, 2025). This approach supports more defensible comparisons of richness across groups when sampling effort and detectability may differ (Kays *et al.*, 2020).

Diversity

Ecological diversity was quantified using Shannon–Wiener (H') and Simpson's diversity ($1-D$) indices, which provide complementary information on evenness and dominance patterns across habitat and land-use categories (Hammer *et al.*, 2001). The indices were calculated as:

Shannon–Wiener diversity (H'):

$$H' = - \sum_{i=1}^S p_i \ln(p_i)$$

Simpson's diversity ($1-D$):

$$1 - D = 1 - \sum_{i=1}^S p_i^2$$

Where p_i is the proportion of detections attributable to i , and S is the total number of.

Naïve occupancy (site use)

To describe the spatial extent of occurrence in the sampled landscape, naïve occupancy was calculated as the proportion of operational camera stations at which a was detected at least once:

$$\psi_{naive} = \frac{S_{det}}{S_{active}}$$

Where S_{det} is the number of stations with detections, and S_{active} is the number of stations that were active during the study. This metric provides an intuitive indication of site use but does not account for imperfect detection (MacKenzie *et al.*, 2002).

Spatial clustering of local richness

Station-level taxa richness (i.e., observed species richness) was calculated for each camera-trap station as the total number of distinct mammal taxa detected over the sampling period (Gotelli & Colwell, 2001; Magurran, 2003). The resulting richness values were joined to the camera-station point layer in a GIS and used to generate two complementary spatial surfaces describing local clustering and corridor-wide gradients in richness.

First, kernel density estimation (KDE) was applied to the station points, using station richness as the weighting ("population") field, to produce a smoothed hotspot surface representing the spatial concentration of high-richness stations across the corridor (Silverman, 2018; Worton, 1989). KDE is widely used to summarize the spatial intensity of point phenomena by spreading weighted values across a search radius to generate a continuous surface (Węglarczyk, 2018; Worton, 1989).

Second, inverse distance weighted (IDW) interpolation was used to create a continuous raster of predicted richness from the same station-level values, consistent with the assumption of spatial autocorrelation (i.e., observations tend to be more similar the closer they are in space) (Childs, 2004). IDW assigns greater influence to nearby stations through a distance-decay weighting function controlled by a power parameter (Childs, 2004). Both raster outputs were clipped to the corridor boundary and classified into five ordinal classes (Very Low–Very High) to enable consistent visual comparison of hotspot patterns and richness gradients between methods.

Because KDE and IDW are deterministic/descriptive mapping procedures and do not explicitly model imperfect detection or effort-related heterogeneity inherent to camera-trap data, these surfaces were interpreted as relative indicators of richness hotspots and spatial

gradients rather than as estimates of absolute ecological “density” (Gotelli & Colwell, 2001; Li & Heap, 2014; MacKenzie *et al.*, 2002).

RESULTS

Survey effort

Camera-trap monitoring yielded 10,157 independent detection events from 33 camera stations, with a sampling effort of 3,245 trap-nights. Sampling was distributed across three habitat classes: grassland, shrubland, and tree cover (Table 1). A total of 39 mammal species were recorded as labelled in the dataset; 28 species were identified to the level, whereas 11 records were retained at higher or uncertain taxonomic resolution (e.g., generic “” labels or family-level assignments). Table 1 summarizes the distribution of camera stations, trap nights, and detections across habitat classes.

Table 1. Sampling effort and detection totals by habitat class.

Habitat	Camera stations	Trap-nights	Detections
Grassland	11	1393	3380
Shrubland	14	1453	5044
Tree Cover	8	399	1733

Sampling completeness at the corridor scale was evaluated using a station-based accumulation curve. The accumulation trajectory approached an asymptote by the final stations, converging

on the observed total of 39 (Fig. 2), indicating that the survey captured most species detectable under the prevailing camera placement and operational coverage.

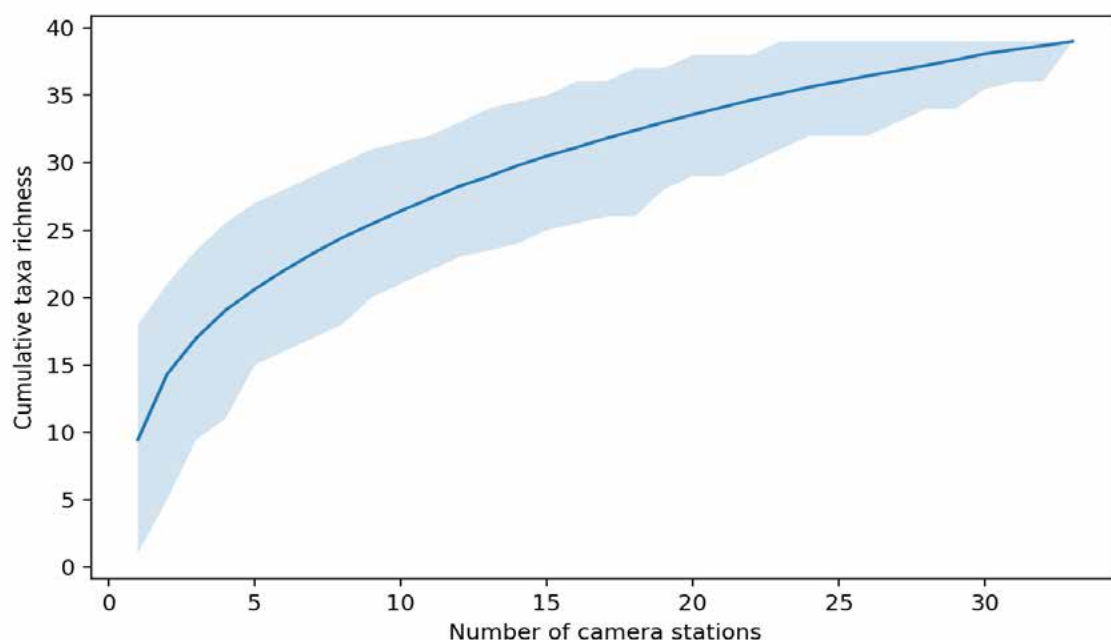


Figure 2. Sample-based species accumulation curve across camera stations (mean with 95% simulation envelope).

These results indicate a wildlife-only assemblage, numerically dominated by large-bodied herbivores, with comparatively fewer detections of carnivores and other low-density or cryptic guilds. Such right-skewed detection structures are typical of camera-trap datasets where encounter rates reflect both true abundance and heterogeneous detectability among species.

Relative abundance and detection structure (RAI)

Habitat-level RAI

Habitat-level encounter rates were indexed using the Relative Abundance Index (RAI), expressed as the number of detection events per 100 trap nights (Table 2). Tree-cover stations exhibited the highest detection-standardised encounter rate (RAI = 434.34 events/100 trap-nights), followed by shrubland (RAI = 347.14) and grassland (RAI = 242.64).

Table 2. Habitat-level trap-nights, detections, and Relative Abundance Index.

Habitat	Trap-nights	Detections	RAI (events/100 trap-nights)
Grassland	1393	3380	242.64
Shrubland	1453	5044	347.14
Tree Cover	399	1733	434.34

These results suggest that, after accounting for effort, detections were concentrated relatively more within tree-cover stations than within shrubland or grassland. However, tree-cover sampling was based on fewer stations and trap-

nights (Table 2), and habitat-level comparisons should therefore be interpreted as encounter-rate contrasts rather than direct evidence of density differences.

Species-level detection structure

At the taxon level, encounter rates were strongly right-skewed, with a small subset of species accounting for most detections (Table 3; Fig. 3). Giraffe was the most frequently recorded taxon (3,772 detections; RAI = 116.24 events/100 trap-nights), contributing 37.14% of all

detections. The three most commonly detected species (giraffe, African buffalo, and common eland) accounted for 69.34% of detections, and the top five species accounted for 84.72%, indicating that corridor-wide encounter rates were dominated by a limited number of widely detected herbivores and primates.

Table 3. Top species by detections, with RAI and naïve occupancy (ψ).

Taxon	Detections	RAI	Naïve occupancy (ψ)	% of detections
Giraffe	3772	116.24	0.939	37.14
African Buffalo	1741	53.65	0.788	17.14
Common Eland	1530	47.15	0.576	15.06
African Elephant	871	26.84	0.636	8.58
Yellow Baboon	691	21.29	0.758	6.80
Common Warthog	464	14.30	0.576	4.57
Common Duiker	307	9.46	0.636	3.02
Impala	157	4.84	0.515	1.55
Bushbuck	129	3.98	0.455	1.27
Spotted Hyaena	83	2.56	0.485	0.82

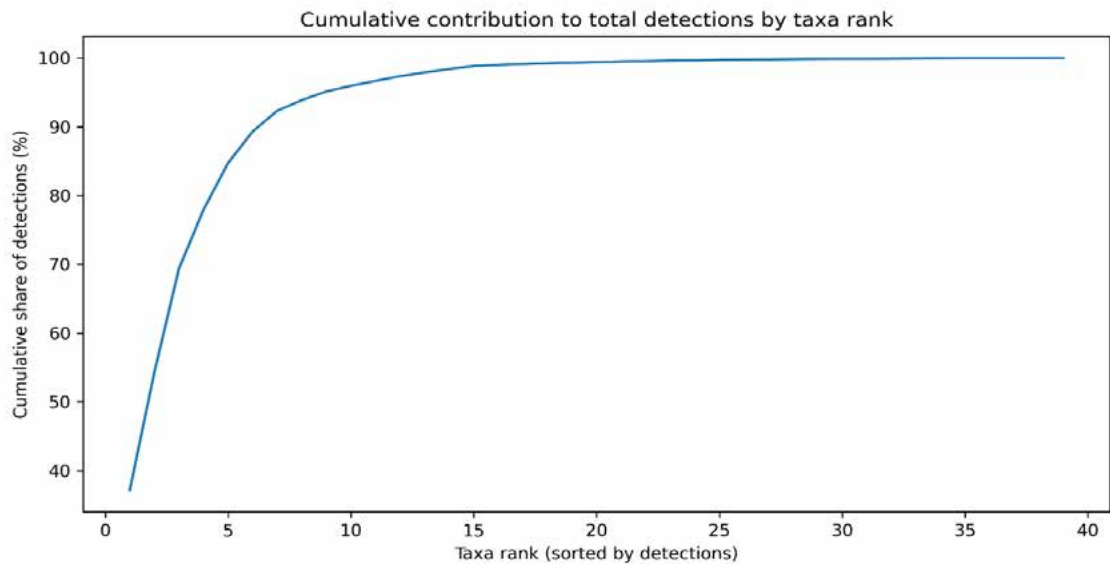


Figure 3. Cumulative contribution of species to total detections, illustrating detection dominance.

The long tail of low-detected species is consistent with a combination of spatially patchy habitat use, low local abundance, and/or reduced detectability for certain guilds (e.g., cryptic carnivores and small-bodied species). Accordingly, species-level RAI values should be interpreted as indices of relative encounter rates rather than direct proxies of density.

Species richness and sampling completeness

Species richness differed among habitat classes (Table 4). Shrubland supported the highest richness (33 species), followed by grassland (25 species), while tree-cover habitats recorded the lowest richness (22 species). These results indicate that shrubland provides the broadest assemblage of mammals within the corridor, consistent with its structural heterogeneity and resource diversity, which can support a broader

range of guilds. In contrast, the lower richness in tree-cover habitats suggests more selective use by a subset of species, potentially reflecting patch size, distribution, and habitat accessibility within the multi-use mosaic.

Species diversity across habitat types

Diversity estimates showed a broadly consistent pattern with richness, but also revealed differences in dominance structure (Table 4; Fig. 4). Shrub land exhibited the highest Shannon diversity ($H' = 2.005$) and high Simpson diversity ($1-D = 0.804$). Grassland ($H' = 1.845$; $1-D = 0.736$) and tree cover ($H' = 1.839$; $1-D = 0.782$) displayed similar Shannon values, although tree cover retained a comparatively higher Simpson index, indicating slightly reduced dominance by a small number of species relative to grassland.

Table 4. Species richness and diversity indices by habitat class.

Habitat	Species richness	Shannon H'	Simpson ($1-D$)
Grassland	25	1.845	0.736
Shrubland	33	2.005	0.804
Tree Cover	22	1.839	0.782

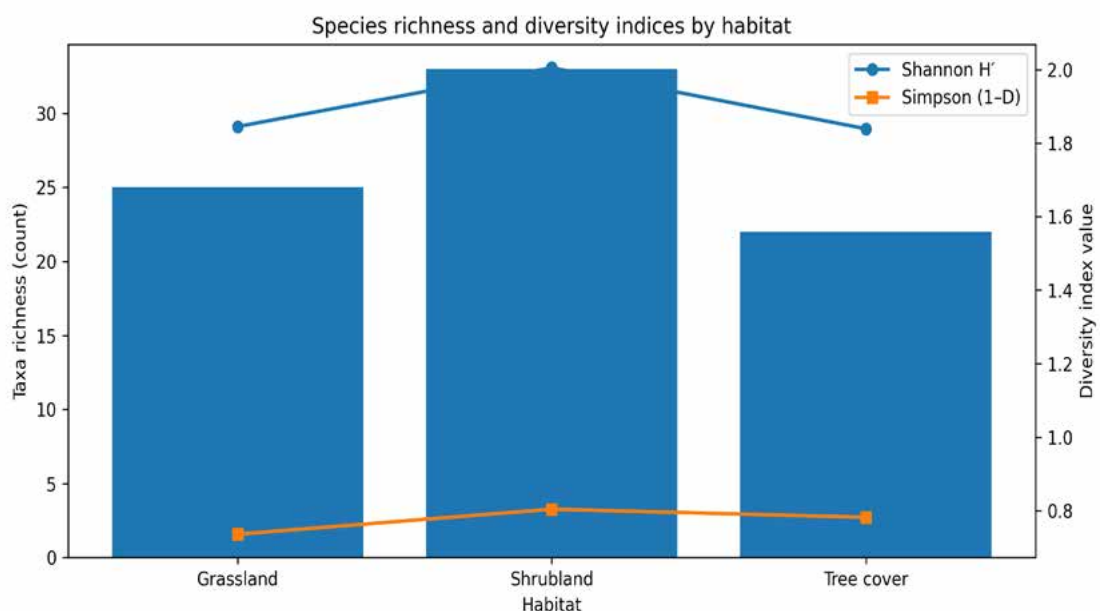


Figure 4: Species richness and diversity indices (Shannon H' and Simpson $1-D$) by habitat class.

These results suggest that shrubland represents the most structurally heterogeneous and resource-diverse habitat, supporting both the broadest assemblage and the most even distribution of detections across species. The lower richness but relatively high diversity in tree-cover habitats implies that fewer species use these patches. Still, those that do occur are more evenly represented, consistent with stable habitat use and/or refuge functions. Conversely, the lower diversity in grassland indicates stronger dominance by open-habitat species, consistent with assemblages structured around grazer-dominated communities.

Community composition across habitat classes

Differences in assemblage composition among habitat classes were evaluated at the camera-station level using Bray–Curtis dissimilarities of taxon detection counts. PERMANOVA indicated weak habitat-associated structuring (pseudo- $F = 1.053$, $R^2 = 0.066$, $p = 0.375$; Table 5), with ordination showing substantial overlap among habitat clusters (Fig. 5).

Table 5. PERMANOVA results for habitat-associated differences in community composition (*Bray-Curtis*).

Factor	Groups (k)	Pseudo-F	R ²	p (permutation)	Permutations
Habitat	3	1.053	0.066	0.375	999

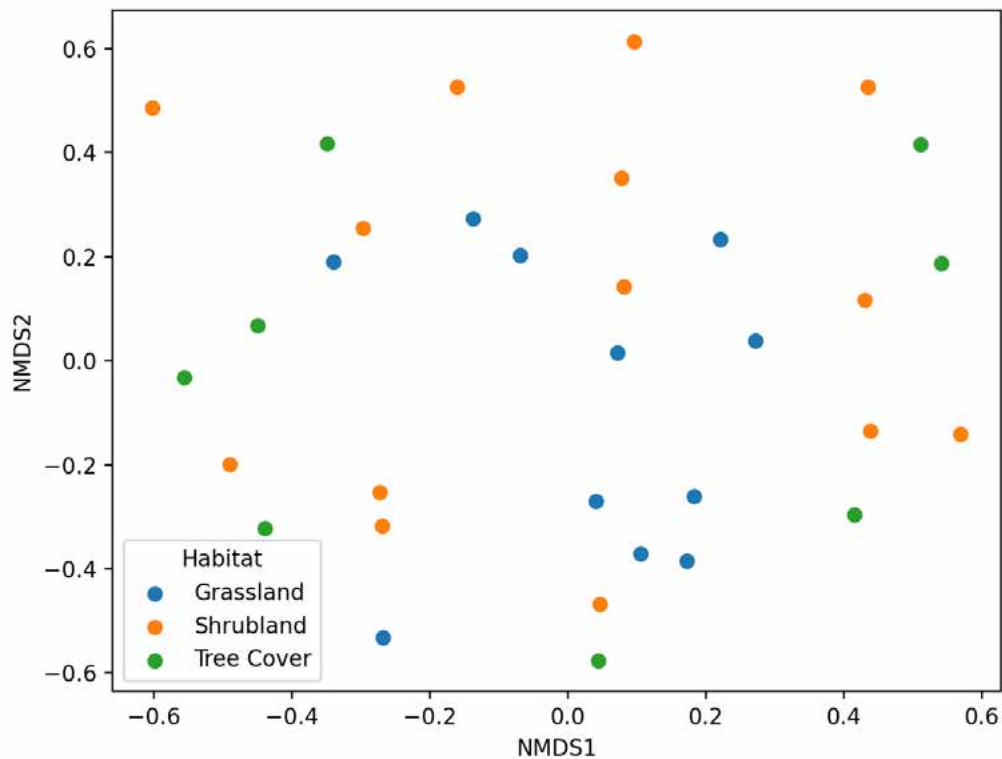


Figure 5: Non-metric multidimensional scaling (NMDS) ordination of camera-station assemblages (Bray-Curtis).

These results suggest that, at the scale captured by camera-station detections, mammal assemblages are broadly shared across the three habitat classes, and that between-habitat differences are modest relative to within-habitat variability. This pattern is consistent with a corridor landscape in which many dominant species are widely distributed and routinely traverse multiple habitat elements.

Site use (naïve occupancy)

Site use was quantified using naïve occupancy (ψ), defined as the proportion of camera stations at which a taxon was detected at least once.

Across the 33 camera stations, ψ ranged from 0.030 to 0.939 (median ≈ 0.091), indicating substantial heterogeneity in site use among species. Giraffe exhibited the highest site use ($\psi = 0.939$; 31 of 33 stations), followed by African buffalo ($\psi = 0.788$) and yellow baboon ($\psi = 0.758$) (Table 6).

A further set of species showed moderate site use ($0.20 \leq \psi < 0.50$), including spotted hyaena, bushbuck, vervet monkey, and dik-dik (-level uncertainty), whereas most species ($\psi < 0.20$) were detected at relatively few stations, consistent with spatially patchy habitat use and/or reduced detectability.

Table 6. Top species by naïve occupancy (ψ) across the camera-station network.

Taxon	Stations (of 33)	Naïve occupancy (ψ)
Giraffe	31 of 33	0.939
African Buffalo	26 of 33	0.788
Common Eland	19 of 33	0.576
African Elephant	21 of 33	0.636
Yellow Baboon	25 of 33	0.758
Common Warthog	19 of 33	0.576
Common Duiker	21 of 33	0.636
Impala	17 of 33	0.515
Bushbuck	15 of 33	0.455
Spotted Hyaena	16 of 33	0.485

Collectively, these results indicate that corridor-wide space use is dominated by a subset of widely distributed species, primarily large-bodied herbivores and primates, while many species exhibit restricted apparent site use. Given that ψ does not correct for imperfect detection, low ψ values should be interpreted cautiously, as they may reflect limited site use, low abundance, or low detectability under the prevailing habitat structure and camera placement.

Spatial clustering of local richness

Mapping station-level species richness revealed evident spatial heterogeneity across the Kitendeni Wildlife Corridor, with both surfaces

identifying non-uniform concentrations of high-richness stations (Fig. 6). The kernel density (KD) surface (Fig. 6 (a)) delineated discrete richness “hotspots,” with the most pronounced Very High cluster (62.47–78.01) occurring in the northwestern portion of the corridor, close to the international boundary, and additional localized High patches (46.93–62.47) distributed within the interior (notably within the central–southern corridor). Much of the remaining corridor fell within the Very Low to Moderate KD classes (0.31–46.93), indicating that high-richness stations were spatially concentrated rather than broadly distributed.

In contrast, the IDW interpolated surface (Fig. 6(b)) showed a smoother corridor-wide gradient, with extensive areas of Moderate predicted richness (7.81–11.20) and a broad High zone (11.20–14.60) spanning large sections of the eastern corridor. The highest

predicted richness (Very High: 14.60–17.99) was concentrated along the eastern to north-eastern margin of the corridor, whereas Very Low to Low values (1.01–7.81) occurred as smaller, discontinuous patches primarily within the central and south-central corridor.

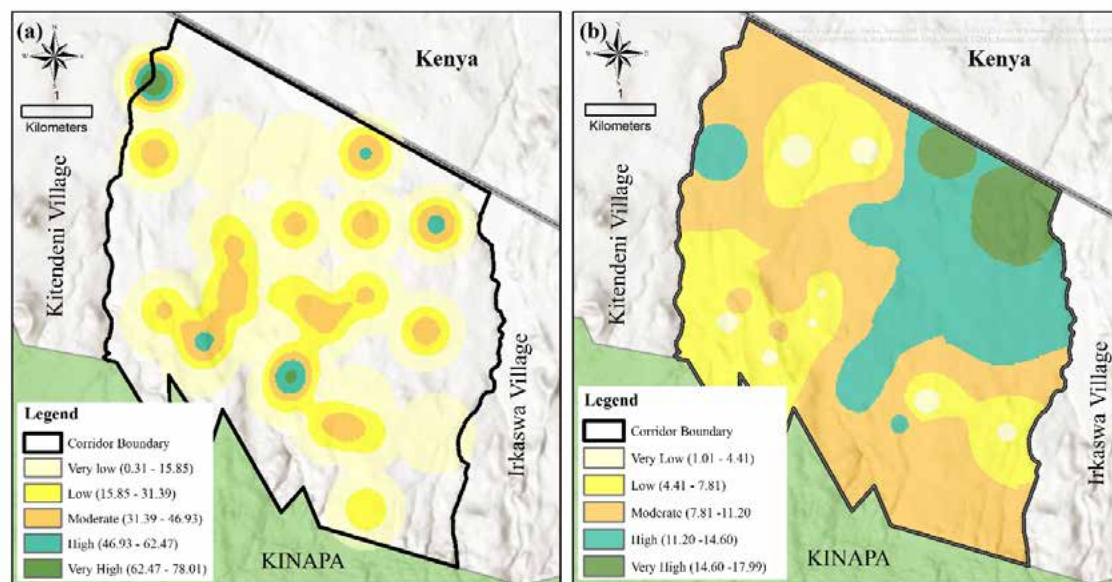


Figure 6. Spatial clustering of station-level species richness across the Kitendeni Wildlife Corridor. Notes: (a) Kernel density hotspot surface (b) IDW-interpolated richness surface

Considered together, the KD and IDW outputs provide consistent evidence that local mammal richness is spatially clustered within Kitendeni, but they emphasise different aspects of that pattern. The KD surface highlights discrete hotspots where high-richness stations are tightly concentrated (notably the strong northwestern hotspot). In contrast, the IDW surface generalises these station-level contrasts into a continuous gradient, suggesting broader zones of comparatively higher richness along the eastern corridor. Because both methods are sensitive to station distribution and do not explicitly account for imperfect detection or unequal effective effort, the mapped hotspots and gradients should be interpreted as descriptive indicators of spatial concentration in station-level richness and used primarily to guide prioritisation of corridor segments for targeted monitoring and management.

DISCUSSION

Community structure and dominance patterns

The camera-trap survey indicates that mammal assemblages in the Kitendeni Wildlife Corridor are characterised by high overall diversity but a strongly uneven detection structure. Across the camera network, encounter rates were concentrated in a small subset of widely distributed large-bodied herbivores and primates. At the same time, most species were recorded at relatively low frequency and at a few stations. Such long-tailed detection distributions are typical in camera-trap datasets and reflect a combination of actual differences in abundance and space use, behavioural responses to cameras, and habitat- and placement-driven variation in detectability

(Burton *et al.*, 2015; Sollmann *et al.*, 2013). Accordingly, dominance by common open-habitat herbivores should be interpreted as a signal of both ecological prevalence and higher detectability in more open settings. In contrast, low encounter rates for rare, cryptic, or disturbance-sensitive species may represent either limited use of the corridor, low densities, or imperfect detection under the prevailing camera placement and vegetation structure (MacKenzie *et al.*, 2002; Sollmann *et al.*, 2013).

Habitat heterogeneity and corridor functioning

Habitat-specific results highlight the role of structural heterogeneity in sustaining corridor biodiversity. Shrubland supported the broadest assemblage and high diversity, consistent with general theory and empirical evidence that heterogeneous habitats provide complementary resources and refuge features, thereby increasing the number of niches available to wildlife (Tews *et al.*, 2004). Tree-cover patches supported fewer species overall but retained relatively high diversity, suggesting that although fewer species use these patches regularly, those that do may do so more consistently and without extreme dominance by a single taxon. In contrast, grassland assemblages showed stronger dominance by a limited number of frequently detected species, consistent with communities structured around open-habitat guilds and potentially influenced by grazing intensity and human activity. Together, these patterns suggest that maintaining a mosaic of shrubland and tree-cover elements within the broader grassland matrix is likely crucial for sustaining functional connectivity and providing continuity of cover and resources along movement pathways.

Spatial clustering of multiple hotspots

Spatial analyses based on the number of species detected per camera station revealed discrete hotspots of multi-clustering that were broadly consistent between the two mapping

approaches. Kernel density estimation (Fig. 6(a)) showed very high clustering values (62.47–78.01) concentrated near the northern and northwestern portions of the corridor, with additional high (46.93–62.47) and moderate (31.39–46.93) kernels distributed across the interior. The IDW-interpolated surface (Fig. 6 (b)) exhibited a concordant gradient, with very high interpolated values (14.60–17.99) occurring in broadly similar areas, and extensive moderate-to-high zones (7.81–14.60) extending across the central parts of the corridor. Convergence between KDE and IDW strengthens the inference that these areas represent robust nodes of elevated site-level richness, rather than artifacts of a single mapping method. Ecologically, such hotspots plausibly reflect spatial convergence of habitat quality (e.g., cover–forage juxtaposition), movement funnels, and localised reductions in disturbance, which together generate repeated detections across multiple species.

Implications for corridor management and monitoring

From a connectivity perspective, the co-occurrence of broadly distributed species and spatially concentrated multi-hotspots supports a view of corridor function that depends on both permeable matrix conditions and a network of high-use nodes. Reviews and syntheses indicate that corridors can enhance movement between habitat patches, although efficacy varies with species and landscape context (Beier & Noss, 1998; Gilbert-Norton *et al.*, 2010; Hilty *et al.*, 2019b). In Kitendeni, hotspot areas identified by KDE and IDW likely represent priority nodes where maintaining habitat integrity and reducing localised pressure will yield disproportionate connectivity benefits. At the same time, inferences from RAI and naïve occupancy should be made cautiously because these metrics do not account for imperfect and variable detection (MacKenzie *et al.*, 2002; Sollmann *et al.*, 2013). Future monitoring would be strengthened by combining consistent placement protocols with detection-corrected occupancy or spatial models for selected

species, thereby improving comparability across seasons and years and increasing the defensibility of management decisions.

The workflow integrating Wildlife Insights automated classification with expert verification aligns with recommended practice in applying machine learning to camera-trap monitoring, in which automated labels are validated to reduce misclassification risk (Ahumada *et al.*, 2020; Beery *et al.*, 2019). Overall, the results indicate that Kitendeni retains substantial biodiversity value and corridor functionality; however, conservation outcomes will depend on safeguarding habitat heterogeneity and maintaining permeability across key hotspot nodes and the linkages that connect them.

CONCLUSIONS

The Kitendeni Wildlife Corridor supports a diverse assemblage of medium- and large-bodied mammals within a pastoral rangeland matrix, but biodiversity and site use are unevenly distributed across habitats and space. Shrublands emerged as a key habitat for high richness and even detection rates, while grasslands displayed stronger dominance by a limited subset of frequently detected species. Spatial clustering analyses identified discrete multi-hotspots that likely represent critical nodes in the corridor. These results demonstrate the utility of year-long camera-trap monitoring for detecting habitat-associated differences in community structure and for identifying spatial priorities for adaptive corridor management.

RECOMMENDATIONS

- i. Prioritize the protection and stewardship of multi-hotspot areas identified by KDE and IDW, including limiting conversion and intensive disturbance within these areas.
- ii. Maintain and restore functional linkages among hotspot nodes by safeguarding movement pathways and reducing barriers across the corridor.
- iii. Continue standardised long-term camera-trap monitoring with consistent placement

protocols and periodic evaluation of detection bias (e.g., trail versus off-trail placements).

- iv. Where feasible, complement RAI and naïve occupancy with detection-corrected occupancy and/or spatial modeling approaches for focal species to strengthen inference for rare or cryptic species.

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SUB-THEME: WATER RESOURCES AND WETLAND CONSERVATION

THREATS, PERCEIVED WETLAND HEALTH, AND DRIVERS OF COMMUNITY PARTICIPATION IN CONSERVATION OF THE MARA RIVER WETLAND, TANZANIA

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ABSTRACT

The Mara River Wetland (MRW) in northwestern Tanzania is a critical ecosystem that provides ecosystem services including provisioning, regulating, supporting and cultural services, yet it faces mounting pressures from population growth, agriculture, grazing, and unsustainable resource use. Conservation success in such human-dominated landscapes requires not only ecological management but also an understanding of community awareness, perceptions, and willingness to participate in conservation. This study assessed community awareness of wetland threats, perceived changes, and willingness to participate in conservation initiatives. A structured questionnaire was administered to 123 respondents across four villages. The collected data were analyzed using descriptive statistics, chi-square tests, thematic coding of qualitative responses and binary logistic regression. Results indicated that invasive species (74.8%) and pollution (61.8%) were the most commonly perceived threats, followed by climate change impacts (53.7%) and agricultural expansion (38.2%). Significant variation across villages was observed for pollution, loss of riparian vegetation, climate change, agricultural expansion, and overgrazing, whereas perceptions of invasive species and overfishing were consistent. A majority (80.5%) reported noticing changes in the wetland, including water level fluctuations, wetland expansion, fish population decline, and the spread of invasive species, with observed changes negatively associated with age, farming occupation, and participation in recreation activities. Regarding conservation engagement, 84.6% of respondents expressed willingness to participate, 10.6% were unwilling, and 4.9% were undecided. Respondents highlighted training and education (74.0%), tools and equipment (47.2%), financial support (32.5%), and enhanced community collaboration (30.1%) as key enablers. Logistic regression revealed that gender and age significantly influenced willingness: females were less likely than males to participate ($\beta = -1.799$, $p = 0.011$, OR = 0.166), and respondents aged 31-45 years were less willing than those aged 18-30 years ($\beta = -2.117$, $p = 0.019$, OR = 0.120). These findings highlight strong community readiness to engage in wetland conservation, particularly when support mechanisms such as education, resources, and collaborative initiatives are provided. Targeted interventions considering age and gender differences can enhance effective community participation and sustainable management of the Mara River Wetland.

Key words: Mara River Wetland; Conservation participation; Community awareness; Wetland threats; Wetland health.

INTRODUCTION

Wetlands are widely recognized as some of the most productive ecosystems on Earth, supporting water security, biodiversity conservation, climate regulation, and human livelihoods. They supply a broad spectrum of ecosystem services, including provisioning services, regulating services, supporting services and cultural values (Costanza *et al.*, 1997; Lalika *et al.*, 2014; Lalika *et al.*, 2017). Despite their immense value, the socioeconomic and ecological significance of wetlands has made them highly vulnerable to degradation. Evidence from global assessments, including the Millennium Ecosystem Assessment, indicates that nearly 60% of ecosystem services are under unsustainable use (MEA, 2005). In both global and regional contexts, wetland loss is occurring at alarming rates, often exceeding the decline of other ecosystems such as forests (Davidson, 2014; Kingsford *et al.*, 2016). The main drivers of this degradation include anthropogenic activities such as unsustainable agriculture, excessive harvesting of natural resources, mining, overgrazing, and pollution, as well as natural stressors like climate variability and change (Kundu *et al.*, 2024; Osoro *et al.*, 2025). In subSaharan Africa, these pressures are further intensified by rapid population growth, agricultural expansion, and infrastructure development, undermining the longterm ability of wetlands to sustain the livelihoods and welfare of both present and future generations (Simaika *et al.*, 2021; Mumuni *et al.*, 2025).

The Mara River Wetland (MRW), located at the outflow of the Mara River into Lake Victoria in northern Tanzania, represents a critical socio-ecological system within the transboundary Mara River Basin shared by Kenya and Tanzania. The basin traverses the Serengeti-Maasai Mara ecosystem, a globally

significant landscape that sustains the annual migration of millions of wildebeest and other wildlife, while also supporting nearly one million people whose livelihoods depend on agriculture, fishing, livestock grazing, and trade (McClain *et al.*, 2014; Dutton *et al.*, 2019). Unlike many wetlands globally that are shrinking, the Mara Wetland has been reported to be expanding, largely as a result of excessive sediment deposition (Mati *et al.*, 2008; Bregoli *et al.*, 2019). Studies attribute this expansion to land use changes in the upper catchment, unregulated livestock grazing, agricultural intensification, and loss of riparian vegetation, all of which accelerate soil erosion and increase sediment delivery to the river (Mati *et al.*, 2008; Matano *et al.*, 2015; Dutton *et al.*, 2019). Suspended sediment concentrations as high as 4 g L⁻¹ have been recorded, highlighting the magnitude of this issue (Bregoli *et al.*, 2019). Mining activities, including both artisanal operations and large-scale developments such as the North Mara mine, have also contributed to sediment influx through direct discharges, spoil erosion, and accidental releases, although comprehensive data remain scarce. Further, the region has experienced a doubling of its population over the past four decades, coupled with unsustainable practices such as charcoal production, wood extraction, and poorly managed agriculture, continues to exert pressure on the wetland and its catchment (Århem & Fredén, 2014). Consequently, while the Mara Wetland plays a vital role in buffering Lake Victoria, sustaining biodiversity, and supporting local livelihoods, it faces mounting threats from both anthropogenic pressures and natural variability. Understanding these dynamics is essential for informing conservation and management strategies that align ecological sustainability with community needs.

Conservation effectiveness in human-dominated wetlands such as the Mara River Wetland depends not only on biophysical interventions but also on the degree to which local communities are aware of ecological changes, perceive risks, and are willing to

engage in stewardship (Macharia *et al.*, 2010; Owiti *et al.*, 2024; Nyumah & Brambilla, 2025). Insights from environmental psychology and conservation social science demonstrate that awareness influences attitudes, norms, and behaviors through frameworks such as the Knowledge Attitude Practice model and the Theory of Planned Behavior (Mastrangelo *et al.*, 2014; Arundati *et al.*, 2020; Delaroche, 2020). In practice, higher awareness of wetland values and threats can translate into greater participation in conservation actions such as riparian restoration, controlled grazing, sustainable harvesting, or community monitoring. Understanding these interrelated drivers is therefore essential for developing locally legitimate and durable management strategies in the Mara River Wetland.

This study aims to assess community awareness of threats to the Mara River Wetland, the perceived impacts on local livelihoods, and the willingness of community members to engage in conservation. Specifically, the study addressed the following questions: (i) What are the most severe threats to the Mara River Wetland as perceived by local communities, and how do these threats relate to trends in livelihood activities such as farming, fishing, and grazing over the past five years? (ii) How do community members' perceptions of wetland health including aspects such as fish stocks, water quality, and habitat integrity align with observed threats, and which conservation measures do they prioritize? (iii) What factors, including awareness levels, livelihood dependence, institutional support, and access to alternatives, influence community willingness to participate in wetland conservation initiatives and how? By addressing these questions, the

study contributes to a better understanding of the human dimensions of wetland degradation and provides insights to guide effective, community-centered conservation strategies in the Mara River Wetland.

MATERIALS AND METHODS

Study area

The Mara River Wetland is situated in the northwestern part of Tanzania, forming part of the greater Mara River Basin (fig.1). This wetland lies along the lower stretches of the Mara River as it approaches Lake Victoria, and is located near the Serengeti ecosystem to the southeast, further increasing its ecological significance. It is within the administrative boundaries of Serengeti, Butiama, Tarime and Rorya District in the Mara Region, covering an estimated area of 204.46 km². It spans approximately 36.8 km in length and reaches a maximum width of 12.9 km (Muraza *et al.*, 2013; Kazimoto & Ikingura, 2014). The MRW lies between longitudes 34°00' and 34°25' East and latitudes 1°08' and 1°39' South. The area is characterized by a bimodal rainfall regime, with long rains occurring between March and May and short rains from October to December, yielding an annual average of 500-800 mm (Mayo *et al.*, 2013). The Mara River wetland displays a non-equilibrium hydrological response, where reduced discharge and stream power in downstream sections contribute to channel breakdown (Gomes *et al.*, 2023). Vegetation is dominated by *Cyperus papyrus*, *Typha domingensis*, and *Phragmites australis*, with *C. papyrus* contributing the greatest biomass at 4.3 kg DW m⁻² (Muraza *et al.*, 2013). Together, these vegetation dynamics and geomorphological processes play a critical role in sustaining the wetland's ecological integrity and buffering Lake Victoria against excessive nutrient inflows

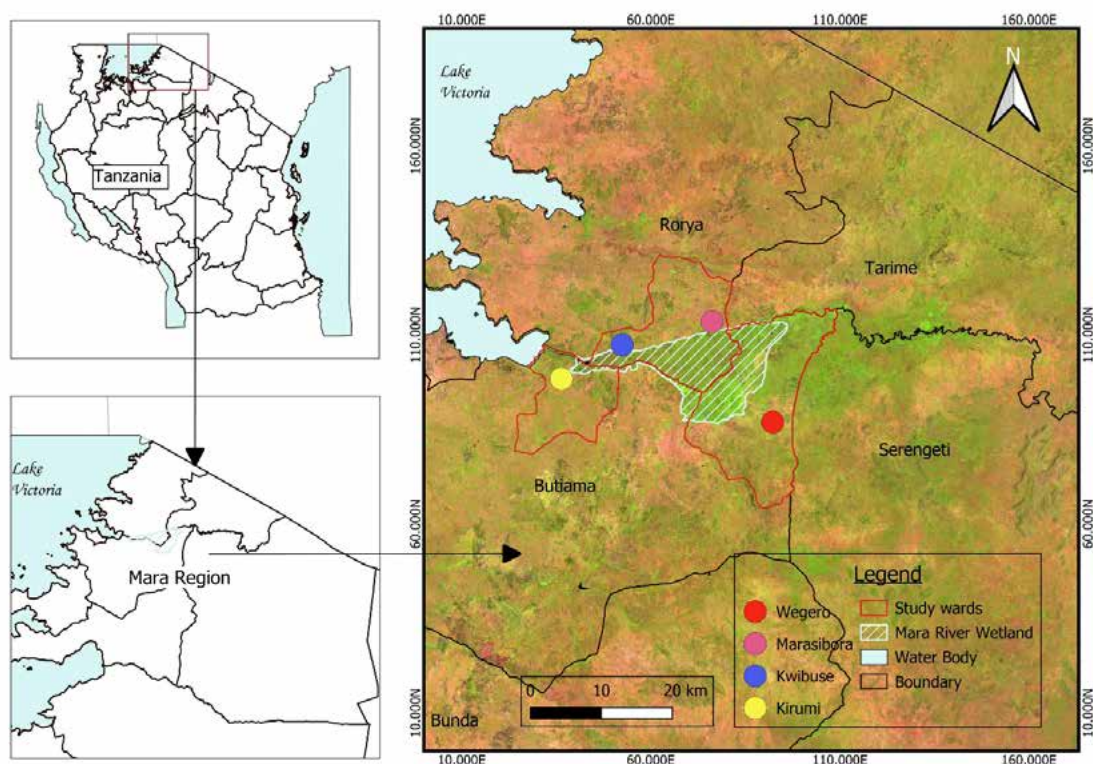


Figure 1: Map of study area showing the Mara River Wetland and surrounding Villages where data were collected.

The wetland falls under the jurisdiction of Lake Victoria basin commission and local government authority, with 16 surrounding villages namely: Bisarwi, Kwibuse, Nyamerambaro, Nkerege, Kembwi Marasibora, Nyanchabakenye and Surubu; and the remaining 8 namely: Kongoto, Buswahili, Kirumi, Wegero, Ryamisanga, Kwisaro, Kitasakwa and Bukabwa lie on the southern side (Omolo *et al.*, 2018). This study covered four villages: Wegero, Kirumi, Kwibuse and Marasibora in Butiama and Rorya districts respectively (Fig. 1).

Sampling Design

This study adopted a cross-sectional research design, which allows the collection of data at a single point in time to capture community awareness, perceptions, and willingness to participate in conservation. Field-based techniques were employed to collect primary data for this study. Households served as the basic unit of analysis, as they represent direct users

of wetland resources and are well positioned to observe and report on environmental changes over time (Kajembe, 1994; Ntongani *et al.*, 2014). At the outset, village registers were used to obtain lists of household heads, which served as the sampling frame. From these lists, simple random sampling was applied to select households for interview. A minimum of 30 households per village was targeted, consistent with recommendations that thirty respondents per site constitute a practical minimum sample size for field research (Saunders *et al.*, 2007; Bailey, 1998). Previous socio-economic studies in sub-Saharan Africa recommend sample sizes ranging between 80 and 120 households to allow meaningful statistical inferences (McClanahan *et al.*, 2005; Jew & Bonnington, 2011). Guided by these benchmarks, a total of 123 households were randomly selected and interviewed across four villages in the Mara River Wetland. This sample size was considered adequate to capture diverse community perspectives and to support reliable analysis of the research questions.

Data Collection

Data were collected using a semi-structured questionnaire administered through face-to-face interviews with household heads or their representatives. The questionnaire was designed to address three core areas aligned with the study objectives: 1) Perceived threats to the Mara River Wetland and their relationship to livelihood activities (farming, fishing, grazing). 2) Community perceptions of wetland health, including changes in fish stocks, water quality, and vegetation, as well as prioritized conservation measures. 3) Willingness to engage in conservation, including factors such as awareness, dependence on wetland resources, activities in which respondents were involved and access to institutional or external support. The questionnaire included both closed- and open-ended questions, enabling the collection of quantitative data while also capturing nuanced qualitative insights.

Data Analysis

Both quantitative and qualitative approaches were employed in the analysis. Quantitative data derived from the household questionnaire were coded and entered into the Statistical Package for Social Sciences (SPSS) version 27 for processing. Descriptive statistics such as frequencies, percentages, and cross-tabulations were used to summarize responses and reveal patterns in community perceptions of wetland threats, ecosystem health, and conservation priorities. Responses from open-ended questions were first examined qualitatively, then systematically coded into thematic categories. These coded responses were subsequently quantified and analyzed alongside the structured questionnaire data. This process ensured that both narrative insights and measurable trends

were integrated to comprehensively address the study objectives. Inferential statistics were also applied to test associations between socio-demographic factors and conservation-related variables. Chi-square tests were used to assess the influence of demographic factors (age, gender, education, and livelihood dependence) on community awareness of threats and willingness to engage in wetland conservation activities. In addition, binary logistic regression analysis was conducted to examine the socio-demographic and livelihood factors influencing willingness to participate in wetland conservation.

RESULTS

Socio-demographic Characteristics of Respondents

A total of 123 household respondents were interviewed across the four study villages. The majority of respondents were male (74.8%), while females accounted for 25.2% of the sample. Most respondents were between the ages of 31-45 years (35.8%) and 46-60 years (32.5%), with younger participants (18-30 years) making up 18.7% and those above 60 years representing 13.0%. Education levels were generally low, with 76.4% having completed only primary education, while 20.3% had secondary education, and 0.8% reported college or university education. In terms of occupation, farming dominated as the primary livelihood (91.9%), followed by fishing (38.2%) and livestock keeping (26.0%), while a smaller proportion engaged in trading (11.4%) and formal employment (2.4%). Respondents were almost evenly distributed across the four study villages: Kirumi (25.2%), Kwibuse (25.2%), Marasibora (25.2%), and Wegero (24.4%) (Table 1).

Table .1 Social demographic characteristics of respondents

Variable	Frequency (N)	Percent (%)
Age		
31-45	44	35.8
46-60	40	32.5
18-30	23	18.7
Over 60	16	13.0
Gender		
Male	92	74.8
Female	31	25.2
Village/Location		
Kirumi	31	25.2
Kwibuse	31	25.2
Marasibora	31	25.2
Wegero	30	24.4
Education level		
Primary education	94	76.4
Secondary education	25	20.3
No formal education	2	1.6
University	1	0.8
College education	1	0.8
Occupation		
Farmer	113	91.9
Fishing	47	38.2
Livestock keepers	32	26.0
Trader	14	11.4
Government Employee	3	2.4

Community-Perceived Threats and Livelihood Trends

Major Perceived Threats

Respondents identified invasive species (particularly water hyacinth, 74.8%) and pollution from chemical runoff and waste dumping (61.8%) as the most common threats to the Mara River Wetland. Other frequently cited issues included climate change impacts such as droughts and floods (53.7%) and agricultural expansion (38.2%). When asked

to rank the most severe threat, invasive species (30.1%) and pollution (27.7%) emerged as the leading concerns, followed by climate change (17.9%). (Table 2)

Chi-square analysis revealed significant variation across villages for several threats. Concerns about pollution ($\chi^2 = 40.550$, $df = 3$, $p < 0.001$), loss of riparian vegetation ($\chi^2 = 13.797$, $df = 3$, $p = 0.003$), climate change ($\chi^2 = 25.138$, $df = 3$, $p < 0.001$), agricultural expansion ($\chi^2 = 38.911$, $df = 3$, $p < 0.001$), and overgrazing ($\chi^2 = 16.116$, $df = 3$, $p = 0.001$)

differed significantly by village. In contrast, overfishing ($\chi^2 = 7.358$, $df = 3$, $p = 0.061$), invasive species ($\chi^2 = 2.760$, $df = 3$, $p = 0.430$), and illegal sand mining ($\chi^2 = 6.034$, $df = 3$, $p = 0.110$) showed no significant variation among villages.

Lack of effective regulation (41.5%) and natural drivers such as climate variability (47.2%) were reported as the main causes of these threats. In terms of livelihood impacts, most respondents linked wetland degradation to reduced fish catch (81.3%), poor water quality (35.8%), and loss of biodiversity (41.5%).

Table 2: Community perceptions of major threats to the Mara River Wetland, their severity, causes, and associated livelihood impacts (N = 123).

Variable	Frequency (N)	Percent (%)
Threats/Impacts Identified		
Invasive species (Water hyacinth)	92	74.8
Pollution (chemical runoff, waste dumping)	76	61.8
Climate change (drought, flooding)	66	53.7
Agriculture expansion	47	38.2
Overgrazing	24	19.5
Loss of riparian vegetation	22	17.9
Overfishing	8	6.5
Illegal fishing	4	3.3
Illegal sand mining	2	1.6
The most severe threats on the wetland		
Invasive species (Water hyacinth)	37	30.1
Pollution	34	27.7
Climate change (drought, flooding)	22	17.9
Agriculture expansion	16	13.0
Others	9	7.3
Loss of riparian vegetation	3	2.4
Overfishing	2	1.6
Main causes of threats		
Natural causes	58	47.2
Lack of regulation	51	41.5
Community practices	31	25.2
Industrial activities	14	11.4
Effects of threats in daily livelihood		
Reduced fish catch	100	81.3
Loss of biodiversity	51	41.5
Poor water quality	44	35.8
Loss of farmland	32	26.0

Livelihood Activities in the Wetland

The wetland serves as a crucial livelihood base, with respondents reporting engagement in farming (61.0%), fishing (43.9%), and livestock grazing (35.0%). A smaller proportion reported activities such as trading (19.5%) and tourism (1.6%). (Fig. 2)

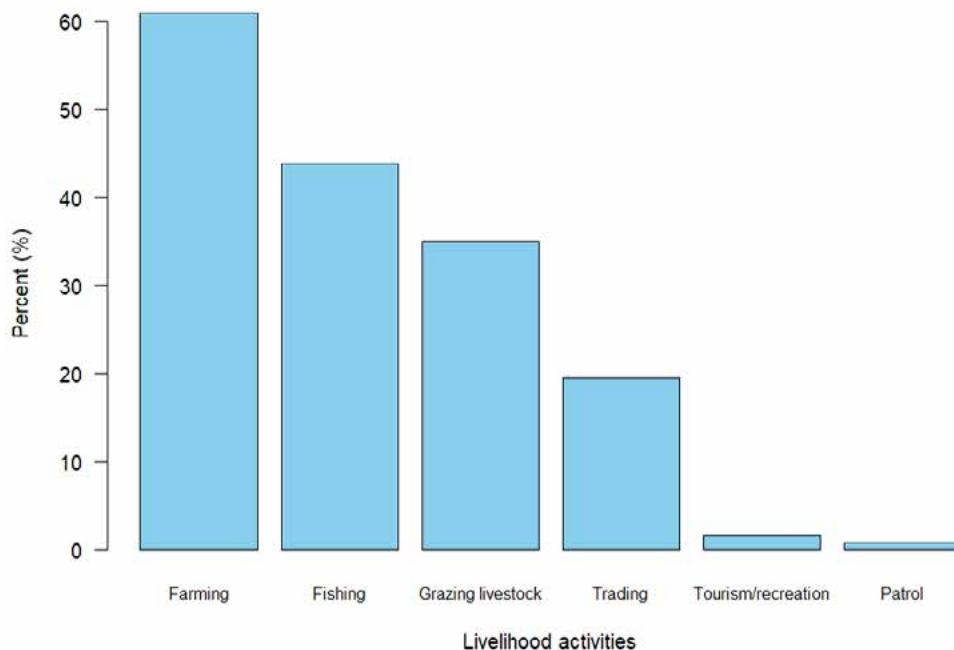


Figure.2 Livelihood activities of local communities in the Mara River Wetland

Trends in Fishing

Fishing activities were perceived to have declined over the past five years, with 44.7% of respondents noting reduced fishing intensity and 76.4% reporting a decrease in the number of fish species caught. Only a minority perceived increases in fishing activity (32.5%) or fish species diversity (12.2%).

Trends in Farming

Farming was reported to have increased substantially in the past five years (68.3%), with

maize (79.7%) and horticultural crops (78.9%) being the most common. Irrigation farming (43.9%) was cited more frequently than rain-fed farming (22.8%), indicating a shift towards more intensive agricultural practices.

Trends in Livestock Keeping

Livestock keeping showed mixed trends, with 47.2% of respondents observing an increase, 29.3% reporting stability, and 21.1% perceiving a decline. Cattle (41.5%) were the most commonly grazed animals, followed by shoats (18.7%).(Fig. 3)

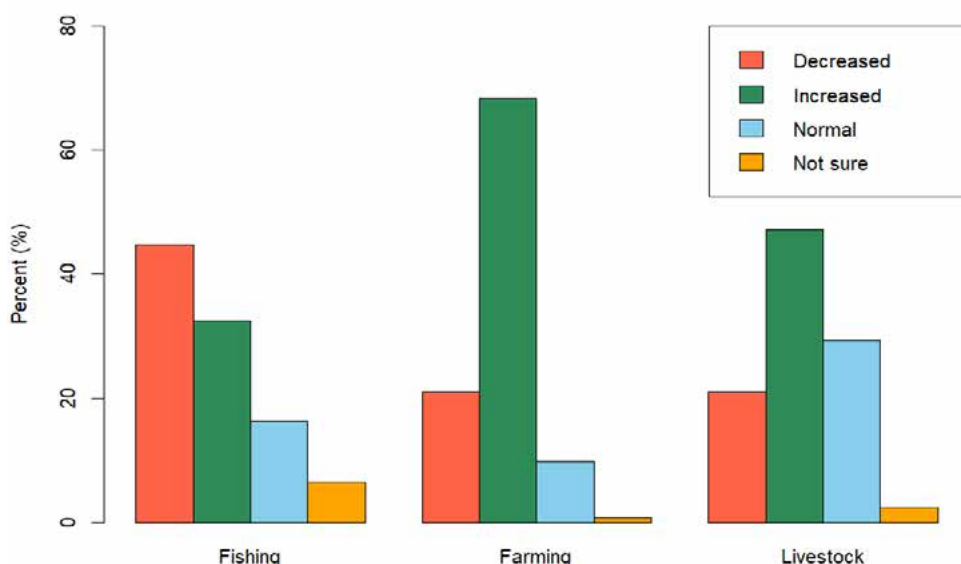


Figure 3: Trends in livelihood activities (fishing, farming, and livestock keeping) in the Mara River Wetland over the past five years. The clustered bar graph illustrates community perceptions of whether each activity has increased, decreased, normal or they are not sure whether the activity has changed.

Community Perceptions of Wetland Health

General Perceptions of Change

A large proportion of respondents (80.5%) reported having noticed changes in the Mara River wetland over the past years, while 9.8% had not observed any changes and an equal proportion (9.8%) were unsure. The most commonly cited changes included water level fluctuations (20.3%), wetland expansion (20.3%), fish population decline (19.5%), and the spread of invasive species (water hyacinth) (17.1%). Other perceived changes included vegetation alterations (12.2%) and

human pressures (10.6%). (Table 4). Chi-square analysis indicated that perceptions of wetland changes did not vary significantly across villages ($\chi^2 = 2.835$, $df = 6$, $p = 0.829$), gender ($\chi^2 = 4.287$, $df = 2$, $p = 0.117$), education level ($\chi^2 = 8.033$, $df = 6$, $p = 0.236$), or original village ($\chi^2 = 2.035$, $df = 2$, $p = 0.361$). However, age ($\chi^2 = 15.938$, $df = 6$, $p = 0.014$), occupation as a farmer ($\chi^2 = 11.362$, $df = 2$, $p = 0.003$), and participation in tourism/recreation activities ($\chi^2 = 18.806$, $df = 2$, $p = 0.000$) were significantly associated with noticing changes in the wetland. (Table 3)

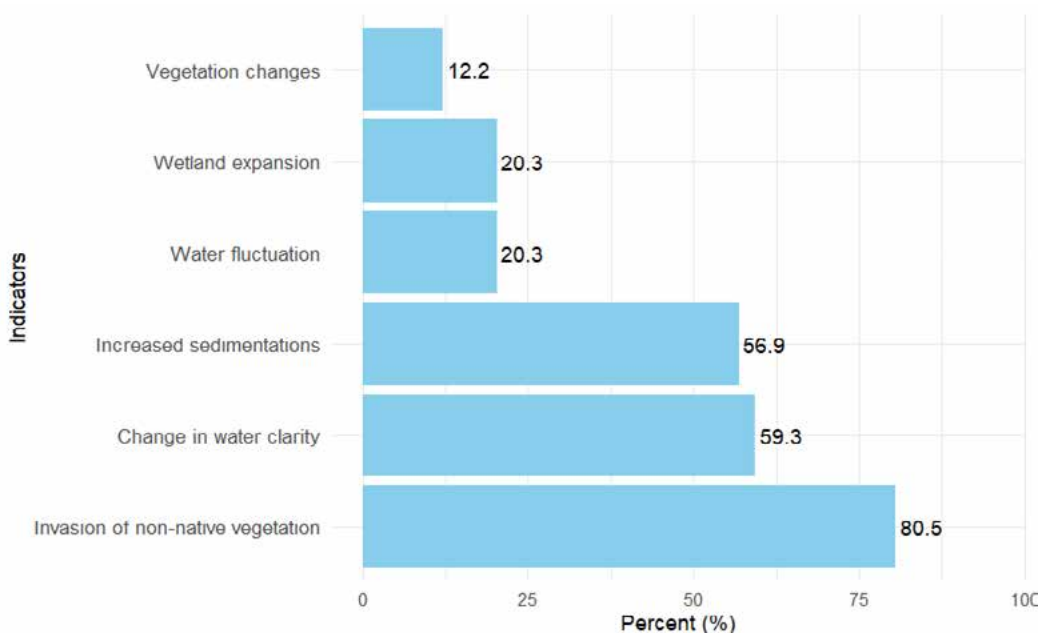
Table 4: Community Perceived Changes in the Wetland over the past years

Variable	Frequency (N)	Percent (%)
Community Perception on Changes in the Wetland		
Yes	99	80.5
No	12	9.8
Not sure	12	9.8
Specific changes		
Water fluctuation	25	20.3
Wetland expansion	25	20.3
Fish decline	24	19.5
Invasive species	21	17.1
Human pressure	13	10.6
Vegetation changes	15	12.2

Perceived change in Environmental Indicators

When asked about specific ecological conditions, the majority of community members reported notable changes in water quality

and habitat integrity. About 59.3% observed reduced water clarity, while 56.9% indicated increased sedimentation. Further, invasion by non-native vegetation (water hyacinth) was highlighted by 80.5% of respondents, and 20.3% noted expansion of wetland (Fig. 4)

**Figure 4.** Community Perceived change in Environmental Indicators

Perceived Wetland Health Status

Overall, most respondents (69.9%) described the wetland as “not good,” while only 11.4% rated it as “good,” and 17.9% as “fair.” In terms of temporal comparisons, 85.4% perceived the wetland’s health to have declined over the past years, while 10.6% felt there had been no change and only 4.1% observed improvement. Community members attributed the decline in

wetland health to multiple factors. The most frequently cited drivers were pollution from farms (48.0%), pollution from mining activities (27.6%), reduced water flow (22.0%), and the spread of invasive species (12.2%). Other less commonly mentioned factors included climate change (5.7%), domestic sewage (4.9%), illegal fishing (3.3%), and overgrazing (1.6%). (Table 5)

Table 5. Community Perceptions of Wetland Health Status, Trends, and Causes of Change

Variable	Frequency (N)	Percent (%)
Community Perceptions		
Not good	86	69.9
Fair	22	17.9
Good	14	11.4
Very good	1	0.8
Wetland Health Status		
Declined	105	85.4
No changes	13	10.6
Improved	5	4.1
Causes for changes in wetland health		
Pollution from farms	59	48.0
Pollution from Mining	34	27.6
Reduced water flow	27	22.0
Invasive species	15	12.2
Climate change	7	5.7
Pollution from domestic sewages	6	4.9
Illegal fishing activities	4	3.3
Loss of riparian vegetation	2	1.6
Overgrazing	2	1.6
Increased vegetation	1	0.8

Perceptions on Fish Stocks

Declining fish populations emerged as a major concern. About 76.4% of respondents reported that fish catches had decreased in the last five years, while only 12.2% believed catches had increased, and 6.5% noted no change. Regarding species composition, the most commonly caught fish included *Haplochromine*

cichlid (41.5%), *Lungfish* (40.7%), *Cat fish* (37.4%), and *Nile Tilapia* (28.5%). When asked which species had declined most significantly, respondents emphasized *Nile Tilapia* (56.9%), followed by *Lungfish* (22.0%), and *Cat fish* (19.5%). A few respondents also mentioned declines in species such as *Nile perches*, *Local catfish*, and *African butter catfish*. (Table 6)

Table 6. Trends in Fish Species Catch and Most Affected Species Over the Past Five Years

Variable	Frequency (N)	Percent (%)
Trends in Fish Species Catch		
Decreased	94	76.4
Increased	15	12.2
Normal	8	6.5
Note sure	6	4.9
Total	123	100.0
Fish species decreased		
Nile Tilapia	70	56.9
Cat fish	24	19.5
Nile perches	11	8.9
African butter catfish	6	4.9
Lake Victoria squeaker	5	4.1
Local catfish	3	2.4
Haplochromine cichlid	1	0.8
Sudan catfish	1	0.8
Eastern bottlenose	1	0.8
Sadler's Robber	1	0.8
Ocekkated Labyrinth fish	1	0.8

Factors Influencing Willingness to Participate in Conservation

Willingness to Participate

The majority of respondents (84.6%) expressed willingness to participate in conservation activities aimed at protecting the Mara River wetland. A small proportion (15.5%) indicated unwillingness. This suggests strong community readiness to engage in wetland conservation if appropriate support mechanisms are provided.

The majority of respondents (51.2%) expressed willingness to engage in sustainable farming, followed by tree planting (39.0%) and river conservation (35.0%), while fewer indicated

interest in education and awareness (28.5%), decision-making (17.1%), patrol activities in the wetland (15.4%), and sustainable livestock keeping (12.2%).

Respondents identified several forms of support that would facilitate their involvement in conservation initiatives. The most frequently cited was training and education (74.0%), reflecting the need to build local knowledge and capacity for effective conservation. Nearly half of respondents (47.2%) emphasized the importance of tools and equipment, while 32.5% highlighted financial support as a key enabler. Additionally, 30.1% mentioned the need for greater community collaboration to enhance collective action. (Table 7)

Table 7. Willingness to Participate in Wetland Conservation and Required Support

Variable	Frequency (N)	Percent (%)
Willingness to Participate		
Yes	104	84.6
No	19	15.5
Activities Respondents Are Willing to Engage In		
Sustainable farming	63	51.2
Tree planting	48	39.0
Conserving river	43	35.0
Education and awareness	35	28.5
Decision making	21	17.1
Patrol in the wetland	19	15.4
Sustainable livestock keeping	15	12.2
Type of support		
Training and education	91	74.0
Tools and equipment	58	47.2
Financial support	40	32.5
Community collaboration	37	30.1

A binary logistic regression analysis was conducted to examine the socio-demographic and livelihood factors influencing willingness to participate in wetland conservation. The final model demonstrated a good fit (Hosmer-Lemeshow $\chi^2 = 8.260$, $p = 0.408$) and explained between 14.5% (Cox & Snell R^2) and 25.1% (Nagelkerke R^2) of the variance in willingness.

Two predictors were statistically significant. Gender had a significant effect, with females

being less likely to express willingness to participate in conservation compared to males ($\beta = -1.799$, $p = 0.011$, OR = 0.166, 95% CI: 0.041-0.660). Similarly, age was significant, where respondents aged 31-45 years were less likely to be willing compared to those aged 18–30 years ($\beta = -2.117$, $p = 0.019$, OR = 0.120, 95% CI: 0.021-0.702). Other age categories, occupations, and activities engaged at the wetland did not significantly influence willingness (Table 8).

Table 8: Effects of demographic factors and wetland activities on willingness to participate in conservation activities. Regression coefficients (B), degrees of freedom (df), and p-values are shown; significant predictors ($p < 0.05$) are indicated.

Variable	B	df	Significance (p)
Age (18-30)	-0.617	1	0.483
Age (31-45)	-2.1	1	0.019
Age (46-55)	-0.985	1	0.203
Gender (Female)	-1.8	1	0.011
Occupation (farmer)	-1.3	1	0.191
Occupation (Trader)	-19.9	1	0.998
Occupation (Livestock keeper)	0.45	1	0.456
Activity engaged in the wetland (Farming)	-0.14	1	0.825
Activity engaged in the wetland(recreation)	-22.36	1	0.99

DISCUSSION

The study evaluates community awareness of threats to the Mara River Wetland, the perceived impacts on local livelihoods, and the willingness of community members to engage in conservation. The study revealed majority of respondents were male and within the productive age group of 31-60 years, with farming identified as the dominant livelihood and low levels of formal education recorded across households. Respondents perceived invasive species (water hyacinth) together with pollution, climate change, and agricultural expansion as the most critical threats to the wetland. Livelihood trends showed an increase in farming activities, a decline in fishing, and mixed changes in livestock keeping, reflecting shifts in resource dependence. Most community members viewed the wetland as degraded, citing reduced water quality, sedimentation, and invasive species spread, while also noting a decline in fish catches and diversity of species, despite these challenges, many respondents expressed a willingness to participate in conservation initiatives, provided they receive adequate training and support. These findings provide an important basis for understanding the socio-ecological dynamics shaping wetland management in the Mara River Wetlands.

The majority of respondents were male and within the productive age group of 31-60 years, with farming as the dominant livelihood and relatively low levels of formal education. These findings reflect the socio-economic stuff of rural East African wetland communities, where resource dependence is high and men are often more engaged in wetland-related activities, while women remain underrepresented in both livelihood decision-making and conservation processes (Ondiek *et al.*, 2020; Niemann *et al.*, 2024). The predominance of primary education indicates limited opportunities for diversification into non-farm employment, which may restrict adaptive capacity and reinforce reliance on wetland resources. Similar demographic characteristics have been reported in studies of communities around the Lake

Victoria basin (Yusuph *et al.*, 2018), suggesting that livelihood dependence on wetlands remains widespread where educational attainment is low.

Respondents identified invasive species (particularly water hyacinth), pollution and climate change as the most critical threats. These perceptions are strongly supported by ecological studies in the Lake Victoria Basin. Water hyacinth has long been recognized as a disruptive invasive species, negatively affecting fisheries, water transport, and water quality (Lubembe. 2023). Concerns over pollution are consistent with evidence that chemical runoff and effluent discharge contribute to eutrophication and biodiversity loss in the Mara River system (Cristiano *et al.*, 2008; Gavrilas *et al.*, 2025). Climate change has been recognized as major threat to wetlands ecosystems as increasing rainfall variability and rising temperatures are altering hydrological regimes, reducing water availability, and exacerbating wetland degradation in East African river basins (McClain *et al.*, 2014; Mutanda & Nhamo, 2024). Importantly, the variation in threat perceptions across villages highlights how localized exposure and livelihood dependence shape awareness of ecological risks (Burcham *et al.*, 2025). Communities more directly dependent on fishing or water use may be more sensitive to ecological degradation than those relying primarily on agriculture or livestock.

Community members reported that farming has increased substantially in recent years, while fishing declined and livestock keeping exhibited mixed trends. This observation aligns with broader land-use changes in East Africa, where wetlands are increasingly converted for agriculture due to population growth and food demand (Liu, 2022). The decline in fishing resonates with regional evidence of reduced fish catches and diversity caused by overexploitation, pollution, and invasive species (Nyamweya *et al.*, 2023). Increased irrigation farming reported by respondents also reflects adaptive strategies to rainfall variability, which are becoming common across East Africa in

response to climate change (Mwalusepo *et al.*, 2015; Alemayehu *et al.*, 2017). These demonstrate how livelihood shifts not only reshape the socio-economic landscape but also accelerate ecological pressures on already vulnerable wetland ecosystems.

The majority of respondents perceived the Mara River wetland as degraded, citing reduced water quality, increased sedimentation, and spread of invasive species. These perceptions are supported by ecological monitoring that documents deteriorating water quality, siltation, and hydrological changes driven by Loss of riparian vegetation, agriculture, and mining in the Mara Basin (McClain *et al.*, 2014). Notably, age and occupation significantly influenced perceptions of wetland condition, with older respondents and those directly reliant on wetland-based activities more likely to report degradation. This supports social-ecological theory emphasizing that direct livelihood engagement provides experiential knowledge, making resource users more observant of environmental change (Charles, 2022; Fiorella *et al.*, 2023). Such insights highlight the importance of integrating community knowledge with scientific assessments to improve wetland management.

The findings revealed that, the community's perception of a decline in fish catches and diversity, particularly of species such as Nile Tilapia (*Tilapia spp.*), Lungfish, and Cat fish. These reports are consistent with scientific evidence documenting reduced native fish populations in the Mara and Lake Victoria systems due to overfishing, pollution, and loss of breeding habitats (Njiru *et al.*, 2008). The community's recognition of declining fish diversity further supports ecological studies linking invasive, water hyacinth, to altered aquatic food webs and decreased resilience of native species (Nyamweya *et al.*, 2023). The convergence between local knowledge and ecological research emphasizes the value of indigenous observations in monitoring biodiversity trends, particularly in data-limited contexts (Berkes *et al.*, 2004).

Most respondents expressed a willingness to engage in conservation initiatives, provided they receive training and equipment, a finding consistent with studies from other African wetlands showing that participation improves when communities perceive tangible benefits, gain knowledge, and are supported with resources (Hassan, 2015; Mombo *et al.*, 2011). However, willingness was shaped by demographic factors, with women and middle-aged respondents less likely to participate, reflecting gendered barriers such as unequal access to resources, restrictive cultural norms, and limited decision-making power (Krietzman, 2019). Age dynamics also played a role, with younger respondents often more adaptable to innovations while older individuals tended to prioritize immediate livelihood needs over long-term sustainability (Saengavut & Somsasdi, 2022; Kizigo *et al.*, 2023). Beyond these socio-demographic influences, the variation in willingness across activities highlights the importance of aligning conservation strategies with livelihood priorities, as communities are more motivated when practices yield direct social or economic benefits (Pretty *et al.*, 2018; Altieri & Nicholls, 2020). Activities such as tree planting and river conservation not only provide ecological resilience but also strengthen social cohesion and local identity, suggesting that embedding conservation within cultural and livelihood narratives may foster sustained engagement (Infield *et al.*, 2018; Ruiz-Mallén & Corbera 2013; Pretty, 2011). In contrast, lower interest in governance-oriented roles, including awareness campaigns, decision-making, and patrols, reflects structural and socioeconomic barriers that often constrain participation in non-income-generating activities (Guragain, 2024; Palmer & Chuamuangphan, 2021).

CONCLUSION AND RECOMMENDATIONS

Conclusion

This study demonstrates that the Mara River Wetland is facing multiple and interacting

threats, including invasive species, pollution, climate change impacts, and agricultural expansion, all of which are directly linked to local livelihood practices. Communities perceive clear signs of ecological degradation, such as reduced water quality, increased sedimentation, fish population decline, and the spread of invasive vegetation. Despite these challenges, the majority of respondents expressed strong willingness to participate in conservation efforts, highlighting the potential for community-based management approaches. However, socio-demographic factors such as age and gender significantly influenced willingness, with women and middle-aged groups showing lower likelihood of participation compared to men and youth. These findings underscore the importance of integrating ecological management with social dimensions, particularly local perceptions, capacities, and demographic dynamics, in order to sustain the ecological and socio-economic functions of the Mara River Wetland.

Recommendations

Sustainable management of the Mara River Wetland requires strengthening community-based conservation by addressing both ecological and social drivers of degradation. First, targeted awareness and training programs should be developed to enhance community knowledge of wetland values and threats, while providing practical skills for sustainable farming, fishing, and grazing practices. Second, provision of tools, equipment, and financial incentives can encourage active participation, particularly among groups with limited resources. Third, gender-sensitive and youth-focused interventions are needed to bridge participation gaps and ensure equitable involvement in conservation decision-making and activities. Finally, stronger institutional coordination between local communities, government agencies, and basin-level authorities is essential to enforce regulations, control pollution, manage invasive species, and support collaborative monitoring. By combining ecological restoration with locally

driven conservation initiatives, the long-term resilience and functionality of the Mara River Wetland can be secured for both people and biodiversity

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WATER QUALITY AND REINTRODUCTION OUTCOMES: A CASE STUDY OF THE KIHANSI SPRAY TOAD

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ABSTRACT

Low post-release survival of captive-bred Kihansi Spray Toads (*Nectophrynoides asperginis*) may be linked to their abrupt exposure to untreated natural water after being raised under treated conditions in captivity. To explore this hypothesis, we assessed the feasibility of establishing an experiment at the Kihansi Captive Breeding Facility to raise toads under untreated water conditions. Prior to the experiment, water quality was evaluated at 15 sites including spray wetlands, the captive breeding facility, and surrounding water sources along the Kihansi River. The data were collected during wet and dry seasons in 2022. Field measurements of pH, temperature, salinity, dissolved oxygen, and conductivity were recorded using a Multi 3430 SET F probe, and preserved samples were analyzed in the laboratory for ammonia, nitrates, phosphates, and other parameters following standard protocols. Most parameters were within tolerable limits for amphibians; however, ammonia concentrations exceeded safe thresholds at all wetland sites in both sampling periods. Although the findings are based on only two seasonal sampling periods, suggest elevated ammonia may pose a risk to released toads. Given potential temporal fluctuations in water quality, we recommend implementing a more frequent and long-term monitoring program to better characterize water conditions and guide adaptive management.

These preliminary insights highlight the need for controlled trials assessing toad acclimation to untreated water and contribute valuable information to improve reintroduction strategies for this critically endangered amphibian.

Keywords: Ammonia toxicity; captive breeding; Kihansi Spray Toad; reintroduction; water quality

INTRODUCTION

It has been reported globally that most species, amphibians in particular, are facing global declines and consequent losses in species richness, largely due to habitat destruction, alteration and fragmentation (Stuart, 2008, Green, *et al.*, 2020). These declines specifically in amphibians are closely related to environmental modifications, diseases, introduction of exotic species, overexploitation, climate change impacts, physiological effects due to the elevation of chemical toxicity in water, altered hydrological

cycle, and deteriorating water quality (Boyer and Grue 1995, Rollins-Smith, 2009). Moreover, amphibians are more vulnerable to water contaminants than other taxa due to their easily permeable skin, limited vagility and complex life cycles that depend on both aquatic and terrestrial habitats (Stebbins and Cohen, 2021). The effects of toxic water contaminants on amphibians have been widely studied mostly documenting: high concentrations of nitrates, nitrites, total phosphates, chloride, unionized ammonium, high values of electrical conductance, deviated from neutral pH values, high concentrations of organochlorine and

organophosphates pesticides, pharmaceuticals, low dissolved oxygen concentrations, and high levels of organic matter have been linked to reduced clutch size, low hatching and larval survival rates, and growth rate (Laposata and Dunson 2000; de Solla, *et al.*, 2002; Sowers, *et al.*, 2009; Babini, *et al.*, 2018). Ammonia is toxic to many aquatic organisms (Thurston, *et al.*, 1981) and normally occurs in two forms when dissolved in water; the un-ionized ammonia, NH_3 , and the ionized ammonium ion, NH_4^+ with the proportion of these two forms depends primarily on the water pH and temperature (Thurston, *et al.*, 1981). However, according to Thurston, Chakoumakos *et al.* (1981) study the former form is more toxic to amphibians and occurs when pH is above 8.5, being 10-fold toxic in every one unit of pH increase. Conductivity, a measure of total dissolved ions (TDS), indicates the salinity of water; elevated levels can cause osmotic stress, disrupting the water balance of amphibians (Sanzo and Hecnar, 2006).

For example, a study by Dunson, Wyman *et al.* (1992) found that low pH obstruct fertilization and inhibit embryonic development of the common frog (*Rana temporaria*) through altering physiological ionic balance in amphibians and consequently cause stunted growth and poor survivorship. Other researchers in Canada found that exposing frogs in organophosphates had effects in increases in the coefficient of variation in the size of genomes, increased adult mortality and physiological malformations to frogs resided to the agricultural fields (Boyer and Grue, 1995). Surveys conducted at the Klamath Basin National Wildlife Refuges found that waters in the refuge had high levels of temperatures, elevated pH, and low dissolved oxygen which was either lethal to, or caused significant malformation of, developing frog embryos (Sorenson and Schwarzbach 1991, MacCoy, 1994). Therefore, amphibians' exposure to toxic water contaminants from runoff including surface and river flows has a high probability of their population decline.

The Kihansi Spray Toad, KST (*Nectophrynoides asperginis*), a small, ovoviviparous toad, was first discovered in 1996 in the Kihansi River Gorge in Tanzania and described as new to science in 1998 (Poynton, *et al.*, 1998) (IUCN, 2009). Its entire global range was limited to less than 2 hectares of specialized wetland habitat, the Kihansi Falls spray zone, which was permanently saturated by mist from the waterfall. Following construction of the Kihansi Dam in 1999, which diverted up to 90% of the Kihansi river's flow, drastically reduced the spray volume, leading to the rapid desiccation of this microhabitat and the subsequent catastrophic decline of the toad population (IUCN, 2009). The species was declared Extinct in the Wild by the International Union for Conservation of Nature (IUCN) in 2009, making its conservation story a stark example of a habitat specialist vulnerable to anthropogenic change.

Since 2010s, the Tanzania Wildlife Research Institute (TAWIRI) and other conservation stakeholders have been conducting reintroduction programs as an attempt to restore the Kihansi Spray Toad population in their natural habitat with more than thirty thousand individuals released. However, despite these efforts, wild population of this species have not yet reestablished. Among many hypotheses raised as to why the KST population has not been re-established in their natural habitat was that, the KSTs lack physiological adaptivity to natural untreated water after being reared under controlled, treated water conditions in the captive breeding, hence fail to survive in the natural environment upon contact with untreated water. It is well known that, amphibian reintroductions rely significantly on habitat quality specifically, water quality as the paramount factor for success and used as a decisive factor in numerous cases (Germano and Bishop, 2009). Therefore, while the dam's construction caused a physical reduction in water quantity, the consequent change in water quality is hypothesized to be a proximate cause of mortality and a continued barrier to re-establishment of the KST in the wild. The

aim of this study was to test the argument of whether the KST lack adaption once contacted with untreated natural water in the wild as a matter of fact that they are reared under controlled treated water in the captivity.

METHODS

Study area

The Kihansi Gorge (coordinates 8°35'12" S, 35°51'02" E) features a narrow band of rainforest bordered by Miombo woodland. In the spray zone of the Kihansi River waterfalls, a distinctive type of vegetation develops, resulting in a series of spray wetlands (Weldon, *et al.*, 2019). It is located on the southern Udzungwa mountain escarpment within the Eastern Arc Mountain range. The gorge is covered with a narrow strip of montane forest specifically along the river channel which stretches about one kilometer (Fig. 1) (Weldon, *et al.*, 2020). The Kihansi gorge montane forest is situated at an elevation of 1,150 meters above sea level and experiences approximately

1,500 millimeters of rainfall each year in a single seasonal pattern (Vandvik, *et al.*, 2014). The warmest temperatures occur in January and February, while June and July are the coolest months, with average daily maximum and minimum temperatures of 25 °C and 13 °C, respectively (Ikanda, *et al.*, 2019).

Data collection

Experimental design

Before implementation of the experiment for examining whether Kihansi Spray Toads failed to survive in the wild because of abrupt exposure to untreated/unpurified water after being raised under treated/purified water in the facility, it was necessary to (i) establish where and how to set up the experiment, and (ii) determine the quality of water used for KST in the Captive breeding Facility and in the spray wetlands to enable identification of natural/ untreated a source point of water for the experiment at the facility.

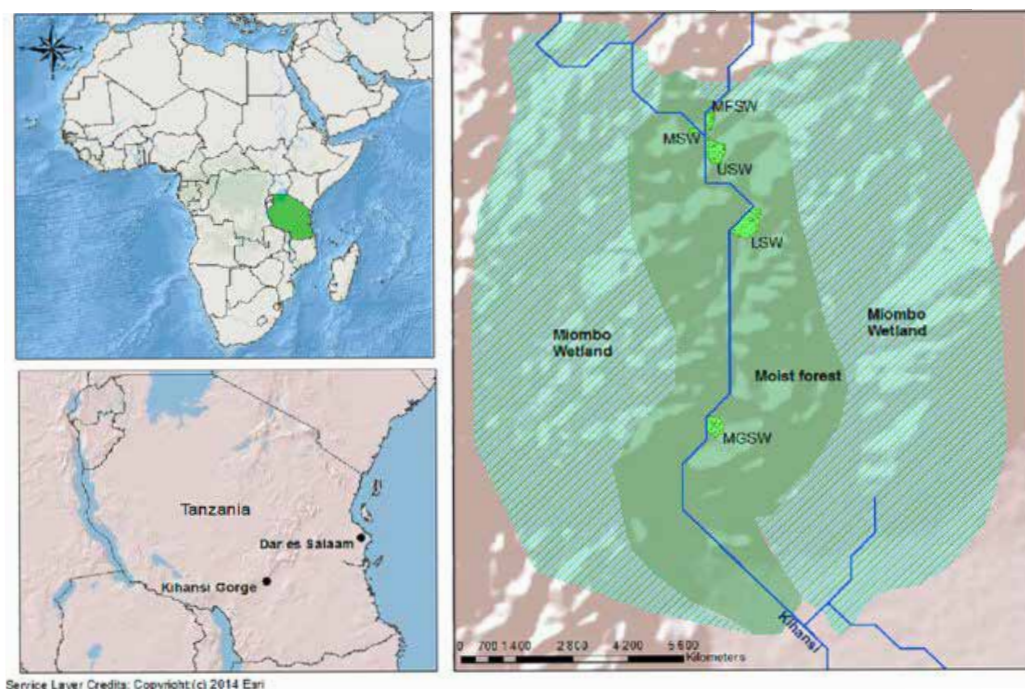


Figure 37: Map of Kihansi Gorge indicating the stretch of the Kihansi River and location of spray wetlands. MFSW -Main Falls Spray Wetland, MSW -Mhalala Spray Wetland, USW -Upper Spray Wetland, LSW -Lower Spray Wetland, MGSW -Mid-Gorge Spray Wetland adopted from (Weldon, Channing et al. 2019)

(i) Where and how to establish the experiment to raise Kihansi Spray Toad under untreated/unfiltered water

A preliminary visit to Kihansi Station was in September 2021 by a team of 7 staff/researchers from TAWIRI. The team held a meeting to discuss the logistics for setting up the intended experiment including space and source of water for the experiment. The following were realized from the discussion and field observations

Space availability for experiment: The Captive Breeding has an experimental room with 15 empty cages and a space for setting other 13 cages. The 13 cages would need to be procured, and the room would need additional water storage tank, pipes and accessories to connect untreated water from identified source which its water will need to be pre-analyzed for the physical, chemical, and biological parameters as basis for monitoring. Alternatively, the option was to expand the facility by repairing, installing, and fitting an old container that would be made available by TANESCO upon official request. But the container would need repairs and installations of air condition, shelves, and the aquariums.

Potential sources of water for the experiment: From the discussion, it was learned that since 2016, the Kihansi Captive Breeding Facility (KCBF) has been using unfiltered water because the filters at the facility were broken. Nevertheless, the KST population was never affected by using such water i.e., unfiltered water. However, water used at Kihansi Captive Breeding Facility is supplied by TANESCO Water Treatment Plant and already pre-filtered and treated for human uses. Therefore, the possible reason why the toad's population kept still striving. The Quality of water at the facility is also not monitored because of lack of monitoring equipment.

(ii) Assessment of water quality used for KST in the Captive breeding Facility in the spray wetlands and irrigation water.

To identify source of water for use in the experiment it was necessary to assess and compare the quality of water maintaining the Kihansi Spray Toads (KST) in the spray wetlands and Captive Breeding facility, and any other potential water source around the facility e.g., irrigation water supplied by TANESCO for flower gardens and compounds.

Experimental procedures

Water samples were collected during the dry and wet seasons (September 2021 and March 2022) from (i) water sources feeding the spray wetlands including the Kihansi dam (at the outlet), water intakes for the upper, lower, and mid-gorge spray wetlands i.e., Jabali, Kihansi River Main falls, lower wetland and Handaki stream intakes); and water coming out of sprinkler system; (ii) other water points along the Kihansi River (after the spray wetlands) that were potential for use in the proposed experiment in KCBF including suspended bridge, Mlumbaji stream, and Kihansi daraja, and Irrigation water just outside KCBF; and inside the KCBF for comparison purposes.

Water samples were stored in plastic bottles and preserved in a cooling box while in the field, and later stored in a freezer at KCBF for two to three nights before being transported to Dar es Salaam, for laboratory analysis at the Department of Water Resource Engineering, College of Engineering Technology. In the lab, the samples were tested and analyzed for nine physio-chemical parameters including total suspended solids (TSS), total dissolved solids (TSD), nitrates, phosphate, ammonia, nitrite, total hardness, free chlorine, and turbidity following standardized methods for the examination of water and

wastewater samples (Alpha *et al*, 1992). Other five physical parameters including pH, temperature, salinity, dissolved oxygen, and electrical conductivity were directly analyzed in the field using a multi-probe gadget (Multi 3430 SET F; S940, FAO 925, TC 925; WMW Germany).

RESULTS

The results for all 14 physico-chemical parameters in all 15 sampling sites are presented in appendix 1, but eight of the 14 parameters were selected including: water hardness (mg/L) dissolved oxygen (DO) (mg/L) ammonia (NH₃) (mg/L), free Chlorine (Cl₂) (mg/L), Nitrite (NO₂) (mg/L), Phosphate (PO₄) (mg/L), Nitrate (NO₃) and pH. When levels of these

parameters in water exceed recommended thresholds in water, it affects the well-being of amphibians (Table 1) (Odum & Zippel, 2011). From the 15 sampling sites, a subset of key results was selected, KCBF and irrigation and present the water quality results for sites considered for the proposed experiment (Table 2).

Except for ammonia, the water quality condition across all sites and seasons, the levels of all other parameters are within levels that can be tolerated by amphibians (Table 2 and 1). The levels of ammonia however, across the three wetlands in wet and dry seasons were above the threshold that can be tolerated by amphibians (Table 2) which may affect amphibians survival.

Table 1. Water quality parameters that are of great concern for amphibian health

Parameter	Tolerable level	Effect on amphibians
Water hardness (mg/L)	< 75 mg/L	Skin problems in some species osmotic regulation of the amphibian.
Dissolved Oxygen (mg/L)	> 80% saturation (> 6.6 mg/L)	Important for respiration and aerobic processes
Ammonia (NH ₃) (mg/L)	< 0.2 mg/L	Very toxic
Free Chlorine (Cl ₂) (mg/L)	Undetectable	Very toxic
Nitrite (NO ₂) (mg/L)	< 0.5mg/L, but ideally zero	Toxic
Phosphate (PO ₄) (mg/L)	Toxicity maybe species specific. EPA limit to 10 mg/L.	Toxic to many animals interferes with calcium metabolism
Nitrate (NO ₃)	< 50 mg/L	Slightly toxic
pH	Near Neutral	Disrupts ion exchange if no within acceptable range

Table 2. Water quality parameters assessed on potential water to be used as non-treated water for the Kihansi Spray Water

Parameter	Tolerable levels	CBF: Tap-water before entering cages		CBF: Spray-water inside cage		Irrigation: Garden Tap		Upper spray wetland		Lower spray wetland		Mid-gorge spray wetland	
		Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet
Water hardness (Calcium and Magnesium) (mg/L)	< 75 mg/L	5.5	6.1	5.5	5.95	7.95	5.86	7.59	6.8	4.01	4	5.9	5.8
Dissolved Oxygen (mg/L)	> 80% saturation (> 6.6 mg/L)	6.25	7.53	6.04	7.4	6.15	7.5	7	7.54	6.87	7.42	7.33	7.39
Ammonia (NH ₃) (mg/L)	< 0.2 mg/L	0.12	0.28	0.13	0.19	0.16	0.07	0.28	0.27	0.22	0.22	0.3	0.28
Nitrite (NO ₂) (mg/L)	<0.5mg/l, but ideally zero	0.003	0.001	0.002	0.001	0.004	0.002	0.001	0.003	0.009	0.01	0.004	0.005
Nitrate (NO ₃)	< 50 mg/L	0.01	2	0.01	1.9	0.3	0.5	0.8	1.6	2	2.3	1	1.9
pH	Near Neutral	7.03	7.36	6.93	7.36	6.9	7.47	6.8	7.61	7.55	8	6.71	7.33
Free Chlorine (Cl ₂) (mg/L)	Undetectable	0	0	0.1		0	0						
Phosphate (PO ₄) (mg/L)	Toxicity may be species specific. EPA limit to 10 mg/L.	0.02	3.9	0.03	2.32	0.1	2.27	0.04	4.3	0.03	5.5	0.04	3.95

DISCUSSION AND CONCLUSION

The quality of water from all sampling stations, both in wet and dry seasons, is overall in good condition as the levels of physio-chemical parameters analyzed are within Tanzania standards allowable limits for human consumption. However, in relation to amphibians, the quality of water in the spray wetlands is questionable as high levels of ammonia were detected (Table 1) both during the dry and wet seasons (Table 2). Given the nature of the study area, the source of this ammonia can be related to anthropogenic activities specifically agriculture in the up streams as application of fertilizers and pesticides are evident in the area. It was established that, Nitrogen pollution from anthropogenic sources enters water bodies through agricultural runoff or percolation linked with nitrogen fertilization, livestock, precipitation and effluents from industries and human wastes (Rouse, *et al.*, 1999). Therefore, having this possibility of sources, more surveys are encouraged in the future to detect the source of this parameter. The occurrence of free chlorine during the dry season of the study only for the water entering the cages was probably sourced from the water disinfected by water supplies authority (TANESCO in this case) for human use as it is worldwide known that the source of free chlorine is from chlorination of municipal water supplies authorities (Brungs, 1973). Given the toxicity of these parameters to amphibians, they may contribute to the low survival of KST in the wetlands. However, these findings are not conclusive because water samples were not collected directly from the sprinkler nozzles. Before water is supplied to sprinkler pipes in the wetlands, it passes through mechanical filters (i.e., from the intake to the mechanical filter) where no samples were collected and tested. Hence, before making any conclusion, we recommend collection and testing of water samples direct from the nozzles and from the mechanical filters to identify source of ammonia. Moreover, since the water sampling was taken once in each dry and wet seasons, we recommend future study which

will collect frequent samplings throughout the year for more reliable findings.

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Appendix 1: Water quality physical and chemical parameters analyzed from natural water (untreated) and non-natural water (treated) used to / expected for use in maintaining Kihansi Spray Toads in the gorge and in the captive breeding facility.

	Site name/ Parameter	pH	Temp (deg C)	EC (uS/ cm)	DO (mg/L)	SAL	Turb (NTU)	TSS (mg/L)	TDS (mg/L)	Nitrate (mg/L)	Phosphate (mg/L)	Ammonia (mg/L)	Nitrite (mg/L)	Total hardness (mg/L)	Free Chlorine (Cl)
	Dry season														
1	Damsite outlet	6.28	26.5	25	8.61	NIL	5.7	NIL	12	0.6	1.31	0.18	0.007	4.14	
3	Mainfall intake	6.82	29.1	21.5	7.44	NIL	4.37	NIL	10	1.5	0.78	0.18	0.005	3.84	
5	Lower wetland water intake	6.79	29.4	21.2	7.4	NIL	5.09	5	10	0.2	0.13	0.18	0.002	3.92	
9	Handaki water intake	6.59	28.9	44.7	7.49	NIL	3.44	16	20	0.8	2.86	0.09	0.005	5.7	
4	Upper wetland - sprinklers	6.8	30.5	49.7	7	NIL	1.1	14	22	0.8	4.29	0.28	0.001	7.59	
7	Lower spray wetland sprinklers	7.55	30.1	23.4	6.87	NIL	14.5	10	10	2	1.5	0.22	0.009	4.01	
10	Midgorge wetland sprinklers	6.71	28.8	44.6	7.33	NIL	30.8	25	22	1	1.45	0.3	0.004	5.9	
6	Mhalala stream	6.78	29.7	22.7	7.32	NIL	9	NIL	10	0.1	0.01	0.15	0.001	3.85	
8	Handaki stream	6.9	29.3	35.2	7.25	NIL	2.94	2	16	3.3	3.24	0.17	0.009	5.6	
12	Mlumbaji stream	6.79	32.3	67.6	6.21	NIL	0.69	NIL	32	0.1	0.7	0.19	0.002	12	
11	Suspension bridge	7.24	28.9	21.8	7.14	NIL	3.63	NIL	10	1	0.28	0.14	0.004	3.89	
13	Irrigation water	6.9	34	33.9	6.15	NIL	1.3	NIL	15	0.3	1.6	0.16	0.004	7.95	
14	Captive Breeding facility (tape)	6.93	34.2	34.2	6.04	NIL	0.21	NIL	16	0.01	0.19	0.13	0.002	5.5	0.1
15	Captive Breeding facility sprays/ mists	7.03	33.5	49.1	6.25	NIL	1.28	NIL	20	0.01	0.5	0.12	0.003	5.5	0
	Wet season														
1	Damsite outlet	7.56	19.5	20.8	7.16	NIL	3.54	NIL	10	0.9	1.84	0.1	0.005	4	
3	Mainfall intake	7.5	20.1	20.2	7.9	NIL	4.82	NIL	10	2.6	1.37	0.09	0.003	3.9	
2	Jabali intake	7.7	20.9	28.6	7.57	NIL	12.6	16	15	0.4	4.1	0.21	0.002	5.5	

	Site name/ Parameter	pH	Temp (deg C)	EC (uS/ cm)	DO (mg/L)	SAL	Turb (NTU)	TSS (mg/L)	TDS (mg/L)	Nitrate (mg/L)	Phosphate (mg/L)	Ammonia (mg/L)	Nitrite (mg/L)	Total hardness (mg/L)	Free Chlorine (Cl)
5	Lower wetland water intake	7.9	20.9	20.2	7.48	NIL	27	18	10	0.8	0.71	0.18	0.001	4	
9	Handaki water intake	7.19	21.7	38.6	7.7	NIL	1.59	NIL	20	1	3.68	0.06	0.004	5.5	
4	Upper wetland - sprinklers	7.61	21.3	28.7	7.54	NIL	6.95	NIL	14	1.6	4.3	0.27	0.003	6.8	
7	Lower spray wetland sprinklers	8	21.9	22.7	7.42	NIL	14.1	20	12	2.3	5.5	0.22	0.01	4	
10	Midgorge wetland sprinklers	7.33	22.5	40.7	7.39	NIL	11.2	17	20	1.9	3.95	0.28	0.005	5.8	
6	Mhalala stream	7.9	21.8	28.2	7.54	NIL	6.16	NIL	14	0.7	0.07	0.13	0.001	4	
8	Handaki stream	7.42	22.2	50.2	7.56	NIL	2.88	NIL	25	1.8	4	0.2	0.006	9.4	
12	Mlumbaji stream	6.99	24.6	59.8	7.6	NIL	44.1	28	31	2	0.6	0.22	0.004	11.5	
11	Suspension bridge	7.22	22.7	41.6	7.68	NIL	10.9	NIL	21	1.3	0.42	0.13	0.005	6.15	
13	Irrigation water	7.35	26.6	40.9	7.3	NIL	6.13	NIL	21	0.2	2	0.21	0.005	6	
14	Captive Breeding facility (CBF) spray	7.36	24.4	39.9	7.4	NIL	10.1	1	19	1.9	2.32	0.19	0.001	5.95	0
15	Captive Breeding facility:tap water	7.35	25.6	45.4	7.53	NIL	5.37	NIL	24	2	3.9	0.28	0.001	6.1	0.09
16	Treated water - Treatment Plant	7.2	27.3	48.7	7.11	NIL	0.47	NIL	25	0.9	2.1	0.03	0.002	5.5	0.08
17	Irrigation water - Treatment Plant	7.47	26.1	33.1	7.5	NIL	4.83	NIL	16	0.5	2.27	0.07	0.002	5.86	0

GENETIC BARCODING UNCOVERS HIDDEN DIVERSITY OF HAPLOCHROMINE CICHLIDS IN SHIRATI BAY, LAKE VICTORIA

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ABSTRACT

Haplochromine cichlids in Lake Victoria have undergone significant population declines following the introduction of the predatory Nile perch (*Lates niloticus*). These declines have been further exacerbated by eutrophication, overfishing, and habitat degradation, which have collectively contributed to the disappearance of many *Haplochromis* species from the catch. However, recent studies show that some species previously thought to have disappeared from catches may be recovering, possibly due to the recent decline in Nile perch populations. Despite these indications, the actual composition of *Haplochromis* species remains uncertain due to a lack of recent inventory data. Additionally, morphological similarities and intraspecific variations complicate identification, leaving populations vulnerable to overexploitation. To address this gap, the present study amplified a 620-base pair fragment of the cytochrome oxidase subunit I (COI) gene from 94 haplochromine individuals collected from Shirati Bay, Lake Victoria to assess species composition and distribution. Seven species were identified: *Haplochromis nyererei*, *H. alluaudi*, *H. erutus*, *H. simpsoni*, *H. sauvagei*, *H. nubilus*, and *H. pyrrhocephalus*. However, *H. nubilus* could not be matched in the NCBI database, highlighting the need to update existing genetic reference libraries. Phylogenetic analysis confirmed the accuracy of morphological identifications, emphasizing the value of genetic barcoding in clarifying taxonomy and supporting fisheries monitoring. Among the identified species, *H. erutus* and *H. simpsoni* were recorded in Tanzania for the first time. Similarly, *H. nyererei* was the most widespread, occurring at all sampling sites. In contrast, *H. erutus* and *H. alluaudi* were rare, despite being classified as Not Evaluated and Least Concern, respectively. These findings highlight a mismatch between conservation status and local abundances, emphasizing the need for periodic, site-specific reassessments of conservation status, particularly for rare and newly recorded species to ensure that management and conservation strategies are based on up-to-date local population data.

Keywords: DNA barcoding, Haplochromine cichlids, Lake Victoria, Mitochondrial cytochrome c oxidase subunit I gene (COX1), Species diversity;

INTRODUCTION

Lake Victoria, the world's largest tropical freshwater lake is globally recognized for its exceptional biodiversity, particularly its remarkable assemblage of haplochromine cichlids (Kishe-Machumu *et al.*, 2015). These fish formed one of the richest vertebrate radiations known, with over 500 documented species, more than 95% of which are endemic to the Lake Victoria Basin (LVB) (Outa *et*

al., 2020). The ecological dominance of haplochromines in the Lake Victoria was abruptly disrupted following the introduction of exotic fish species in the 1950's, specifically the predatory Nile perch (*Lates niloticus*) and Nile tilapia (*Oreochromis niloticus*), which were introduced to enhance fishery productivity and economic returns (Kishe-Machumu *et al.*, 2015; Witte, Wanink, Kishe-Machumu, *et al.*, 2007). At first, the introduced Nile perch constituted a very small portion of the fish biomass,

making up less than 0.5% (Witte, Wanink, & Kishe-Machumu, 2007). However, by the early 1980's, its population had exploded, triggering a severe collapse in haplochromine diversity (Mkumbo & Marshall, 2015). Previous studies reported nearly 200 haplochromine species became extinct, marking one of the most substantial species losses in the LVB (Outa *et al.*, 2020). The decline of haplochromines was influenced by factors such as their smaller size and habitat overlap with Nile perch, which made them highly susceptible to predation (Witte, Wanink, & Kishe-Machumu, 2007). Additionally, the introduction of exotic Nile tilapia is suspected to have negatively impacted haplochromine biodiversity through competition for food and the introduction of diseases and parasites (Awiti, 2011). While destructive fishing practices played a role, Nile perch predation and eutrophication were the primary factors that caused the decline of haplochromine populations (Witte, Wanink, & Kishe-Machumu, 2007).

Despite efforts to manage the fishery, including the formation of Beach Management Units (BMUs) under a co-management framework, current conservation and policy efforts largely prioritize economically valuable species like Nile perch and Nile tilapia (Njiru *et al.*, 2014; Outa *et al.*, 2020). In contrast, haplochromines continue to face threats such as habitat loss, overfishing, and are used as bait in longline fisheries (URT, 2021b). Although some studies suggest a potential partial recovery of haplochromine populations, driven by a decline in Nile perch catch from 40% in 2011 to 30% in 2020 (Mkumbo & Marshall, 2015; URT, 2021a), the actual extent of haplochromine composition is uncertain due to a lack of recent inventory data. Furthermore, the morphological

similarities and intraspecific variations among haplochromines complicate their identification and conservation, leaving the remaining populations vulnerable to overexploitation (Witte, Wanink, & Kishe-Machumu, 2007). This data gap limits accurate species identification and constrains the determination of their species composition and biodiversity hotspots, limiting effective haplochromine conservation strategies. Therefore, this study assessed the composition of Haplochromine cichlids in Shirati Bay using DNA barcoding for precise species identification. The results will support more inclusive biodiversity assessments and inform efforts to restore the ecological integrity of Lake Victoria's fish communities.

MATERIALS AND METHODS

Study area description

This study was conducted in Shirati Bay (SB), located in the Mara region in Tanzania along the south eastern shore of Lake Victoria (Figure 1). The site was selected because it is among the main landing site in Lake Victoria (URT, 2020). Also, the SB was selected due to its role as an important breeding and nursery ground supporting the recruitment of *Haplochromis* species (Waya *et al.*, 2017). Shirati Bay spans approximately 14 km² and has an average depth of 10 meters (Kashindye *et al.*, 2015). The region is characterized by a bimodal rainfall pattern, comprising two rainy seasons and a prolonged dry season from July to September (Waya *et al.*, 2017). Fishing is the predominant economic activity within the bay, where local communities depend heavily on the lake's resources.

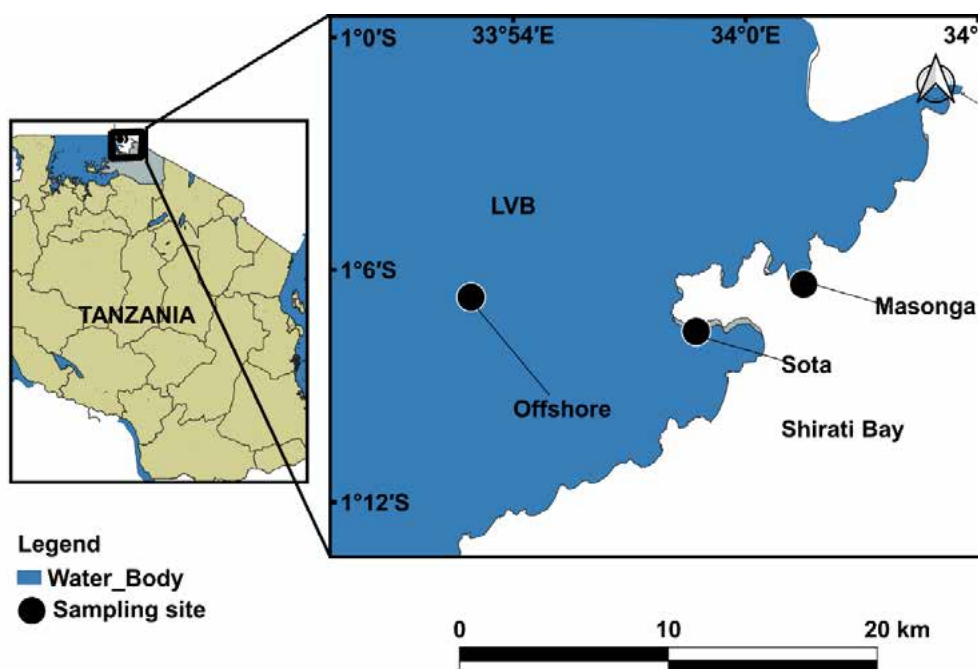


Figure 1. Map of the study area showing locations where haplochromine samples were collected. The map was created using QGIS software version 3.10.10 and shapefiles from the Database of Global Administrative Areas (GADM) (<https://gadm.org/maps/TZA.html>, accessed on 28th July 2025).

Fish sampling

Fish sampling was conducted at two primary landing sites within Shirati Bay: Sota and Masonga with an average distance of 12.87 km between sites, during February and March 2025 (Figure 1). These sites were purposively selected due to their high fishing activity and known presence of *Haplochromis* species. In addition, samples of haplochromine cichlids were obtained from local fishers fishing offshore of Shirati Bay to capture species from different habitats. Fish were collected using a beach seine net measuring 200 meters in length and 4 meters in width, with a cod-end mesh size of 4 mm. At each station, three separate hauls were conducted per sampling event. The harvested catch was sorted immediately to isolate *Haplochromis* species. A total of 32 individuals from each site were collected and identified using *A Guide to the Freshwater Fishes of Tanzania* (Eccless, 1992). Subsequently, approximately 2 g of muscle tissue was dissected from the pelvic

fin of each individual and preserved in 1.5 ml microcentrifuge tubes containing 99.9% ethanol for further analyses.

DNA extraction, COI amplification and sequencing

Genomic DNA was extracted from fin clip tissues using the FlAPure Animal Tissue/Cell/Blood DNA Extraction Kit (Genesand Biotech Co., Ltd, Beijing, China) in accordance with the manufacturer's guidelines. The integrity of the extracted DNA was assessed using a 1% agarose gel (Saiperaki *et al.*, 2024a). Afterwards, a 620 base pair fragment of the mitochondrial Cytochrome c Oxidase Subunit I (COI) gene was amplified using the primers FishF1 (5'-TCAACCAACCACAAAGACATTGGCAC-3') and FishR1 (5'-TAGA CTTCTGGGTGGC CAAAGAATCA-3') (Saiperaki *et al.*, 2024b). PCR amplification was conducted in 40 µL reaction volumes comprising 3.2 µL template DNA, OneTaq 2X Master Mix with Standard Buffer (New

England BioLabs Inc., MA, USA), 5 mg bovine serum albumin (BSA), and 0.3 µM of each primer (Rumisha *et al.*, 2023). The thermal cycling profile included an initial denaturation at 94°C for 5 minutes, followed by 35 cycles of denaturation at 94°C for 40 seconds, annealing at 54°C for 45 seconds, and extension at 72°C for 60 seconds. A final extension step at 72°C for 15 minutes was included to ensure complete DNA elongation (Morandus & Rumisha, 2024). PCR products were visualized using 1.5% agarose gels stained with SafeView DNA stain and visualized under a blue-light transilluminator (Mbilinyi *et al.*, 2023). Successfully amplified products were sent for Sanger sequencing at MacroGen Europe Laboratory using the ABI 3730XL DNA Analyzer (Applied Biosystems, Foster City, USA) (Saiperaki *et al.*, 2024a).

Data analysis

A total of 94 haplochromine cichlid specimens collected from Shirati Bay were successfully sequenced. The resulting sequences were first trimmed and translated into amino acid sequences using the vertebrate mitochondrial genetic code in MEGA version 11 (Tamura *et al.*, 2021) to detect and exclude sequencing errors and pseudogenes (Morandus *et al.*, 2025). The refined sequences were then collapsed into distinct haplotypes using FaBox

DNA Collapser version 1.61 (Villesen, 2007). Each unique haplotype was subsequently compared to sequences in the GenBank Nucleotide Database using the Basic Local Alignment Search Tool (BLAST) to assess similarity and confirm species identity (Fabiani *et al.*, 2023). A sequence match of ≥ 99.0% to known references was considered a successful identification (Saiperaki *et al.*, 2024a). Bayesian phylogenetic analysis was then performed using BEAST version 1.10.4 to reconstruct evolutionary relationships among the identified taxa (Morandus *et al.*, 2025). Additionally, the conservation status of each species was compiled from the IUCN Red List to support ecological interpretation and inform conservation measures (Rumisha *et al.*, 2024).

RESULTS

Identified *Haplochromis* species

A total of 94 COI sequences were obtained from the analysed samples. DNA barcoding confirmed the identity of nine haplotypes, with percentage identities ranging from 99.57 to 100 % (Table 1). Similarly, the expected value for all sequences analysed was 0, query coverage was 100 while the maximum score ranged from 857 to 869.

Table 1: Summary of the results obtained after comparing the obtained COI sequences of *Haplochromis* species from the SB, Tanzanian with those in the GenBank and BOLD databases.

Haplo- type	Species	Base pair	Maximum score	GenBank		BOLD	
				Query cover (%)	E- value	% Identity	% Identity
H1	<i>H. nyererei</i>	470	869	100	0	100	100
H2	<i>H. erutus</i>	470	869	100	0	100	100
H3	<i>H. sauvagei</i>	470	863	100	0	99.79	99.79
H4	<i>H. nubilus</i>	470	No match	No match	No match	No match	100
H5	<i>H. nyererei</i>	470	863	100	0	99.79	99.79
H6	<i>H. simpsoni</i>	470	869	100	0	100	100
H7	<i>H. alluaudi</i>	470	857	100	0	99.57	99.57
H8	<i>H. nyererei</i>	470	857	100	0	99.57	99.57
H9	<i>H. pyrrhocephalus</i>	470	863	100	0	99.79	99.79

The constructed Bayesian phylogenetic tree provided additional confirmation to the identified fish species (Figure 2). Closely related species were clustered under the same node implying that the amplified barcodes correctly identified the species.

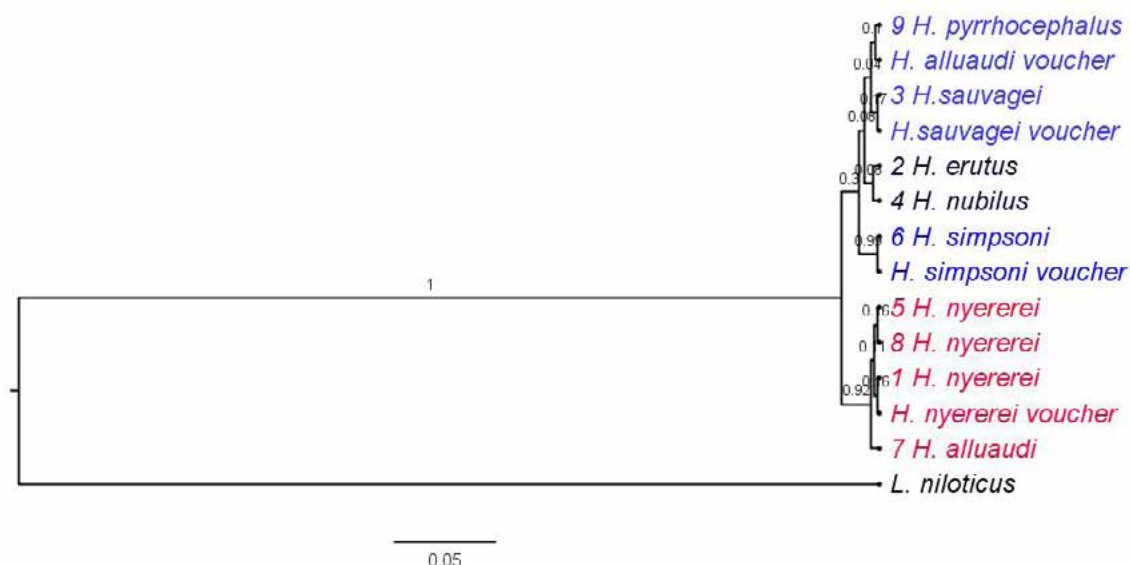


Figure 2. Bayesian phylogenetic tree showing the evolutionary relationships among cytochrome oxidase subunit I haplotypes of the sampled *Haplochromis* species

Haplochromis species composition and conservation status

The composition of the sampled *Haplochromis* species varied from 0% to 28.72% (Table 2). *Haplochromis nyererei* was the most abundant species, accounting for 28.72, 15.96% and 10.64% of samples collected from Sota, Masonga and Offshore site, respectively. Of

the seven *Haplochromis* species sampled, three (including *H. nyererei*, *H. alluaudi*, and *H. pyrrhocephalus*) were classified as Least Concern, two (*H. sauvagei* and *H. nubilus*) as Vulnerable, one (*H. simpsoni*) as Endangered, and one (*H. erutus*) as Not Evaluated. Although *H. alluaudi* is categorized as Least Concern and *H. erutus* as Not Evaluated, both were recorded at very low abundances in the catch (Figure 3).

Table 2: Composition of sampled *Haplochromis* species and their conservation status.

Specie name	% Sota	% Masonga	% Offshore site	IUCN list	Red list
<i>H. nyererei</i>	28.72	15.96	10.64	LC	
<i>H. erutus</i>	1.06	0	0	NE	
<i>H. sauvagei</i>	9.57	0	0	VU	
<i>H. nubilus</i>	10.64	8.51	0	VU	
<i>H. simpsoni</i>	1.06	0	0	EN	
<i>H. alluaudi</i>	3.19	0	0	LC	
<i>H. pyrrhocephalus</i>	0	3.19	7.45	LC	

VU - Vulnerable, LC - Least Concern, EN - Endangered, and NE – Not Evaluated.

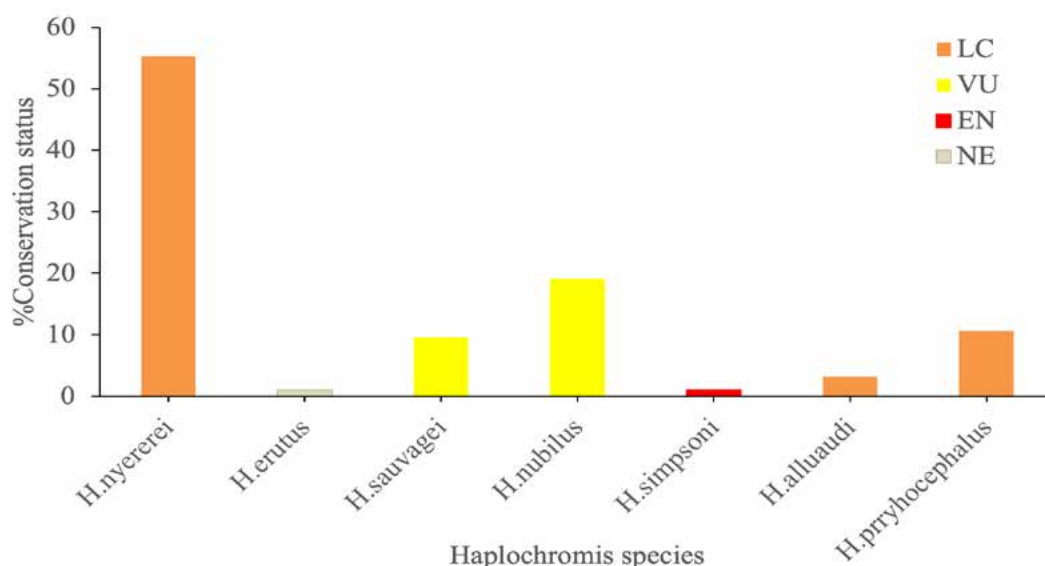


Figure 3. Percentage composition of each *Haplochromis* species in the total catch

DISCUSSION

Haplochromis species landed in the Shirati Bay

This study identified seven distinct *Haplochromis* species from the SB: *H. nyererei*, *H. alluaudi*, *H. erutus*, *H. simpsoni*, *H. sauvagei*, *H. nubilus*, and *H. pyrrhocephalus*. All species, except *H. erutus* and *H. simpsoni*, have previously been reported in Tanzanian freshwater ecosystems (Froese & Pauly, 2025). However, this is the first record of *H. erutus* and *H. simpsoni* in the country and at the landing site. Previous studies reported *H. erutus* and *H. simpsoni* are endemic to Lake Edward and Lake Nabugabo respectively (Ogutu-ohwayo, 1993; Vranken *et al.*, 2020). Their occurrence in Shirati Bay could be attributed to hydrological connectivity, seasonal migrations, or potential range expansions facilitated by environmental changes, allowing these previously restricted species to colonize new habitats (Nagl *et al.*, 2000). However, because previous studies relied solely on morphological identification, it is possible that these species were present in Lake Victoria but were overlooked in earlier regional surveys due to their cryptic or morphologically similar characteristics.

The barcoding approach used in this study effectively resolves the taxonomic uncertainties that have complicated earlier surveys, where *Haplochromis* species were frequently grouped together because of identification difficulties. Additionally, the phylogenetic analysis clustered sequences of the sampled *Haplochromis* species into well supported clades, confirming the taxonomic validity of the morphologically identified species. This molecular confirmation enhances taxonomic resolution, particularly for morphologically similar species, and supports more effective conservation strategies by ensuring that species-level data guide management decisions. Additionally, the results stress the importance of updating global taxonomic databases such as FishBase and BOLD Systems to reflect this refined understanding, thereby improving the accuracy and relevance of conservation planning, ecological monitoring, and biodiversity reporting.

Composition and Conservation Status of the sampled *Haplochromis* species

The findings revealed that *H. nyererei* was the most abundant haplochromine species across all sites. Previous studies did not report this species

individually, often grouping *Haplochromis* species due to the morphological challenges in distinguishing species (Kishe-Machumu *et al.*, 2015). The high abundance of *H. nyererei* may be attributed to its ecological versatility, allowing it to occupy both shallow and deeper habitats. Its feeding plasticity and aggressive reproductive behavior likely contribute to its ecological success and dominance (Katunzi *et al.*, 2003). Conversely, the low representation of *H. simpsoni* and *H. erutus* suggests that these species may be more habitat-specific or sensitive to environmental stressors such as overfishing and habitat modification. Their limited presence in the catch highlights a potential population decline and calls for urgent conservation action. Furthermore, the observed low abundances of *H. simpsoni*, *H. erutus*, and *H. alluaudi* highlight an important discrepancy between IUCN conservation categories and local catch composition. *H. simpsoni*, although classified as Endangered, appears underrepresented at the landing site, emphasizing the potential for localized overexploitation and the need for targeted monitoring. Conversely, *H. alluaudi*, categorized as Least Concern, was also recorded at very low abundance, suggesting that broad conservation assessments may not fully capture site-specific population dynamics. The rarity of *H. erutus*, currently Not Evaluated, highlights the importance of reassessing its conservation status in Lake Victoria, particularly given its low representation in catches. These findings indicate that conservation status at a global or regional scale may not always align with local abundance patterns, highlighting the need for site-specific surveys to inform management and conservation strategies.

CONCLUSION

This study recorded seven haplochromine cichlid species in Shirati Bay, including the first national records of *Haplochromis erutus* and *H. simpsoni*, species previously considered endemic to Lake Edward and Lake Nabugabo, respectively. Their detection may reflect range expansions or distributional shifts, but it is also possible that these species were previously present and overlooked due to limitations in

earlier surveys. The dominance of *H. nyererei* across all sampling sites suggests a shift in species composition, potentially linked to the decline of Nile perch populations in Lake Victoria. However, the limited number of reference barcodes in nucleotide databases highlights the need to update and expand these libraries to facilitate accurate species detection and effective conservation monitoring. Similarly, the low abundances of *H. simpsoni*, *H. erutus*, *H. alluaudi* and other rare *Haplochromis* species, combined with the mismatch between their conservation status and local catch composition indicates the need for periodic, site-specific reassessments of conservation status, particularly for rare and newly recorded species to ensure that management and conservation strategies are based on up-to-date local population data. Additionally, management strategies should prioritize the protection of these rare and newly detected species through sustainable fishing practices, habitat preservation, and community engagement to raise awareness about practices that minimize impacts on fish populations. The establishment of protected areas or fish refugia within Shirati Bay is recommended to safeguard vulnerable *Haplochromis* species and prevent further declines. Furthermore, future studies should utilize hypervariable genetic markers such as simple sequence repeats (SSRs) and single nucleotide polymorphisms (SNPs) to examine population structure, genetic diversity, and connectivity. Such targeted research is essential for informed conservation planning, monitoring the spread of newly detected species, and ensuring the long-term ecological sustainability of haplochromine cichlids in Tanzania.

ETHICAL STATEMENT

All handling of haplochromine cichlids and experimental procedures were conducted in strict accordance with the Animal Ethics Guidelines established by the Sokoine University of Agriculture Research and Ethics Committee, ensuring minimal stress and ethical compliance throughout the study.

ACKNOWLEDGEMENTS

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SUB-THEME: WILDLIFE DISEASES AND ECOSYSTEM HEALTH

PRELIMINARY SURVEY OF ECTOPARASITES ON RODENTS AND SHREWS ON MOUNT KILIMANJARO

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ABSTRACT

Rodents and shrews play a pivotal role in the transmission of zoonotic diseases, acting as reservoirs of pathogens and hosts for ectoparasites that facilitate the spread of infections posing significant risks to both wildlife and human populations. This study aimed to investigate ectoparasitic infestations in rodents and shrews along an elevational gradient on Mount Kilimanjaro, ranging from 1,700 to 3,200 meters above sea level, and to examine how altitude influences the abundance, intensity, and prevalence of ectoparasites. A total of 814 ectoparasites were recovered from 155 of the 287 rodents and shrews sampled, comprising 555 mites and 259 fleas. Among the ectoparasite species identified, mites included *Echinolaelaps echidninus* (1.09) and *Laelaps nuttalli* (0.84). Fleas were represented by *Dinopsyllus lypusus* (0.54), *Ctenophthalmus calceatus* (0.34), and *Xenopsylla brasiliensis* (0.01). Host species such as *Pelomys fallax*, *Lemniscomys zebra*, and *Mastomys natalensis* exhibited particularly high parasite loads, with some individuals reaching 100% infestation prevalence. Statistical analyses indicated no significant differences in mean ectoparasite abundance ($F_{3,44} = 2.082$, $P = 0.09$) or intensity ($F_{3,44} = 2.161$, $P = 0.08$) across elevations. However, prevalence varied significantly with altitude ($F_{3,44} = 2.885$, $P = 0.04$), and a strong negative correlation between prevalence and altitude was observed ($R^2 = 0.93$, $P = 0.03$), indicating that ectoparasite prevalence decreased with increasing elevation. The infestation indices for medically important fleas (*X. brasiliensis*, *D. lypusus*) and mites (*E. echidninus*) were 0.55 and 1.93, respectively, highlighting the potential for substantial disease transmission. These findings highlight the need for year-round surveillance to capture seasonal and climatic variations influencing host–ectoparasite–endoparasite dynamics across multiple altitudinal zones. Additionally, integrating modern molecular techniques would improve the accuracy of parasite identification and enhance the understanding of parasite diversity and transmission pathways. Such surveillance is essential for assessing the ecological and public health risks associated with parasitism in this region.

Keywords: Abundance, altitude, mites and fleas, prevalence, rodents and shrews

INTRODUCTION

Parasites contribute significantly to the diversity of life on Earth and play important roles in ecosystem dynamics (Korallo *et al.*, 2007). Among them, ectoparasites are widely distributed and regularly infest the skin of humans and other animals (Heukelbach and Feldmeier, 2004; Hanafi-Bojd *et al.*, 2007). Ectoparasites employ feeding strategies that are considered more common than many other approaches among parasites (Sukdheo and Bansemir, 1996). Rodents and shrews, being widely distributed and abundant mammals, serve as potential hosts for a variety of ectoparasites in both wild and human-influenced habitats (Battersby, 2015). The main orders of ectoparasites infesting rodents include Siphonaptera (fleas), Prostigmata (chiggers), Phthiraptera (lice), Acarina (ticks), and Mesostigmata (mites) (Nieri-Bastos *et al.*, 2004). While fleas and some mites spend most of their life in the host's nest, lice and certain gamasid mites maintain a long-term association with their hosts (Thanee *et al.*, 2009). Patterns of infestation may involve ectoparasites from the same or different taxa, and the degree of interspecific competition is believed to influence these patterns (Oguge *et al.*, 2009; Krasnov *et al.*, 2010).

The abundance and diversity of ectoparasites are influenced by both biotic and abiotic factors. External environmental factors such as temperature, altitude, vegetation, and climate can directly or indirectly affect ectoparasite populations (Edman, 2000; Randolph, 2004; Carrillo *et al.*, 2007). Biotic, host-related factors include sex (Krasnov *et al.*, 2011), reproductive condition, and host population density (Stanko *et al.*, 2002). Habitat modification, such as agricultural development or other human activities, is a major external factor that can alter ectoparasite abundance and distribution, often indirectly by affecting host behaviour and population densities (Luyon *et al.*, 2007; Hamidi *et al.*, 2015). Abiotic conditions may also affect the development and survival of ectoparasite larvae, particularly fleas (Guernier

et al., 2014). In combination, these factors facilitate species co-occurrence and influence ectoparasite community structure (Malenke *et al.*, 2011).

Rodents and shrews have been intensively studied in terms of their ecology and distribution (Mulungu *et al.*, 2008; Stanley, 2014; Thomas *et al.*, 2022a, 2022b). Despite their recognized importance as reservoirs of zoonotic pathogens and hosts of ectoparasites, no study has documented the ectoparasite fauna of small mammals in this region. This lack of baseline information limits understanding of potential host–vector interactions and the associated risk of zoonotic disease transmission. Given this knowledge gap and the potential public health implications, this study aimed to examine the abundance and distribution of ectoparasites on rodents and shrews along Mount Kilimanjaro. Specifically, we asked how ectoparasites vary among host species and along the altitudinal gradient, and assessed the potential implications for public health. By establishing foundational knowledge on ectoparasite–host dynamics, this research provides critical insights for hosts and vector surveillance in the Kilimanjaro landscape.

MATERIALS AND METHODS

Study area

The study was conducted on the western slopes of Mount Kilimanjaro (Fig. 1), located at approximately 3°07'S and 37°35'E northeastern part of Tanzania. Mount Kilimanjaro covers an area of approximately 756 km² and rises to 5,895m above sea level, making it the highest mountain in Africa (Mulungu *et al.*, 2008). Temperatures varies considerably along the elevational gradient, averaging approximately 29 °C in the lowlands and -6 °C at higher elevations. Annual rainfall ranges from 500 mm in the lowlands to approximately 2,000 mm in the highlands. The climate is classified as tropical montane, and vegetation zones are strongly structured by altitude (Hemp, 2006). The area is jointly managed by the Kilimanjaro

National Park Authority (KINAPA) and the Tanzania Forest Service(TFS), with management responsibilities varying across altitudinal zones. Distinct vegetation zones occur along the mountain's altitudinal gradient. These include cultivated farmland (<1,900 m a.s.l.), followed by plantation forests, fallow land, and scattered cultivated fields (1,900 - 2,500 m a.s.l.), montane rainforest (2,500 - 2,900 m a.s.l.), and moorland vegetation (2,900 - 3,300 m a.s.l.) (Mulungu *et al.*, 2008). Plantation and cultivated land zones are heavily influenced by human activities, including timber harvesting and the cultivation of seasonal crops such as *Pisum sativum*, *Daucus*

carota, *Solanum tuberosum*, and *Brassica oleracea* (Thomas *et al.*, 2022). Fallow land occurs in transitional areas between agricultural zones and montane rainforest and also serves as a boundary between protected areas managed by KINAPA and TFS. The montane rainforest is dominated by tree species such as *Ficus thonningii*, *Olea europaea*, *Cassipourea malosana*, and *Podocarpus latifolius*. At higher elevations, the vegetation shifts to *Erica* bushland and later transitions to *Helichrysum* species communities up to approximately 4,500 m a.s.l., accompanied by rocky and sparsely vegetated landscapes (Hemp, 2006; Thomas *et al.*, 2022).

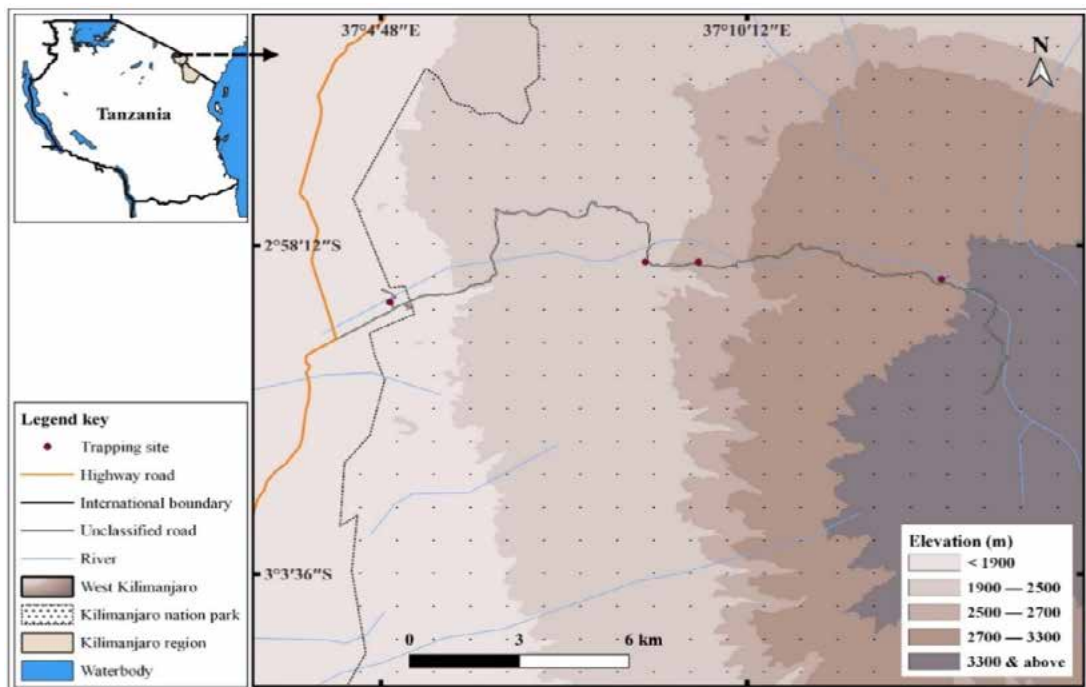


Figure 1: Map of West Mount Kilimanjaro showing study sites in the selected four elevations along the Shira route.

Study design

Sampling was conducted between April and September 2021, covering both the early rainy season and the dry season. Four sampling sites were selected along an elevational gradient: 1,700 m (farm), 2 400 m (fallow), 2,700 m (forest), and 3,200 m (moorland). Sites were selected based on altitude and dominant

vegetation type, and their locations were determined using Global Positioning System (GPS) coordinates.

At each site, five trap lines were established. Each trap line was 50 m long, with 10 m spacing between traps, resulting in a total of 50 medium-sized Sherman live traps (23 × 9.5 × 8 cm; H. B. Sherman Traps Inc., Tallahassee,

Florida, USA). Additionally, two Havahart traps (60 × 15 × 170 cm) were randomly placed at each site. Sherman traps were baited with a mixture of peanut butter and maize flour, while Havahart traps were baited with banana, fresh maize, and avocado to increase capture success. Traps were set in the evening and inspected the following morning at 07:00 h, after which they were re-baited. Captured rodents were identified using external morphology and morphometric measurements following the taxonomic keys provided by Happold (2013).

Ectoparasites collection, processing and identification

Captured rodents and shrews were lightly anesthetized using diethyl ether. Each animal was then carefully brushed with a fine toothbrush over an enamel tray to dislodge ectoparasites. Special attention was given to body regions commonly harboring ectoparasites, including axillary regions, groin, tail base, genital and anal areas, behind the ears, and under the chin. Dislodged ectoparasites were collected using a camel hair brush and preserved in 70% ethanol in labeled sample tubes. All specimens were transported to the Institute of Pest Management Centre (IPM) at Sokoine University of Agriculture for laboratory processing and identification.

Flea Preparation and identification

Fleas were cleared by boiling in 10% potassium hydroxide (KOH) for 10 minutes, followed by immersion in cold water for 1 hour. Specimens were then transferred into acidified distilled water (distilled water containing a drop of glacial acetic acid) for 30 minutes. After rinsing, specimens were dehydrated through an ethanol series:

50% ethanol (1 hour), 70% ethanol (1 hour), 95% ethanol (1 hour), absolute ethanol (1 hour). Specimens were then cleared overnight in clove oil before being mounted on microscope slides. Mounted fleas were examined under a compound microscope at 40× and 100×

magnification and identified using taxonomic keys following Smit (1957).

Mites preparation and identification

Specimens were first immersed in potassium hydroxide (KOH) to clear tissues. A concave microscope slide was then prepared, and a drop of Hoyer's medium was placed at the center. Each specimen was individually positioned in the drop with its orientation either dorsoventral or lateral, and a cover slip was placed on top. The slides were then incubated in an oven at 45–50°C for 24 hours. After incubation, additional Hoyer's medium was added, and the slides were allowed to cool. Mounted specimens were examined under a microscope for identification using taxonomic keys, illustrations, and references following Pratt (1975).

Data analysis

Data was analyzed using Microsoft excel and PAST (Paleontological Statistics) software version 4.03. Prior to analysis, total abundance and richness of both host and ectoparasites were estimated. Mean abundance, mean intensity and prevalence were used to express ectoparasites infestation on rodents and shrews. Mean abundance (M.A) was calculated as the ratio of the total number of a particular ectoparasites species in a particular rodent host to the total number of the rodent hosts examined, Mean intensity (M.I) was calculated as the ratio of total number of a particular ectoparasites species found in a sample to the number of rodent hosts infected and prevalence was calculated as the ratio of number of rodent hosts infested with a particular parasite species to the total number of rodent hosts examined for that particular parasite species (Okeke *et al.*, 2013; Dada, 2016; Maaz *et al.*, 2018). One-way analysis of variance (ANOVA) was used to compare the means in mean abundance, mean intensity and prevalence of ectoparasites among elevations. Linear regression model was used to determine the effect of altitude on mean abundance, mean intensity and prevalence of ectoparasites on rodents and shrews.

RESULTS

A total of 814 ectoparasites were collected from the four study sites. Of these, 809 specimens were recovered from rodents and nine from shrews. Five ectoparasite species were identified, including three flea species (Order Siphonaptera) and two mite species (Order Mesostigmata). Flea species included *Dinopsyllus lypusus*, *Ctenophthalmus calceatus*, and *Xenopsylla brasiliensis* whereas *Echinolaelaps echidninus* and *Laelaps nuttalli* were mites. Among these species, *E. echidninus* was the most abundant, whereas *X. brasiliensis* was the least abundant. Out of 269 rodents and 18 shrews examined, 151 rodents (52.6%) and four shrews (1.4%) were infested with at least one ectoparasite species. Mixed infestations with both fleas and mites were observed in 12.9% of rodents and 0.7% of

shrews, while 53.3% of rodents and 0.7% of shrews were infested with only one ectoparasite group. Mean abundance, prevalence (Table 1) and mean intensity (Table 2) of ectoparasites varied among rodents and shrews. While the prevalence of ectoparasites on rodent and shrews differed significantly between elevations ($F_{3,44} = 2.885$, $P=0.04$), no significant differences among elevations was observed in mean abundance ($F_{3,44} = 2.309$, $P=0.08$) and mean intensity ($F_{3,44} = 2.461$, $P=0.07$). Unlike in mean abundance and mean intensity, elevation showed strong negative relationship with prevalence of ectoparasites ($R^2 = -0.93$, $P = 0.03$) (Table 3). Generally, mites showed high ectoparasite infestation than fleas in all hosts examined, and male hosts (53.5%) showed higher prevalence of ectoparasites than females (45.5%).

Table 1: Mean abundance (M.A) and prevalence (P %) of ectoparasites on rodents and shrews on Mount Kilimanjaro

ECTOPARASITES	HOST SPECIES																							
	RODENT																						SHREW	
	<i>M. natalensis</i>	<i>L. zebra</i>	<i>R. dilectus</i>	<i>P. fallax</i>	<i>O. angolensis</i>	<i>L. verhegenii</i>	<i>D. mesomelas</i>	<i>G. dolichurus</i>	<i>M. minutoides</i>	<i>P. delectorum</i>	<i>G. murinus</i>	<i>C. Oliveri</i>												
<i>X. brasiliensis</i>	0.06	6.5	—	—	0.02	1.5	—	—	—	—	0.03	3	—	—	—	—	—	—	—	—	—	—	—	—
<i>C. calceatus</i>	0.03	3.2	2.4	100	0.24	16.9	0.42	28.6	0.14	14.3	0.73	45.4	0.13	13.04	0.5	30	0.5	25	0.34	20.8	0.43	28.6	0.17	20
<i>D. lypusus</i>	1.77	67.7	0.2	20	0.19	10.8	0.28	28.6	1.43	43	0.48	21	0.43	30.4	0.5	30	—	—	0.51	30.2	0.28	28.6	0.17	30
<i>E. echidninus</i>	6.8	83.9	4.4	40	0.06	3.1	8	28	—	—	0.06	3	—	—	—	—	—	—	0.09	9.1	—	—	—	—
<i>L. nuttalli</i>	3.29	58.1	8.2	80	0.4	13.7	9.07	85.9	—	—	0.03	3	—	—	—	—	—	—	0.12	9.1	—	—	0.17	10
TOTAL	11.95	93.5	15.2	100	0.91	32.3	17.77	100	1.57	57.14	1.33	60.6	0.56	34.8	1	40	0.5	25	1.06	49.4	0.71	42.9	0.51	8.33

Table 2: Mean intensity of ectoparasites on rodents and shrews on Mount Kilimanjaro

ECTOPARASITES	HOST SPECIES											
	RODENT											SHREW
	<i>M. natalensis</i>	<i>L. zebra</i>	<i>R. dilectus</i>	<i>P. fallax</i>	<i>O. angoniensis</i>	<i>L. verhegenii</i>	<i>D. mesomelus</i>	<i>G. dolichurus</i>	<i>M. minutoides</i>	<i>P. delectorum</i>	<i>G. murinus</i>	<i>C. oliveri</i>
	MEAN INTENSITY											
<i>X. brasiliensis</i>	0.06	—	0.05	—	—	0.05	—	—	—	—	—	—
<i>C. calceatus</i>	0.03	2.4	0.75	0.73	0.25	1.2	0.38	1.25	2	0.7	1	0.21
<i>D. lypusus</i>	1.9	0.2	0.52	0.28	2.5	0.8	1.13	1.25	—	1.02	0.67	0.21
<i>E. echidninus</i>	7.28	4.4	0.2	8	—	0.1	—	—	—	0.18	—	—
<i>L. nutalli</i>	3.51	8.2	0.87	9.07	—	0.05	—	—	—	0.26	—	0.21
TOTAL	12.78	15.2	2.39	18.08	2.75	2.2	1.51	2.5	2	2.16	1.67	0.63

Table 3: Relationship between altitude and mean abundance, mean intensity and prevalence of ectoparasites (Linear regression results)

Variable	R ²	F	P
Mean abundance	-0.71	4.81	0.16
Mean intensity	-0.67	4.05	0.18
Prevalence	-0.93	29.8	0.03

Table 3: Overall indices/ mean abundance of fleas and mites in rodent and shrews on Mount Kilimanjaro

Rodent species	<i>X.brasiliensis</i>	<i>D.lypusus</i>	<i>C. calceatus</i>	<i>E. echidninus</i>	<i>L. nuttali</i>
<i>M. natalensis</i>	0.06	1.77	0.03	6.81	3.29
<i>L. zebra</i>	—	0.2	2.4	4.4	8.2
<i>R. dilectus</i>	0.02	0.18	0.25	0.06	0.29
<i>P. fallax</i>	—	0.29	0.43	8	9.14
<i>O. angoniensis</i>	—	1.43	0.14	—	—
<i>L. verhageni</i>	0.03	0.48	0.73	0.06	0.03
<i>D. insignis</i>	—	0.43	0.13	—	—
<i>G. dolichurus</i>	—	0.5	0.5	—	—
<i>M. minutoides</i>	—	—	0.5	—	—
<i>M. delectorum</i>	—	0.51	0.35	0.27	0.18
<i>G. murinus</i>	—	0.29	0.43	—	—
SHREW SPECIES					
<i>C. Olivieri</i>	—	0.17	0.17	—	0.17
Subtotal	0.01	0.54	0.34	1.09	0.84
TOTAL	0.55		0.34	1.93	

DISCUSSION

This study identified *Echinolaelaps echidninus*, *Laelaps nuttalli*, *Ctenophthalmus alceatus*, *Xenopsylla brasiliensis*, and *Dinopsyllus lypusus* as the principal ectoparasites infesting rodents and shrews along the elevational gradient of Mount Kilimanjaro in northeastern Tanzania. Notably, *X. brasiliensis* and *D. lypusus* are recognized vectors of zoonotic diseases such as plague, and their occurrence in the study area may pose significant public health risks. Previous studies conducted in northern Tanzania (Haule *et al.*, 2013; Makundi *et al.*, 2015; Mawanda *et al.*, 2020) similarly confirmed the role of these flea species in the transmission of plague, particularly when their populations increase and their associations with rodent hosts reach epidemiologically significant levels. Although *X. brasiliensis* is recognised as one of the dominant plague vectors among sylvatic rodents in sub-Saharan Africa, its relatively low abundance in this study may be attributed to its ecological behavior. This flea species spends much of its life cycle within rodent nests rather than directly on the host (Kilonzo *et al.*, 2006). Occasional infestations observed on *Mastomys natalensis*, *Lophuromys verhageni*, and *Rhabdomys dilectus* suggest a high degree of host-switching potential, reflecting the mobility and ecological adaptability of this flea across different rodent species.

High mean abundance, prevalence, and intensity of ectoparasites recorded on *Pelomys fallax*, *Lemniscomys zebra*, and *M. natalensis* may be associated with agricultural disturbances in the areas where these species were captured. Habitat modification resulting from cultivation often reduces vegetation complexity and alters microhabitat conditions, potentially facilitating parasite persistence and transmission. Similar patterns have been reported by Khajeh *et al.* (2017), who observed increased parasite infestation in disturbed habitats. In contrast, the lower ectoparasite loads recorded on *Crociodura*

olivieri and *R. dilectus* may be associated with cooler microclimatic conditions prevailing in their habitats, which are less favorable for the development and survival of ectoparasites. Temperature and humidity are known to strongly influence the survival, reproduction, and activity patterns of ectoparasites.

The exclusive occurrence of *D. lypusus*, *C. calceatus*, and *X. brasiliensis* on *M. natalensis* within cultivated areas indicates an elevated zoonotic risk in these environments. This rodent species is well known for its synanthropic behavior and frequent proximity to human settlements. Similar findings by Zimba *et al.* (2011) highlighted the role of *M. natalensis* as a potential reservoir host for *Yersinia pestis*, the causative agent of plague. The predominance of Mesostigmata mites, particularly *L. nuttalli* and *E. echidninus*, may reflect their strong host–parasite associations and relatively permanent parasitic lifestyle. In contrast to fleas, which are generally more mobile and capable of leaving hosts when disturbed, many mite species remain closely associated with their hosts for extended periods. Comparable patterns of mite dominance have been reported in rodent ectoparasite communities in Southeast Asia (Madinah *et al.*, 2011; Ng *et al.*, 2017).

Among the flea species identified, *D. lypusus* showed the highest overall abundance. This may be linked to favorable environmental conditions within the study area. Previous studies (Njunwa *et al.*, 1989; Makundi *et al.*, 2003; Makundi *et al.*, 2008) reported that populations of *D. lypusus* thrive at temperatures ranging between 10 °C and 30 °C, conditions that correspond closely to those recorded in the mid-elevational zones of Mount Kilimanjaro. The occurrence of multiple ectoparasite taxa on individual hosts suggests the presence of polyparasitism, which may be influenced by host-related factors such as immune status and physiological condition. Hosts with compromised immune defenses may be more susceptible to infestation by multiple parasite species. Similar observations have been

documented in previous studies (Kia *et al.*, 2009; Changbunjong *et al.*, 2010; Wei *et al.*, 2010), where rodents frequently carried several ectoparasite taxa simultaneously.

Ectoparasite infestation was slightly higher in male hosts than in females, possibly due to the larger body size and wider home ranges typically observed in male rodents, which increase their exposure to parasites. Similar sex-biased infestation patterns have been reported in earlier studies (Krasnov *et al.*, 2005; Rynkiewicz *et al.*, 2013). While earlier studies (Luyon *et al.*, 2007; Bendek *et al.*, 2013) reported no significant relationship between altitude and ectoparasite infestation, the present findings support studies by Meliyo *et al.* (2014) and Eisen *et al.* (2017), which demonstrated that altitude can significantly influence the prevalence and composition of ectoparasite communities in rodent populations on Mount Kilimanjaro.

CONCLUSION AND RECOMMENDATION

This study provides important insights into the abundance, and distribution of ectoparasites infesting rodents and shrews along the elevation gradient of Mount Kilimanjaro. The identified taxa *Echinolaelaps echidninus*, *Laelaps nuttali*, *Ctenophthalmus calceatus*, *Xenopsylla brasiliensis*, and *Dinopsyllus lypus* showed marked variation in prevalence and abundance influenced by habitat type, host species, and altitude. The dominance of *Mastomys natalensis* as a host for key flea species, particularly *X. brasiliensis* and *D. lypus*, highlights its potential role as a reservoir of zoonotic pathogens such as *Yersinia pestis*, the causative agent of plague. The dominance of *Mastomys natalensis* as a host for medically important flea species, particularly *X. brasiliensis* and *D. lypus*, highlights its potential role as a reservoir host for zoonotic pathogens such as *Yersinia pestis*, the causative agent of plague. Given the presence of these vector species within the Kilimanjaro ecosystem, there is a clear need

to strengthen rodent and vector surveillance programs in the region. Continuous ecological monitoring, combined with molecular screening of ectoparasites for zoonotic pathogens, would significantly improve understanding of disease-risk dynamics within the rodent–vector system. Future research should also incorporate seasonal sampling and detailed microclimatic measurements to better capture temporal variability in ectoparasite–host interactions across different elevational zones.

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WHAT DOES THE RESEARCHER NEED TO CONSIDER WHEN PLANNING FOR CAPTURE AND COLLARING OF CARNIVORES IN NYERERE NATIONAL PARK AND SELOUS GAME RESERVE?

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ABSTRACT

The Nyerere National Park (NNP) and Selous Game Reserve (SGR) form a critical component of the Nyerere–Selous–Mikumi Ecosystem (NSME) in Tanzania, the largest protected ecosystem in the country and one of the most significant remaining wilderness landscapes globally. This ecosystem represents an important stronghold for endangered African wild dog (*Lycaon pictus*) populations and supports substantial numbers of African lion (*Panthera leo*) and spotted hyena (*Crocuta crocuta*). Chemical immobilization and collaring are essential tools for ecological and behavioral research on these large carnivores; however, the environmental and logistical conditions of the NSME present unique operational challenges. Drawing on extensive field experience, this study provides a practical guide outlining key considerations for researchers planning immobilization and collaring operations for wild dogs and lions in this ecosystem. Critical factors influencing capture success include seasonality (wet versus dry seasons), time of day (night captures for lions and early morning captures for wild dogs), appropriate drug combinations and dosages, the presence and density of competing predators such as hyenas at capture sites, and the expected capture success rate per individual per day. Our findings indicate that the anesthetic combination of Tiletamine–Zolazepam (Zoletil) should not be used alone for immobilizing wild dogs in this ecosystem; instead, it is more effective when combined with a sedative such as Medetomidine. In addition, capturing a target number of wild dogs requires approximately twice as much time as capturing lions due to differences in social behavior, movement patterns, and habitat use. The dry sub-humid climate of the NSME, characterized by high ambient temperatures (up to 41 °C) and an average annual rainfall of approximately 1,025 mm, strongly influences animal physiology, anesthetic performance, and field accessibility. A comprehensive understanding of these ecological and operational factors during project planning is essential for maximizing capture success, ensuring the safety of research personnel, safeguarding equipment, and minimizing risks to target animals.

Keywords: *Panthera leo*, *Lycaon pictus*, Chemical immobilization, Nyerere-Selous, Field methodology

INTRODUCTION

Large carnivores are widely recognized as flagship species for biodiversity conservation and play a critical role in maintaining ecosystem structure and function through top-down ecological regulation (Ripple *et al.*, 2014). However, their wide-ranging behavior, low population densities, and often remote habitats make them difficult to study. In the

extensive and remote landscapes of sub-Saharan Africa, Global Positioning System (GPS) collaring has become an indispensable tool for investigating movement ecology, population dynamics, social organization, and human–carnivore interactions (Kaczensky, 1999; Davoli *et al.*, 2023). The Nyerere–Selous–Mikumi Ecosystem (NSME) in southern Tanzania represents one of Africa’s largest and most important protected

landscapes, covering more than 50,000 km². The ecosystem includes Nyerere National Park and Selous Game Reserve, forming a globally significant wilderness area (UNESCO, 2022). This ecosystem is a major stronghold for the endangered African wild dog (*Lycaon pictus*) and also supports substantial populations of African lion (*Panthera leo*) and spotted hyena (*Crocuta crocuta*) (TAWIRI, 2016).

Although the principles of wildlife chemical immobilization are well established (Kreeger *et al.*, 2002; Arnemo & Kreeger, 2018), the success and safety of capture operations are strongly influenced by local ecological and environmental conditions. Factors such as climate, terrain, vegetation structure, and species behaviour vary considerably among ecosystems and can significantly affect animal physiology, drug efficacy, and operational logistics. The NSME presents a unique operational context characterized by a dry sub-humid climate, high temperatures, and pronounced seasonal variations in rainfall and accessibility. These environmental conditions directly influence capture feasibility, anesthetic performance, and animal welfare during immobilization procedures.

Despite the conservation importance of this ecosystem, detailed practical guidance on carnivore capture operations within the NSME remains limited in the scientific literature. This study aims to address this gap by synthesizing field experience from multiple capture campaigns involving lions and wild dogs in Nyerere National Park and Selous Game Reserve. We highlight key operational considerations that can improve capture success while minimizing risks to animals and research teams

MATERIALS AND METHODS

Study Area

The study was conducted in Nyerere National Park (29,276 km²), Selous Game Reserve (18,020 km²), and surrounding

areas including Ingirito village in Kilwa District. The ecosystem consists of a mosaic of habitats, including miombo woodland, savanna grasslands, and riverine forests. The climate is dry sub-humid, with a wet season from November to May and a dry season from June to October. Average annual rainfall is approximately 1,025 mm, and daily temperatures frequently exceed 35 °C, reaching up to 41 °C during the hot dry season.

Methods

This study is based on a retrospective analysis of capture operations conducted between 2021 and 2025 as part of research and wildlife management activities. These operations involved immobilization of lions and wild dogs for GPS collaring, veterinary treatment, snare removal, and translocation for human–lion conflict mitigation. Data were compiled from veterinary reports and immobilization records covering 78 successful capture events, involving 59 lions and 19 wild dogs. Of these, 27 lions and 14 wild dogs were fitted with GPS collars, and two individuals were translocated.

Capture and Immobilization Protocols

Capture operations were conducted by multidisciplinary teams consisting of a wildlife veterinarian, field ecologists, and experienced park rangers. Animals were located using several approaches, including VHF telemetry tracking of previously collared individuals, opportunistic sightings, spoor tracking, information from rangers, guides, or hunting camps. Acoustic call-ins were frequently used to attract animals. The call-in system consisted of speakers connected to a vehicle battery and amplifier. Prey distress calls (e.g., buffalo or warthog) or lion vocalizations were used to attract lions, whereas wild dogs were attracted using recorded pack vocalizations such as greeting or hunting calls.

Immobilization Drugs: The primary immobilization agents used were Tiletamine–

Zolazepam (Zoletil®, Virbac) and Ketamine (V-Tech®, Novecy Pharmaceutical). For lions, we used Zoletil at total dosages of 100-120mg for females and 130-150mg for males, while for the Ketamine 200-250mg for females and 250-400mg for males. For wild dogs, Zoletil was used at 60-100mg without sedative, and 50-60mg when combined with a sedative, while for Ketamine we used 50-60mg total doses. The Medetomidine (V-Tech®, Novecy Pharmaceutical), as a sedative was used at a dose of 2-4 mg for wild dogs and 4-6 mg for lions.

Delivery System: Drugs were delivered via remote injection using dart rifles (DAN-INJECT® or Model 389 Pneu-Dart® Projector). Darting was performed from a vehicle.

Monitoring: Once immobilized, animals were monitored for vital signs (respiration rate, capillary refill time, body temperature). Shade was provided, and water was poured on the animal's body to mitigate hyperthermia. Procedures (physical examination, morphometric measurements, GPS collar fitting) were conducted as swiftly as possible.

Reversal: For both lions and wild dogs where Medetomidine was used as sedative, reversal was achieved with Atipamezole (V-Tech®, Novecy Pharmaceutical) at 2-3 times the Medetomidine dose, administered intramuscularly.

Data collection

Data recorded for each event included species, date, time of day, season, drug combination and dosage, induction time, procedure duration, samples collected, number of competing predators in the vicinity, any complications and post-recovery behaviours.

RESULTS AND DISCUSSION

Field experience revealed several factors that significantly influence the success and safety of carnivore capture operations in the NSME.

Season strongly influenced capture logistics and animal physiology. The dry season which is between June and October generally provides the best conditions for capture operations. Reduced vegetation improves visibility for tracking and darting, and road access is better. However, extremely high temperatures increase the risk of capture-related hyperthermia. During immobilization, animals experience a strong sympathetic stress response, releasing catecholamines such as adrenaline that elevate heart rate, blood pressure, and metabolic heat production (Cattet *et al.*, 2008). Capture operations were therefore conducted during cooler periods, typically between 05:00 and 10:00.

Capture during the wet season starting from November to May, is logistically difficult due to flooded rivers and impassable roads. Dense vegetation also reduces visibility. Consequently, capture operations during this season were generally restricted to emergency interventions such as rescue or veterinary treatment.

Species-specific activity patterns strongly influenced capture success. As primarily crepuscular and nocturnal predators, lions were most effectively captured during evening and night operations using call-ins and spotlights with red filters. Night operations using spotlights are highly effective for lions, though in most cases the call-ins were used to attract them. While calling, the night-vision goggle was used to screen the animals as the call-in not only attract the lions but also in most cases the other predators like hyenas too. Once the lion is identified by the call-in team on the roof of the vehicle, they inform the vet in the vehicle using the UHF radio call. Once the lion approaches the vehicle the call-in team shine the spotlight covered with red filter to the targeted animal so that the vet can dart. Using the red filtered spotlight, the lions are more easily approached and are less likely to be lost in thick vegetation after darting as they feel secure and not running away. However, the time of the day did not prevent us to capture the lions, as about 25% of captured lions were captured during the day time.

African Wild Dogs are primarily diurnal cursorial hunters. The early morning hours are the most successful time for capture. Packs are often active after the night's rest, and the cooler temperatures reduce the risk of hyperthermia during immobilization. Attempting captures in the heat of the day is discouraged since it increases the risk of hyperthermia. Once we found spoors or encountered them in a restless situation we used to call them, however in this situation the call-in team and the darting team stayed in separate vehicles in a L-shaped positions. Therefore, as the dogs approach the call-in vehicle, the darting team gain good visibility for target selection and darting site. The choice of immobilizing agents is critical for animal safety, especially in a hot climate. We strongly advise against using Tiletamine-Zolazepam (Zoletil) alone in this ecosystem as recommended by Kock *et al*, (2006), though in other ecosystems like Katavi-Rukwa, and Ruaha-Rungwa we managed to capture the wild dogs using that protocol at the total dose of 60-80mg.

In NSME we tried to capture wild dogs using this protocol at different doses with poor result. At 60mg the female wild dog expressed initial signs of effects of anaesthetics including isolation from the pack and staggering, which persisted for about 20 minutes before recovering. This also observed at 80mg Zoletil in male wild dogs, which persisted for about 30-40 minutes before recovering. Then, we managed to capture the male wild dog with inadequate sedation at 100mg Zoletil, however, this ends with recurring complications of hyperthermia. Though hyperthermia was managed and animal released, on the second day was found dead, which revealed to be a result of heat stroke on necropsy. The combination of Zoletil and Medetomidine at 50mg and 2-3mg respectively produced good results though to some males (3) there were abrupt recoveries without antidote. At the doses of 50mg Zoletil and 4mg Medetomidine there was smoother induction, though during recovery it was rough as always Zoletil does. It is important to know that Tiletamine, one component of Zoletil, can cause a slight

increase in muscle activity and metabolic rate, which is problematic in an animal already on the brink of hyperthermia. While a higher dose doesn't mitigate this risk, a dose that is too low can be even more dangerous; an animal that is partially immobilized may struggle violently, generating far more body heat than one that is fully anesthetized. Therefore, the higher dose is also a welfare consideration to ensure a rapid and smooth induction, minimizing the period of exertional heat generation and allowing the veterinary team to quickly intervene with cooling measures (e.g., water, shade) (Kreeger & Arnemo, 2012). However, after some other attempts of using Ketamine at 50-60mg with medetomidine 3-4mg, we found that at 60mg Ketamine and 4mg Medetomidine, the combination provides smoother, more controllable induction (down in 5 minutes), better muscle relaxation, and a rapid, calm recovery after reversal, which was essential for ensuring the animal can rejoin its pack quickly.

African Lions are generally more robust and tolerate Zoletil well with good knocking down effect when combined with sedative. In our case the doses of Zoletil at 100-120mg for females and 130-150mg for males with 4-6mg Medetomidine provided smoother induction. However, dosages must be carefully calculated based on estimated weight, since in two males with measurements of 150-160cm from back of the skull to start of tail required a top-up of 100-150mg Ketamine with initial dose of 150mg Zoletil and 6mg Medetomidine. For the case of Ketamine as the main drug, the dose of 200-250mg for females and 250-400mg for males with 4-6mg Medetomidine provided smoother induction. However, in four females the lowest dose of 130mg, 150mg, 150mg, and 160mg Ketamine undergone partial anaesthesia which required a top-up of 100mg, 50mg, 70mg, and 70mg Ketamine respectively. On the other hand, one male darted with 250mg Ketamine expressed partial anaesthesia which was managed with additional 150mg Ketamine. Some challenges observed with Ketamine was delayed induction of up to 20 minutes and hallucinations with roaring while in deep anaesthesia in 2 lionesses and 1 lion.

Since, we aimed at smoother induction with high knock-down effects and security purposes, Zoletil at 120mg for females and 150mg for males with 6mg Medetomidine was the best with 8 minutes induction time in about 80% of animals captured with this dose. However, in two exception cases of capturing male lions in human settlement a dose of 170mg Zoletil and 7mg Medetomidine was used. The higher the dose, the shorter induction time, hence minimizing the risk of animal lost in narrow space and ensure the security of the capture team.

The high density of spotted hyaenas in the NSME poses a significant consideration at calling point. Hyaenas are attracted to capture sites, they come to calling point aim potentially scavenging on carcasses used as bait or attacking immobilized animals. We recorded a 100% incidence of hyaena presence at calling point where bait was used, with a maximum number of up to 35 in a point. About 30% of all calling points for lions the hyenas appeared in less than 2 minutes after the first call-in, and where the number of hyenas exceeded 4 there was a less chance of lion to appear at calling point. On the other hand, the areas where the packs of wild dogs were encountered there was no hyenas observed, except in one location where we encountered a pack of four females in a dispersal stage we observed one hyena. This has been described clearly in some studies conducted in Serengeti (Frame, 1981; Fanshawe & Fitzgibbon, 1993; Swanson *et al.*, 2014), that the competition from hyenas has negative effects to African wild dogs and lions when it comes the protection of their killings. Therefore, the lions and wild dogs they avoid areas with large number of hyenas, though the wild dogs are the mostly affected by increased populations of lions and hyenas. The capture team must protect the immobilized animal until is on full recovery and/or with other pride or pack members to protect. Leaving an immobilized animal unattended, even briefly, is unacceptable.

The social structure, population density and

behaviour of the species directly influence the effort required. Lions are often easier to locate and approach, when compared to wild dogs. Capturing wild dogs is inherently more time-consuming as our data indicates that capturing the same target number of animals takes approximately twice as long for wild dogs compared to lions. It can take a maximum of two weeks to locate, capture and collar a pack of wild dogs. This is due to their extensive daily movements, and the challenge of approaching a moving pack closely enough for a safe dart shot. However, in SGR you can spend shorter time to locate and collar lions and wild dogs compared to NNP.

CONCLUSION AND RECOMMENDATIONS

The unique environmental and ecological context of the Nyerere-Selous complex demands a tailored approach to carnivore capture and collaring. Success is not merely a function of technical skill but of meticulous planning that accounts for seasonality, species-specific ethology, and the local climate's impact on drug efficacy and animal physiology. We recommend that researchers planning such work in this region should schedule field campaigns for the dry season to maximize accessibility and visibility, plan operations according to species-specific activity patterns (night and day for lions, early morning for wild dogs), avoid using Tiletamine-Zolazepam alone for wild dogs; instead, adopt a combination protocol with sedative for safer induction and recovery, always incorporate a predator deterrence plan into the operational protocol to mitigate the risk from spotted hyaenas, and allocate sufficient time and resources, understanding that capturing wild dogs is a more logistically challenging and time-intensive endeavour than capturing lions. By integrating these considerations into project design, researchers can significantly increase the likelihood of a successful, ethical, and safe capture campaign, thereby generating valuable data for the conservation of these iconic carnivores in one of Africa's last great wildernesses.

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SURVEILLANCE OF ZONOTIC DISEASES IN WILD CHIMPANZEE POPULATIONS: LESSONS FROM THE GREATER GOMBE ECOSYSTEM, TANZANIA

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ABSTRACT

Over the years, infectious diseases have been identified as a significant threat to the population. Historical outbreaks, including polio and recurrent respiratory events, have increased chimpanzee mortality and jeopardized their long-term survival. Recognizing the likely zoonotic origin of many outbreaks, the Gombe Ecohealth Project was established in 2004 under Jane Goodall Institute. This initiative employs non-invasive health surveillance through quarterly fecal sample collection, which is analyzed using molecular techniques, including PCR and sequencing, for samples that require further analysis to detect a wide range of pathogens. This is done onsite in Gombe National Park in the established molecular Laboratory, facilitating timely access to actionable data for local conservation action. Integrating clinical observations and post-mortem examinations, the project has successfully established a comprehensive health baseline for the population. Our findings have identified numerous known zoonotic pathogens in chimpanzee communities (e.g., *Shigella* spp, with prevalence of 6.2 % (95% CI: 0.049 - 0.079)), highlighting the critical role of a One Health approach in early outbreak detection, which has helped to quickly react and prevent the further spread of an outbreak to healthy individuals and prevent loss of life. Ultimately, this long-term surveillance program underscores the importance of continuous health monitoring in wild primate populations and provides a valuable model for preventing spillover events that threaten both wildlife conservation and human public health.

Keywords: Gombe National Park, One Health, Zoonotic Diseases.

INTRODUCTION

The emergence and re-emergence of infectious diseases, particularly those of zoonotic origin defined as diseases transmitted between animals and humans, pose a significant threat to global public health and biodiversity conservation. Approximately 60.3% of all emerging human infectious diseases are zoonotic (Jones *et al.*, 2008), highlighting the importance of integrated disease surveillance across human,

animal, and environmental systems. The One Health approach, which recognizes the interconnectedness of human, animal, and ecosystem health, provides a critical framework for preventing pathogen spillover events at the human–wildlife interface. Great apes, specifically chimpanzees (*Pan troglodytes*), are highly vulnerable to infectious disease from humans for two primary reasons. First, they share more than 98.8% of their genome with humans, making them susceptible to

many of the same pathogens, often resulting in severe or fatal outcomes (Williams *et al.*, 2008). Second, increasing anthropogenic encroachment, including habitat fragmentation and proximity to human settlements, livestock, and tourism activities, elevates the frequency of contact, thereby increasing the risk of cross-species transmission (Zimmerman *et al.*, 2022). Consequently, infectious diseases are now a leading non-poaching cause of mortality and a major threat to great ape conservation (Williams *et al.*, 2008; Lonsdorf *et al.*, 2022).

The Greater Gombe Ecosystem in Western Tanzania is a uniquely important site for studying these dynamics. Research started in 1960 by Dr. Jane Goodall, the long-term research at Gombe has provided unmatched historical insights into pathogen threats to chimpanzees. Documented outbreaks, such as the devastating Polio epidemic of 1966 and frequent, high-mortality respiratory diseases over the decades, like the 1980s, 1990s, and 2000s, have shown that disease significantly limits population growth and survival (Goodall, 1983, 1986; Nutter, 1996; Mlengeya, 2000). Additionally, research from this ecosystem was crucial in understanding the Simian Immunodeficiency Virus (SIVcpz) found in chimpanzees, the ancestor of human HIV, emphasizing Gombe's role as a sentinel site for understanding zoonotic risks (Sharp & Hahn, 2022).

Despite the established history of outbreaks, a major gap in the conservation framework was the lack of a proactive, standardized system to monitor for known and novel pathogens in real-time. To address this, the Gombe Ecohealth Project (GEHP) was formally established in 2004, designed specifically to implement a long-term, non-invasive health surveillance program (Lonsdorf *et al.*, 2006; Travis *et al.*, 2008). A key innovation of the GEHP was the development of an on-site molecular diagnostic laboratory, enabling rapid sample analysis and immediate feedback to conservation managers.

This study outlines the operational model and key findings from 21 years of continuous surveillance under the GEHP. We start by describing the integrated methodology that combines systematic non-invasive sampling, clinical monitoring, and rapid molecular diagnostics, which has been essential to the project's success. Next, we highlight important findings, including the prevalence of select zoonotic pathogens such as *Shigella* spp. (6.24%) within the chimpanzee populations. Ultimately, this work shares the lessons learned from the Gombe model, demonstrating the vital role of ongoing, adaptive, and technology-driven surveillance in early outbreak detection and response. The success of the GEHP offers a template for addressing zoonotic and reverse zoonotic threats, providing actionable insights for the conservation of great apes and the protection of global public health through the One Health approach.

METHODOLOGY

This study describes the establishment and operation of a long-term, non-invasive health surveillance system targeting wild chimpanzees within the Greater Gombe Ecosystem. The surveillance strategy integrates systematic field observations, non-invasive sample collection, molecular diagnostics, and a collaborative outbreak response framework.

Study Site and Target Population

The study was conducted within Gombe National Park (GNP), a protected area of approximately 35 km² located on the eastern shore of Lake Tanganyika in Western Tanzania. The ecosystem is characterized by a mosaic habitat of riverine forest, open woodland, and grassland. The target population for this surveillance program consists of the two primary habituated chimpanzee communities within the park: the Kasekela community and the Mitumba community. This surveillance effort covers the period from 2004 to 2025. The high degree of habituation of these communities allows for close observation

and direct identification of individuals during sample collection.

Health Surveillance Protocols and Sample Collection

The surveillance strategy relies on integrating systematic non-invasive sampling with continuous clinical monitoring and opportunistic post-mortem examinations.

Non-Invasive Fecal Sample Collection

Fecal samples were the primary medium for pathogen detection. Sampling was conducted quarterly throughout the study period. The protocol ensured that each collected sample was attributed to a known, individually identified chimpanzee.

Immediate Collection: Fresh fecal material was collected immediately following defecation, ensuring minimal environmental contamination.

Preservation: Samples were immediately subsampled and placed into RNALater (for nucleic acid preservation) or 10% neutral buffered formalin (for parasitological analysis) and stored in a freezer at -20°C until processing.

Clinical Health Monitoring and Post-Mortem Examination (PME).

Clinical health observations were recorded daily as part of the routine behavioural monitoring. Trained field researchers used standardized protocols to document any observable signs of illness across five main categories: gastrointestinal (e.g., diarrhea), respiratory (e.g., coughing, sneezing), body condition, dermatologic (e.g., skin lesions), and wounds/lameness (Lonsdorf *et al.*, 2018). When a chimpanzee's death was confirmed, a post-mortem examination (PME) was performed by a veterinary doctor. Tissue samples, including lung, skeletal muscle, heart, kidney, small and

large intestines, stomach, brain, liver, spleen, and lymph nodes, were collected and preserved in 10% formalin for histopathological analysis.

Laboratory Diagnostics and Pathogen Detection

A major component of this study is the operation of an on-site molecular laboratory located within Gombe National Park, which significantly reduces sample degradation and sample processing time.

Nucleic Acid Extraction and Preparation

Nucleic acids were extracted from the preserved fecal samples using Zymo Research's Quick-DNA Fecal/Soil Microbe Miniprep kit for the extraction of DNA and Quick-RNA Fecal/Soil Microbe Microprep kit for the extraction of RNA. Extracted DNA and RNA were then stored at -20 °C for molecular analysis.

Molecular Screening

Samples were initially screened using a panel of molecular assays targeting zoonotic pathogens of concern to primate conservation.

Bacterial Pathogen: Detection of *Shigella* spp. was performed using the Polymerase Chain Reaction (PCR) protocol, targeting the *ipaH* gene. Specific primer pairs and PCR conditions were used. **Wider Pathogen Panel:** In addition, samples were screened for a wide range of other pathogens, including *Streptococcus pneumoniae*, Enterohemorrhagic *Escherichia coli*, *Salmonella enterica*, *Campylobacter* spp., *Entamoeba histolytica*, *Cryptosporidium* spp., Human metapneumovirus, Respiratory syncytial virus, and Influenza A virus.

Sequencing: All PCR products confirmed as positive for *Shigella* spp. were subjected to Sanger sequencing to confirm the identity and genotype of the isolates.

Statistical Analysis

Pathogen prevalence was calculated as the number of positive individuals divided by the total number of individuals sampled during the surveillance period. 95% Confidence Intervals (CIs) for prevalence estimates were determined using the Wilson Score Interval, and all analyses were performed in R.

Outbreak Response

The timely results generated by the onsite lab are integrated into a formal Outbreak Response Protocol that embodies the One Health concept, connecting wildlife health, environmental management, and human public health.

Data Integration and Risk Assessment

Field health data (clinical observations) and laboratory data (pathogen detection) are continually merged into a centralized database. Detection of a novel pathogen or a spike in clinical signs (e.g., cluster of respiratory illness or diarrhea) triggers an immediate Risk Assessment by the surveillance team.

Intervention and Management Strategies

Upon detection of a high-risk pathogen or confirmed outbreak, the following management strategies are employed:

Isolation and Quarantine: Field teams increase distance and take all necessary precautions for affected individuals to limit anthropogenic spillover risk. Symptomatic chimpanzees are monitored closely but not approached unless necessary for emergency veterinary intervention.

Veterinary Intervention: When the risk of mortality is high, veterinary intervention is authorized, which may involve medication administration and supportive care.

RESULTS

Surveillance Coverage and Sample Acquisition

The Gombe Ecohealth Project (GEHP) has operated continuously for 25 years (2004–2025), establishing a robust baseline for health monitoring within the Gombe National Park. This has enabled testing a total of 1057 non-invasive fecal samples for *Shigella* spp from individually identified chimpanzees across the Kasekela and Mitumba communities. The samples collected were preserved in RNALater and frozen to ensure optimal integrity for downstream molecular analysis. The high frequency of sampling allowed for timely molecular screening across a panel of endemic and possible emerging zoonotic pathogens, ensuring statistical testing for prevalence estimates.

Pathogen Prevalence

Molecular analysis confirmed the presence of several enteric and respiratory pathogens, establishing baseline prevalence estimates for key pathogens within the chimpanzee population. Enteric bacterial pathogens consistently demonstrated the highest rates of detection. The overall prevalence of *Shigella* spp. was 6.24% (95% CI: 0.049-0.079) (Table 1), confirming this bacterium as a frequent and endemic threat within the ecosystem. Prevalence rates were found not to vary much by chimpanzee community, with the Kasekela community showing slightly higher detection rates (6.56% (95% CI: 0.048-0.088)) than in Mitumba (5.76% (95% CI: 0.037-0.084)) (Fig.1) A Chi-squared test of independence was performed to examine the relationship between communities (Kasekela and Mitumba) and the prevalence of *Shigella* spp. The difference in prevalence between the two communities was found to be statistically insignificant ($X^2(1, N = 1057) = 0.16, p = 0.69$).

Table 1: Positive and Negative Cases for Kasekela and Mitumba Communities.

	Positive	Negative	Total	Prevalence
Kasekela	42	598	640	6.56%
Mitumba	24	393	417	5.76%
Total	66	991	1057	6.24%

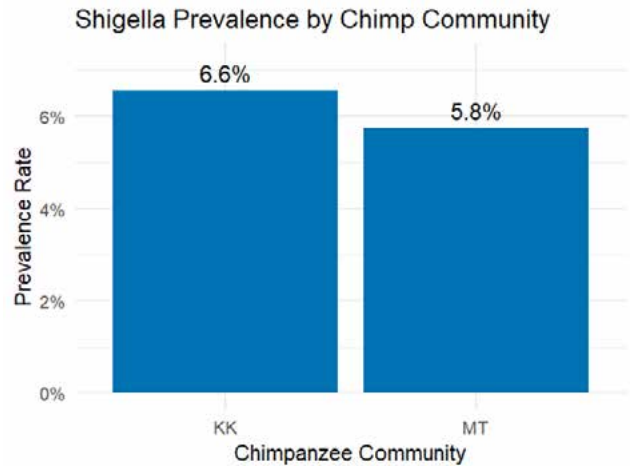


Figure 1: Shigella prevalence by Chimps

DISCUSSION

The findings from the Gombe Ecohealth Project (GEHP), spanning over two decades of continuous monitoring (2004–2025), demonstrate the value of long-term, non-invasive disease surveillance for wildlife populations at the human–wildlife interface. By integrating systematic field observations, molecular diagnostics, and veterinary assessments, the GEHP has established a robust framework for detecting and responding to infectious diseases in wild chimpanzee populations. This integrated surveillance system enables the transformation of observational health data into molecularly confirmed diagnoses, which is often difficult to achieve in remote conservation settings.

The success of the GEHP highlights the importance of transdisciplinary collaboration. By linking behavioral monitoring, veterinary science, and molecular epidemiology, the program moves beyond reactive responses to disease outbreaks and instead functions as an

early-warning system for emerging pathogens. Such surveillance is particularly critical in ecosystems like the Greater Gombe landscape, where close ecological proximity between humans, domestic animals, and wildlife increases the risk of pathogen spillover and spillback events.

The detection of *Shigella* spp. with an overall prevalence of 6.24% confirms that enteric bacterial infections represent a persistent health challenge for chimpanzees in Gombe National Park. *Shigella* species are well-known human pathogens responsible for dysentery and severe gastrointestinal illness (Mattock & Blocker, 2017), and their presence in wild chimpanzee populations suggests continuous pathogen circulation within the ecosystem. Several mechanisms may explain this persistent prevalence. First, chimpanzee-to-chimpanzee transmission is likely facilitated by their highly social behavior and frequent close contact among community members. Second, environmental contamination from human or domestic animal sources may contribute to repeated pathogen

introductions into the ecosystem. Given the proximity of surrounding human settlements and increasing anthropogenic pressures, these findings highlight the importance of improving sanitation practices and minimizing environmental contamination near protected areas. For chimpanzee conservation, the presence of endemic enteric pathogens poses a long-term threat to population viability. Gastrointestinal disease has historically been associated with morbidity and mortality in chimpanzee populations at Gombe (Terio *et al.*, 2018). Consequently, continued surveillance is essential for understanding long-term disease dynamics and identifying periods of elevated transmission risk.

One of the most significant contributions of the GEHP is the establishment of an on-site molecular diagnostic laboratory, which enables rapid detection of pathogens and timely intervention. The ability to process samples locally greatly reduces delays associated with transporting samples to distant laboratories, allowing conservation managers to respond more quickly to potential outbreaks. The importance of this capability was demonstrated during the 2024 respiratory disease cluster, in which molecular diagnostics rapidly confirmed the presence of *Streptococcus pneumoniae*. Early identification allowed field teams to implement precautionary measures, including increased distancing protocols and enhanced monitoring of symptomatic individuals. Although definitive evidence of reverse zoonotic transmission was not established, the event illustrates the critical importance of maintaining strict health protocols for researchers and park personnel interacting with habituated primate populations. Rapid diagnostic capacity therefore serves not only as a research tool but also as an operational management component of conservation programs, enabling adaptive responses that reduce mortality risk during disease outbreaks.

The results of this study reinforce the importance of adopting a One Health framework, which recognizes the interconnected health of

humans, animals, and ecosystems. Infectious diseases affecting wildlife populations rarely occur in isolation; instead, they often emerge from complex ecological interactions involving multiple species and environmental factors. In the Greater Gombe Ecosystem, chimpanzees share space with human communities, livestock, and other wildlife species, creating opportunities for cross-species pathogen transmission. The GEHP model demonstrates how collaborative approaches between conservation organizations, veterinary teams, and public health authorities can strengthen disease prevention strategies. Future surveillance efforts would benefit from stronger integration with regional public health monitoring programs, enabling coordinated detection and response to pathogens that affect both humans and wildlife. Such collaboration could improve understanding of transmission pathways and facilitate interventions aimed at reducing zoonotic and reverse-zoonotic risks.

Although the GEHP has generated valuable insights into chimpanzee health over the past two decades, several opportunities remain to expand its impact. First, incorporating metagenomic sequencing approaches would allow the detection of previously unknown pathogens and provide a broader understanding of microbial communities within chimpanzee populations. Such methods are increasingly important for identifying emerging infectious diseases before they cause large-scale outbreaks. Second, strengthening community engagement and public health education programs in surrounding villages could significantly reduce pathogen transmission risks. Improving sanitation infrastructure, increasing awareness of disease transmission pathways, and promoting responsible waste management practices may help minimize environmental contamination near chimpanzee habitats.

Finally, continued long-term monitoring will remain essential. Disease dynamics often fluctuate over time in response to environmental conditions, demographic changes, and human activities. Sustained surveillance programs like

the GEHP provide the longitudinal datasets required to detect these trends and guide adaptive conservation strategies.

CONCLUSION

The Gombe Ecohealth Project (GEHP) stands as an example for a successful, long-term health surveillance at a critical wildlife interface. The 21 years of continuous, non-invasive monitoring (2004–2025) within the Greater Gombe Ecosystem have not only established a vital epidemiological baseline for the critically endangered wild chimpanzee population but have also provided the methodological framework for effective, real-time outbreak response under the One Health mandate. The primary ecological and conservation finding of this study is the confirmation of endemic circulation of *Shigella* spp., detected with an overall prevalence of 6.24% in the chimpanzee communities. This high, persistent rate demonstrates the severity of ongoing pathogen exposure and identifies enteric disease as a major, continued threat to chimpanzee survival. Crucially, the system shows its ability to mitigate acute risks, evidenced by the rapid detection and containment of the 2024 *Streptococcus pneumoniae* outbreak. This success highlights the power of integrating standardized clinical field observations with an on-site molecular diagnostic laboratory, enabling quick confirmation of the cause of a particular disease and implementation of immediate adaptive management interventions. Ultimately, the GEHP model provides a robust template for the conservation of great apes worldwide. It demonstrates that the key to safeguarding these populations lies not only in physical protection but in the proactive, transdisciplinary surveillance of infectious diseases. Moving forward, the continued operational strength of this model and its expansion, both in diagnostic scope (e.g., metagenomics for unknown agents) and in community collaboration, remain essential to mitigating the dynamic risks to a rapidly evolving human-wildlife interface. Continuous, integrated surveillance is not just a scientific endeavor; it is a fundamental

requirement for the long-term survival of the Gombe chimpanzees and a core strategy in securing global health resilience.

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CLIMATE AND INFRASTRUCTURE DRIVE THE SPREAD OF *Anopheles stephensi* IN KENYA

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ABSTRACT

The recent detection of *Anopheles stephensi* in Kenya poses a growing threat to malaria control and raises concerns about its potential role in transmitting other pathogens, including O'nyong'nyong virus (ONNV). Originally endemic to South Asia and the Middle East, *An. stephensi* is an urban-adapted and thermally resilient vector of malaria parasites (*Plasmodium falciparum* and *P. vivax*) and has demonstrated laboratory competence for ONNV. First confirmed in Marsabit County in 2022, the species has since expanded across arid and semi-arid (ASAL) counties in northern Kenya, raising urgent questions regarding its environmental drivers, dispersal mechanisms, and potential distribution under climate change. To investigate these patterns, we integrated national entomological surveillance data (2022–2025) with geospatial environmental and anthropogenic predictors to assess the current and potential distribution of *An. stephensi* in Kenya. Two complementary modeling approaches were applied: Geographically Weighted Regression (GWR) using presence–absence data and Mahalanobis Distance modeling using presence-only data. Predictor variables included anthropogenic, topographic, climatic, and vegetation factors, and incremental warming scenarios (+1 °C to +5 °C) were simulated to evaluate climate sensitivity. GWR predicted the highest spatial risk in northern counties such as Marsabit County, Mandera County, and Isiolo County, with the lowest risk in southern regions. However, Mahalanobis analysis identified ecologically similar conditions in southern counties, particularly Kajiado County and Makueni County, indicating potential for future establishment. Under progressive warming scenarios, habitat suitability declined overall but shifted toward central Kenya. Urban environments may remain suitable due to localized thermal effects and the species' behavioral adaptability. The distribution of *An. stephensi* closely follows major road networks, suggesting that vehicle-mediated dispersal contributes to its spread across northern Kenya. However, cooler transport corridors through Nairobi may currently limit southward expansion. Although its role in ONNV transmission has been demonstrated primarily in laboratory studies, the species' dual competence for malaria and arboviruses highlights the need for integrated, spatially targeted vector surveillance that considers infrastructure, human mobility, and climate adaptation.

Keywords: *Anopheles stephensi*, malaria, Kenya, ecological niche modeling, vector surveillance

INTRODUCTION

The recent incursion of *Anopheles stephensi* into sub-Saharan Africa presents a substantial threat to malaria control and potentially to the emergence of other vector-borne diseases. Originally restricted to South Asia and the Middle East, *An. stephensi* is an urban-adapted

vector capable of transmitting both *Plasmodium falciparum* and *P. vivax* (Sinka *et al.*, 2020; WHO, 2022). Unlike most African *Anopheles* species that prefer rural environments, *An. stephensi* thrives in densely populated settings by exploiting artificial habitats such as rooftop tanks, cisterns, and discarded containers (Carter *et al.*, 2018; Seyfarth *et al.*, 2019). This urban

resilience, coupled with pronounced thermal flexibility, allows *An. stephensi* to persist in environments previously considered unsuitable for malaria vectors. Ecological modeling by Sinka *et al.* (2020) demonstrated that *An. stephensi* possesses a substantially broader thermal niche, with a lower developmental threshold 3.4 °C below and an upper threshold 3.4 °C above that of multispecies malaria models. Its crepuscular activity patterns and apparent tolerance to elevated temperatures suggest behavioral thermoregulation that may buffer the species against climate extremes (Tjaden *et al.*, 2018; Mordecai *et al.*, 2019). These traits make *An. stephensi* particularly well-suited for colonizing rapidly urbanizing regions in Africa, where built infrastructure, water storage practices, and population density can converge to create novel mosquito habitats (Gari *et al.*, 2018; Ahmed *et al.*, 2021).

Since its first detection in Djibouti in 2012, *An. stephensi* has been implicated in malaria outbreaks across multiple countries in the Horn of Africa, including Ethiopia, Somalia, and Sudan (Tadesse *et al.*, 2021; Emiru *et al.*, 2023; Taylor *et al.*, 2024). In Kenya, the species was first confirmed in Marsabit County in late 2022 during routine entomological surveillance (Ochomo *et al.*, 2023). Subsequent surveys indicate that *An. stephensi* has expanded its range across several arid and semi-arid (ASAL) counties in northern Kenya (Samake *et al.*, 2024), raising concern about its potential for long-term establishment and contribution to local malaria transmission. In addition to its role in malaria epidemiology, *An. stephensi* may act as a bridge vector for arboviruses. Laboratory studies have shown it to be a competent vector for O'nyong'nyong virus (ONNV), an alphavirus endemic to sub-Saharan Africa and one of the few known to be transmitted by *Anopheles* mosquitoes (Lederhann *et al.*, 2014; Althouse *et al.*, 2015). Although field transmission by *An. stephensi* has not been documented, ONNV has caused several large outbreaks in East Africa, including Kenya, with periodic resurgences along the Uganda–Kenya border (Williams *et al.*, 1965; Posey *et al.*,

2005). The convergence of malaria and ONNV transmission potential in the same urban settings underscores the emerging complexity of *An. stephensi*'s public health threat.

Despite its growing spatial expansion, the environmental factors driving *An. stephensi*'s spread in Kenya remains poorly defined. While some ecological studies have mapped its potential range across Africa (Sinka *et al.*, 2020; Kraemer *et al.*, 2023), few have examined fine-scale drivers of establishment within Kenya. Moreover, the role of climate change in modulating habitat suitability remains unclear, especially given the species' broad ecological plasticity and ability to decouple breeding from rainfall patterns (Seyfarth *et al.*, 2019; WHO, 2022).

Understanding the local and regional determinants of *An. stephensi* distribution is essential for evidence-based vector control and surveillance. In Kenya, where ecological zones range from arid savannas to highland forests, predictive modeling can help prioritize resource allocation and anticipate areas at risk of future colonization. In this study, we tested the hypothesis that the distribution of *An. stephensi* in Kenya is shaped by a complex interplay of climatic, ecological, and anthropogenic variables. We integrated national surveillance data with Geographic Information Systems (GIS), Remote Sensing (RS), and niche-based spatial modeling to: (i) identify environmental predictors associated with *An. stephensi* presence, (ii) generate spatial risk maps under current environmental conditions, and (iii) project future distribution under incremental climate warming scenarios (+1 °C to +5 °C). To achieve these goals, we applied two complementary species distribution models (SDMs). First, we used Geographically Weighted Regression (GWR) to capture spatial heterogeneity in the relationship between presence–absence data and environmental covariates (Brunsdon *et al.*, 1996; Fotheringham *et al.*, 2002). Second, we implemented Mahalanobis Distance modeling, a presence-only approach suitable for estimating

ecological niche similarity and projecting habitat suitability in unsampled regions (Hirzel *et al.*, 2002; Farber and Kadmon, 2003). This dual modeling framework allows for a robust assessment of both current and potential distribution, informing surveillance priorities, vector control interventions, and early warning systems for malaria transmission in Kenya.

MATERIAL AND METHODS

Study Area

The recent spread of *An. stephensi* across the Horn of Africa has been associated with specific ecological and anthropogenic drivers: tolerance for arid climates, adaptation to artificial containers in urban environments, and dispersal along major transportation routes (Sinka *et al.*, 2020; Samake *et al.*, 2024). These

same conditions are increasingly present in Kenya, making it an ecologically plausible corridor for further establishment and spread. Kenya spans a range of ecological zones, from humid highlands to ASALs, the latter covering ~80% of the national territory (Fig. 1). ASAL counties such as Marsabit, Mandera, Wajir, and Turkana in the north, and Kajiado, Makueni, and Machakos in the south, are characterized by low precipitation, high temperatures, sparse vegetation, and limited piped water infrastructure, conditions similar to areas where *An. stephensi* has become established in Ethiopia, Djibouti, and Sudan. In addition, Kenya has rapidly expanding urban centers and informal settlements where open water storage is common, creating artificial breeding habitats that reduce reliance on rainfall. These urban environments mirror the settings where *An. stephensi* thrives in Asia and the Middle East.

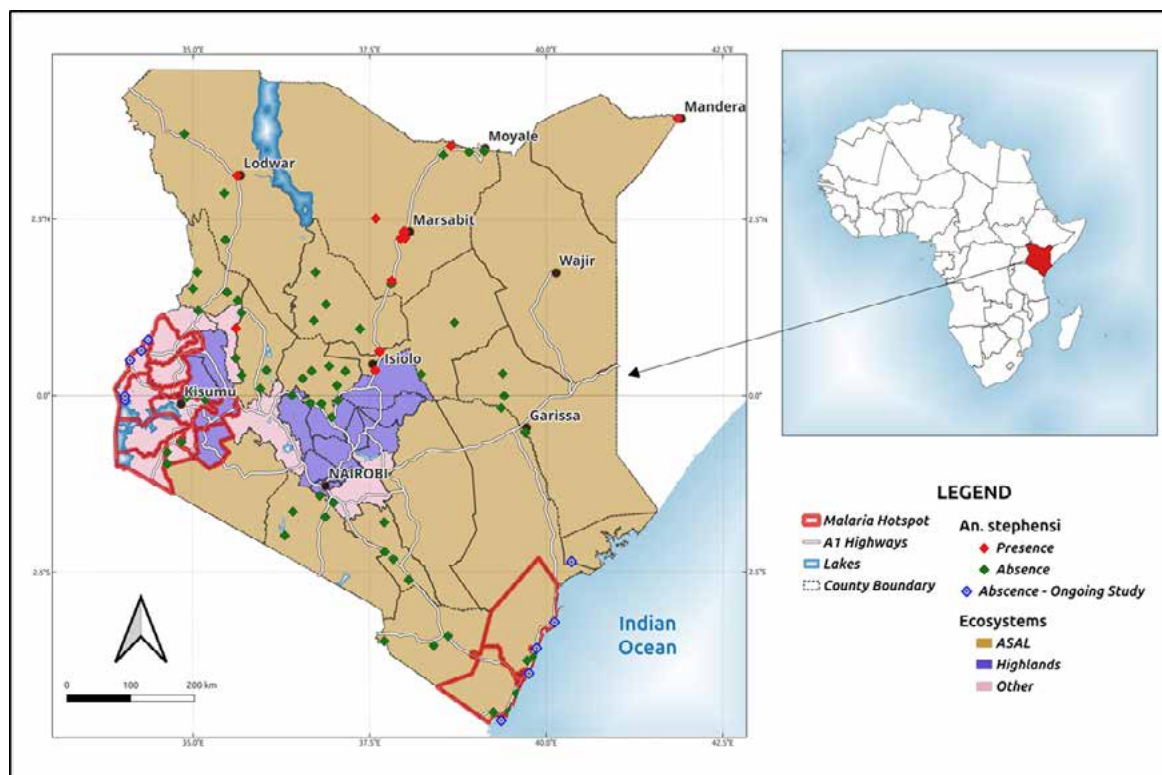


Figure 1: Map of Kenya showing the spatial distribution of the mosquito, *Anopheles stephensi*, during the 2022-2025 national surveillance, malaria zones, ecosystems and road network.

he country is also intersected by major transboundary transportation corridors, including the Mombasa-Nairobi-Addis Ababa Highway, which enters through Moyale and passes through Marsabit and Isiolo, and the Trans-African Highway 4, connecting Egypt to Kenya via Sudan and South Sudan. These corridors provide potential pathways for vehicle- and trade-mediated dispersal, as observed elsewhere in the species' range. This combination of ecological suitability, urban habitat availability, and human-mediated connectivity makes Kenya a high-priority setting for predictive modeling of *An. stephensi* distribution. This study leverages these gradients to assess spatial risk using environmental variables known to influence mosquito ecology.

Data Sources

Anopheles stephensi Distribution Data

We compiled *An. stephensi* distribution data from two sources: published records (2022–2024) and entomological field surveillance conducted between 2024 and 2025. Early detections were concentrated in northern Kenya, including Mandera, Marsabit (Moyale), Isiolo, and Turkana counties (Samake *et al.*, 2024; Ndega *et al.* 2022; Ochomo *et al.*, 2023). A single detection in western Kenya (Mamboleo Slaughterhouse, Kisumu) was reported by Ndega *et al.* (2022), though subsequent surveys failed to confirm continued presence. Recent fieldwork expanded surveillance to under-sampled and ecologically diverse regions, including Bungoma, Busia, Nairobi, Makueni, Tana River, Kilifi, and Kwale (Fig. 1). Site selection was based on environmental risk profiles and logistical accessibility. Adult mosquitoes were collected using CDC light traps and BG-Sentinel traps, and identified morphologically using standard taxonomic keys (Gillies & Coetzee, 1987). Geographic coordinates were recorded for all sampling sites, and each was classified as *An. stephensi*-positive or -negative. Presence–absence data from 28 out of Kenya's 47 counties were

used as inputs for spatial modeling. Absences were defined as sites with zero *An. stephensi* captured after sampling. Identification was verified by expert entomologists.

Biogeographic Variables

A set of biogeographic predictor variables was selected based on their established ecological relevance to mosquito distribution, particularly for *An. stephensi*, and their potential influence on spatial patterns of habitat suitability. These variables were grouped into four categories: anthropogenic, topographic, climatic, and vegetation. We selected Distance to major towns and Distance to major roads as proxies for human settlement density and infrastructure-driven dispersal pathways. These anthropogenic variables have been widely applied in the construction of Human Settlement Pressure Indices and have been shown to influence mosquito colonization in urban environments (e.g., Ferrer *et al.*, 2024; Venter *et al.*, 2016). To generate these variables, data on the location of major towns ($n = 56$) and data on Kenya's A1-class road network were obtained and processed in QGIS 3.34 (QGIS Development Team, 2023) using the Proximity tool, resulting in raster datasets representing Distance from Roads and Distance from Towns.

We selected Elevation and Topographic Wetness Index (TWI) as our topographic variables. Elevation, derived from the 30-meter Shuttle Radar Topography Mission (SRTM) Digital Elevation Model, affects both temperature and rainfall, indirectly shaping mosquito development and survivability. TWI, calculated from SRTM data, represents the spatial distribution of soil moisture accumulation, identifying potential larval habitats. Climatic variables that include mean monthly minimum temperature (°C), mean monthly maximum temperature (°C), and mean monthly precipitation (mm) were estimated from the TerraClimate dataset on Google Earth Engine. Temperature thresholds influence vector competence and the extrinsic incubation period of malaria and arboviruses, while precipitation

affects the formation and persistence of aquatic breeding sites. Normalized Difference Vegetation Index (NDVI) and Vegetation Continuous Fields (VCF) were selected as vegetation variables. NDVI data, averaged across a year (2024), were used as an indicator of vegetation productivity, whereas VCF layers, also derived from MODIS, captured fine-scale vegetation heterogeneity using three indices: Percentage Tree Cover or the fraction (%) of ground area covered by tree canopies, Percentage non-tree cover or the fraction (%) of ground area covered by other vegetation, and Percentage Bare ground. All raster layers were clipped to the Kenyan national boundary and aligned in QGIS 3.34 (QGIS Development Team, 2023) to the same spatial resolution of 1 km² and projected to UTM 36S. The raster layers were then imported into R version 4.3.3 (R Core Team, 2024) using the raster package. Although some variables exhibited moderate inter-correlation (Pearson's $r > 0.6$), all were retained based on their distinct ecological contributions and prior evidence of relevance to *An. stephensi* ecology.

Data Analysis

Statistical Modeling

GWR was used to model spatially varying relationships between *An. stephensi* presence intensity and environmental predictors. Presence-absence data for *An. stephensi* were georeferenced, re-projected to UTM 36S, and overlaid with environmental covariates. To summarize spatial variation across administrative units, environmental predictors were aggregated by county ($n = 47$) using mean, minimum, and maximum values. For counties with multiple sampling locations, the point with the highest number of *An. stephensi*-positive records was selected for modeling. Analyses were performed in R (R Core Team, 2024), using the spgwr package (Bivand, 2024) with the count of *An. stephensi*-positive records per county as the dependent variable. Model performance for GWR was assessed using the global coefficient of determination (R^2) to

evaluate overall model fit, while the corrected Akaike Information Criterion (AICc) was used for bandwidth selection and model comparison. Predictor importance was evaluated using a jackknife approach, where each variable was removed in turn, and the resulting model's performance was compared to the full model using the difference in AIC and R^2 from values calculated for the full set of variables. Variables contributing most to model fit (i.e., those whose exclusion caused the largest increases in AIC or decreases in R^2) were identified as primary drivers of *An. stephensi* presence intensity.

To assess habitat suitability under current and future scenarios, we used Mahalanobis Distance modeling in the raster package (Hijmans, 2025). Presence-only data (i.e., confirmed *An. stephensi* locations) were used to estimate similarity between unsampled regions and known habitats. Predictor variables identified after the GWR jackknife assessment were used for Mahalanobis Distance analysis. To simulate future climate conditions, we increased the minimum temperature in 1 °C increments, from +1 °C to +5 °C, to represent a range of warming scenarios. Risk maps were generated for each scenario to visualize potential expansion under climate change. Because Mahalanobis Distance is an unsupervised presence-only method, traditional classification metrics such as AIC or TSS are not directly applicable. Instead, we evaluated model performance by visual inspection of suitability the spatial distribution of predicted suitability, assessing the habitat suitability maps for ecological plausibility and correspondence with known vector distributions, urban centers, and road networks. For the climate warming scenarios, relative change in suitability (Δ Suitability) was calculated per pixel compared to the baseline (current climate), and spatial trends in expansion were summarized descriptively.

RESULTS

The jackknife analysis of the Geographically Weighted Regression (GWR) model revealed that the Topographic Wetness Index (TWI)

and non-tree vegetation cover were significant positive predictors of *Anopheles stephensi* occurrence. In contrast, bare ground cover and maximum temperature were associated with moderate negative effects. Mean tree cover and mean elevation had minimal explanatory power and, when excluded, led to improved model performance based on reduced Akaike Information Criterion (AIC) values and increased global R^2 . Spatial patterns of predicted risk varied considerably across the country.

High-risk areas were concentrated in northern Kenya, with Marsabit County classified as *Very High Risk*. Isiolo, Mandera, Samburu, Turkana, Baringo, Wajir were classified as *High Risk*. Garissa, Kitui, Laikipia, Tana River, and Meru, Nyandarua were classified as *Moderate Risk*, while Kajiado, Makueni, Narok, and Migori counties were classified as *Moderately Low Risk*. All remaining counties were classified as *Low Risk* for *An. stephensi* colonization (Fig. 2).

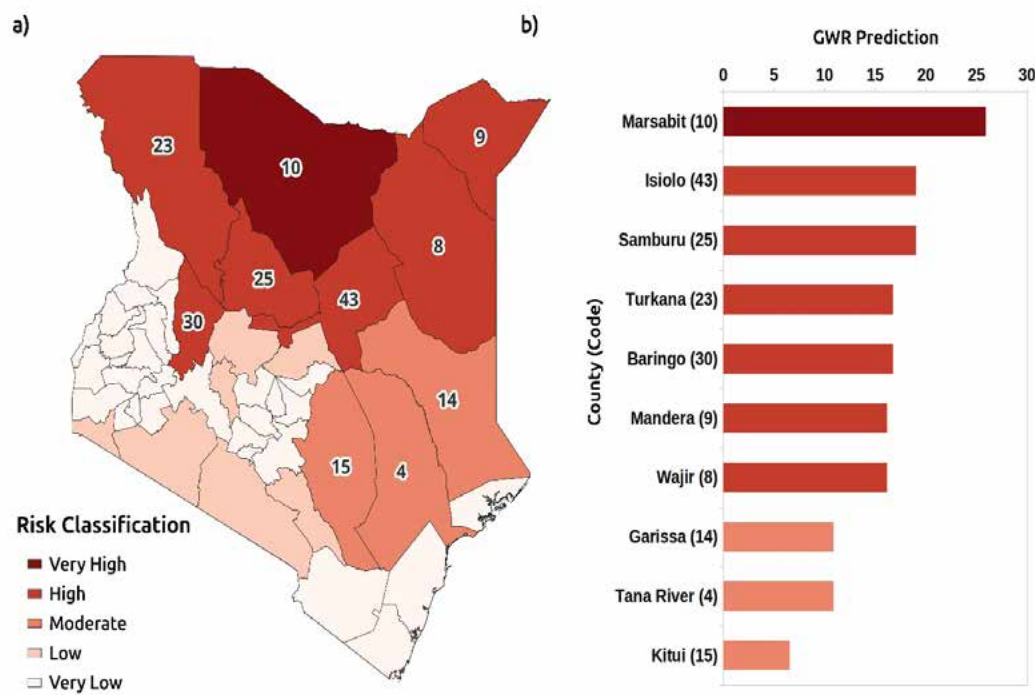


Figure 2. Map of Kenya showing (a) County risk classification based on natural breaks in predicted Geographically Weighted Regression (GWR) analysis. Dark red areas are high risk, whereas light red areas are low risk, and b) predicted sampling numbers for *Anopheles stephensi*.

Habitat Suitability and Future Climate Scenarios

Mahalanobis Distance analysis indicated that, under current climatic conditions, most of Kenya exhibits low to moderate suitability for *An. stephensi* establishment, with Habitat Suitability Index (HSI) scores generally below 0.6. Only a limited number of regions approached optimal suitability ($HSI \geq 0.8$), primarily within ASALs (Fig. 3). In addition to the northern ASAL counties, several counties

in southern Kenya—Kajiado, Machakos, and Makueni—exhibited biogeographic profiles similar to high-risk northern regions, indicating a strong potential for future colonization by *An. stephensi*. Under simulated climate warming scenarios, habitat suitability declined progressively with rising minimum temperatures. As minimum temperatures increased from +1 °C to +5 °C above current baselines, the overall extent of suitable habitat contracted. Habitat loss was most pronounced in the northern and southeastern counties,

which currently represent high suitability zones. Suitability hotspots shifted toward central Kenya, indicating a potential altitudinal or latitudinal range shift in response to increasing temperatures (Fig. 3). These results suggest that while *An. stephensi* currently finds

optimal conditions in limited areas within Kenya's ASALs, climate-driven warming may reduce habitat suitability in these regions while creating new opportunities in previously cooler areas.

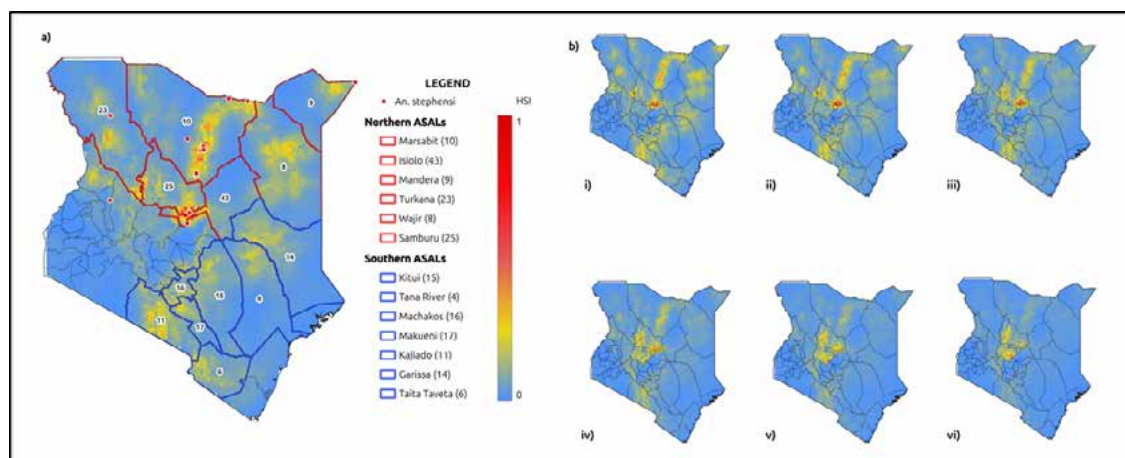


Figure 3. Map of Kenya showing a) Habitat Suitability Index for *Anopheles stephensi* based on Mahalanobis Distance analysis, and b) future climate scenarios based on i) current conditions, and incremental temperature increases of 1 degree Celsius, i-vi (1+ to 5+) degrees Celsius.

DISCUSSION

This study provides new evidence on the ecological and anthropogenic determinants of *An. stephensi* distribution in Kenya. Our geographically weighted regression (GWR) analysis showed a strong association between *An. stephensi* presence and semi-arid landscapes characterized by sparse non-tree vegetation, especially grasslands and bushlands. These findings align with earlier work in Ethiopia and Djibouti, where *An. stephensi* was found to exploit xeric environments traditionally considered unsuitable for malaria vectors (Carter *et al.*, 2018; Tadesse *et al.*, 2021). Notably, precipitation was not a significant predictor in our models, reinforcing the growing consensus that *An. stephensi* is less dependent on rainfall than other *Anopheles* species due to its ability to exploit artificial containers in urban environments (Seyfarth *et al.*, 2019; Sinka *et al.*, 2020). Topographic Wetness Index (TWI) was also positively associated with the mosquito presence, suggesting that semi-permanent water accumulation in depressions,

often near informal settlements, can serve as larval habitats. Previous studies in the Kenyan highlands have demonstrated similar relationships between topographic features and *Anopheles* breeding site density (Cohen *et al.*, 2008). Conversely, maximum temperature and dense tree cover were moderately associated with *An. stephensi* occurrence. While temperature is a known driver of mosquito development and parasite maturation (Mordecai *et al.*, 2019), our results suggest that extreme thermal conditions, both hot and cold, may limit establishment, particularly in regions with less built infrastructure to buffer temperature variability.

Importantly, the highest predicted risk was concentrated in ASAL counties in northern Kenya, including Marsabit, Mandera, Wajir, and Isiolo. This distribution is consistent with reports from other ASAL zones in the Horn of Africa (Tadesse *et al.*, 2021; Samake *et al.*, 2022), indicating a preference for environments where other *Anopheles* species are relatively uncommon. Our results also point to significant

risk in southern ASAL counties, such as Kajiado and Makueni, where *An. stephensi* has not yet been detected, but where ecological profiles closely match northern hotspots. The limited detection of *An. stephensi* in Kenya thus far, only 115 reports in total (Ndega *et al.* 2022; Ochomo *et al.*, 2023; Samake *et al.*, 2024), raises important questions about its current establishment status and sampling sufficiency. Nevertheless, the detection of even a small number of individuals is epidemiologically significant due to the species' vectorial capacity and urban adaptability (Sinka *et al.*, 2020). Its apparent spread along transportation corridors suggests a mode of expansion consistent with human-mediated dispersal. Observations from Ethiopia and Sudan similarly show *An. stephensi* occurrences clustering near major highways and livestock trade routes (Seyfarth *et al.*, 2019; Tadesse *et al.*, 2021), supporting the hypothesis that transport infrastructure facilitates the species' range expansion.

In Kenya, the road network connecting northern ASALs to Nairobi may play a dual role as both a conduit and a barrier to *An. Stephensi* migration. While vehicles likely serve as vectors for passive mosquito transport, the relatively cooler conditions in central Kenya, particularly the highland corridor between Isiolo and Nairobi, may suppress survival and onward transmission to southern Kenya. However, proposed infrastructure developments that bypass Nairobi, such as the Lamu Port–South Sudan–Ethiopia Transport (LAPSSET) Corridor, could potentially enable more direct *An. stephensi* expansion into southern and coastal ASAL zones. As noted by Tadesse *et al.* (2021), expansion into new ecological zones often precedes vector establishment, highlighting the importance of proactive surveillance in seemingly low-risk areas.

Our Mahalanobis Distance modeling revealed that most of Kenya currently provides only suboptimal habitat ($HSI < 0.6$) for *An. stephensi*, but warming scenarios suggest possible niche shifts. As minimum temperatures increase

incrementally, suitable habitats are predicted to contract in the north and shift toward more central counties. This suggests the species' ecological envelope is sensitive to thermal thresholds, although urban heat island effects may buffer these impacts. Previous studies have shown that *An. stephensi* can persist in high-temperature urban microclimates, and even exploit thermal heterogeneity within cities to avoid extreme conditions (Tjaden *et al.*, 2018; Mordecai *et al.*, 2019). Although *An. stephensi* is a proven vector of *Plasmodium falciparum* and *P. vivax* in its native range, its role in arbovirus transmission remains less well understood. Laboratory studies have demonstrated vector competence for O'nyong'nyong virus (ONNV) (Ledermann *et al.*, 2014; Althouse *et al.*, 2015), but field transmission has not been documented. Nonetheless, the combination of high malaria transmission potential and possible ONNV involvement underscores the need for integrated surveillance. Given that ONNV outbreaks have occurred periodically in East Africa (Williams *et al.*, 1965; Posey *et al.*, 2005), and that *An. stephensi* is capable of thriving in densely populated urban centers, future co-circulation of malaria and ONNV in the same locales is a credible threat that warrants further entomological investigation. Together, these findings support a growing body of evidence that *An. stephensi* is not strictly limited by climate and can exploit a range of anthropogenic habitats across diverse ecological zones (Carter *et al.*, 2018; Sinka *et al.*, 2020). Unlike native *Anopheles* vectors that dominate in rural and highland areas, *An. stephensi* appears uniquely positioned to establish persistent transmission cycles in urban settings potentially reversing recent gains in urban malaria control. Our results reinforce calls by the World Health Organization (WHO, 2023) for enhanced cross-border vector surveillance and regionally coordinated responses.

CONCLUSION

This study identifies key ecological and anthropogenic drivers of *An. stephensi* distribution in Kenya, with the highest risk

in northern ASAL counties along major road networks. Our modeling highlights sparse vegetation, topographic wetness, and proximity to infrastructure, not climate alone, as the primary predictors of current distribution. Mahalanobis-based niche modeling under warming scenarios predicts contraction in the north and potential expansion toward central Kenya. Given its competence for both malaria and potentially ONNV transmission, *An. stephensi* represents a growing multi-pathogen threat. We advocate for spatially targeted entomological surveillance and integrated vector control, especially in underserved peri-urban areas and projected expansion zones.

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PREVALENCE OF HAEMOPARASITES IN RODENTS AND SHREWS TRAPPED FROM DOMESTIC AND PERIDOMESTIC HOUSES IN MOROGORO MUNICIPALITY, TANZANIA

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ABSTRACT

Small mammals inhabiting domestic and peridomestic environments are recognized reservoirs of haemoparasites with potential for zoonotic transmission to human populations. Despite this risk, baseline data on haemoparasite prevalence and diversity in small mammals across different habitat types in Morogoro Municipality remain limited. This study aimed to determine the prevalence of haemoparasites in small mammals captured in domestic and peridomestic habitats in Morogoro Municipality, Tanzania, and to identify the haemoparasite types present. A total of 200 small mammals comprising *Mastomys spp.* (n=80), *Rattus spp.* (n=70), and *Crocidura spp.* (n=50) were captured using Sherman live traps in domestic and peridomestic settings. Thin blood smears were prepared from each animal, stained with Giemsa, and examined under 1000x oil-immersion magnification for haemoparasites. An overall haemoparasite prevalence of 32.00% (64/200) was recorded. Four haemoparasite types were identified: *Trypanosoma spp.* (12.50%), *Babesia spp.* (10.00%), *Anaplasma spp.* (7.00%), and *Plasmodium spp.* (2.50%). Prevalence was significantly higher in the peridomestic habitat (35.83%) compared to the domestic habitat (26.25%). Average prevalence across host species was similar: *Mastomys spp.* 31.4%, *Rattus spp.* 30.8%, and *Crocidura spp.* 30.8%. The high haemoparasite prevalence and diversity detected across all three host genera confirm their collective role as competent reservoirs. The elevated infection burden in the peridomestic zone highlights it as a priority area for vector control and public health intervention in Morogoro Municipality.

Keywords: Haemoparasites, Rodents, Shrews, Tanzania, Zoonotic Risk

INTRODUCTION

Small mammals, particularly rodents (Order Rodentia) and shrews (Family Soricidae), play a critical role in the ecology and transmission of numerous zoonotic pathogens. These animals frequently inhabit domestic and peridomestic environments, including houses, refuse sites, gardens, and storage areas. These habitats provide abundant food resources and shelter, enabling sustained interactions between reservoir hosts, arthropod vectors, and human populations (Gratz, 2005; Taylor, 2008). Studies across sub-Saharan Africa have reported diverse haemoparasitic infections in rodents and shrews, including *Trypanosoma*,

Babesia, *Plasmodium*, and *Anaplasma* species (Boulangé *et al.*, 2021). These parasites are commonly transmitted by hematophagous arthropods whose distribution and abundance are influenced by environmental conditions such as vegetation cover, humidity, and host availability (Giraudoux *et al.*, 2012). Consequently, ecological factors governing vector populations play a major role in shaping the spatial distribution and prevalence of haemoparasitic infections. In Tanzania, several studies have demonstrated that small mammals inhabiting human-dominated environments carry a wide range of haemoparasites. Investigations conducted in various regions have identified infections caused by *Anaplasma*,

Babesia, *Trypanosoma*, and *Plasmodium* species in rodent populations (Bernad *et al.*, 2024; Mramba *et al.*, 2024). However, most available studies have focused primarily on the detection of parasites, while comparatively little attention has been given to how infection prevalence varies across different habitat types within urban and peri-urban environments.

Domestic and peridomestic habitats represent distinct ecological settings that may influence host–vector interactions and pathogen transmission. Domestic environments typically include indoor spaces or areas immediately surrounding human dwellings where human activity and sanitation practices may reduce vector abundance. In contrast, peridomestic habitats such as refuse dumps, gardens, and vegetated areas surrounding houses often provide favorable microhabitats for arthropod vectors and support higher densities of small mammal hosts. These transitional zones can function as critical interfaces where pathogen circulation is maintained and from which infections may spill over into human populations.

Despite the recognized importance of these ecological interfaces, limited information is available on the prevalence and distribution of haemoparasites among small mammal communities in many Tanzanian municipalities. In Morogoro Municipality, environmental conditions such as increasing urban expansion, poor waste management, and abundant peridomestic vegetation may facilitate the persistence of both reservoir hosts and arthropod vectors. However, there remains a lack of empirical data on haemoparasite prevalence in small mammals inhabiting domestic and peridomestic environments in this region. Understanding the epidemiology of haemoparasitic infections in small mammal populations is essential for improving disease surveillance and prevention strategies. Identifying high-risk habitats and reservoir host species can provide valuable insights into the ecological drivers of pathogen transmission and help guide targeted control measures.

Therefore, this study aimed to assess the prevalence and diversity of haemoparasites in small mammals captured in Morogoro Municipality, Tanzania, and to examine differences in infection patterns between domestic and peridomestic habitats. By generating baseline epidemiological data on haemoparasitic infections in local small mammal populations, this research contributes to a better understanding of zoonotic disease ecology in human-dominated landscapes and provides information relevant for public health risk assessment and disease control strategies.

MATERIALS AND METHODS

Study Area and Sampling

This study was conducted in Morogoro Municipality, which is located within Morogoro Region in east-central Tanzania (6°49'12"S, 37°39'36"E). The municipality covers an area of approximately 260 km² and lies at an elevation ranging from about 500 m to 600 m above sea level on the floor of the Eastern Rift Valley escarpment, at the base of the Uluguru Mountains. It is situated approximately 196 km west of Dar es Salaam along the A7 national highway and serves as the administrative headquarters of Morogoro Region.

The municipality experiences a sub-humid tropical climate with two rainfall seasons: a long rainy season from March to May and a short rainy season from October to December. Mean annual rainfall ranges between 800 mm and 1,000 mm, and mean annual temperature is approximately 25°C. The surrounding landscape is characterised by a mosaic of urban settlement, agricultural plots, scattered woodland, and riparian vegetation along the Kizinga River and its tributaries, which drain from the Uluguru Mountains through the municipality.

Trapping was conducted across six localities within the municipality, selected to represent a range of urban and peri-urban settlement patterns. Within each locality, sampling was

carried out in two distinct habitat types. Domestic sites comprised indoor rooms, store rooms, and areas immediately adjacent to the walls of human dwellings, where food storage, cooking activities, and regular human occupancy were evident. Peridomestic sites were located in the transitional zone surrounding the dwellings, encompassing refuse dumps, compost areas, vegetable gardens, overgrown vegetation patches, and general yard areas within a radius of approximately 10–30 m from the household structure. These zones were characterised by dense ground cover, accumulated organic waste, and reduced frequency of human disturbance relative to the domestic interior, conditions known to support higher densities of arthropod vectors. A total of six domestic sites (D1–D6) and six peridomestic sites (P1–P6) were established, distributed across the neighbourhoods of Mwanjenga, Kichangani, Kihonda, Chamwino, Mazimbu, Mji Mwema, Bonde la Mpunga, and Mafiga within the municipality.

Host Capture and Identification

A total of 200 small mammals were captured, comprising *Mastomys spp.* (n=80), *Rattus spp.* (n=70), and *Crocidura spp.* (n=50). Host identification was performed morphologically to the genus level using standard external morphological characters (Satterfield *et al.*, 2007).

Parasite Detection

Blood samples were collected from each animal by tail tip incision using a sterile lancet (Satterfield *et al.*, 2007). Thin blood smears were prepared, allowed to air dry, fixed with absolute methanol for two minutes, and stained with 10% Giemsa stain for 20 minutes. Smears were examined under 1000x magnification using oil

immersion for the presence and morphological identification of haemoparasites. Each smear was examined by scanning at least 100 oil-immersion fields before being classified as negative. Haemoparasites were identified to genus level based on established morphological criteria (Singh & Singh, 2015).

Data Analysis

Prevalence (P) was calculated for each group as the percentage of infected animals within the total captured sample: $P(\%) = (\text{Infected} / \text{Captured}) \times 100$. The average prevalence for each species was calculated as the arithmetic mean of their domestic and peridomestic prevalence rates. Differences in prevalence between habitats were assessed descriptively.

RESULTS

Overall Haemoparasite Prevalence

A total of 200 small mammals were captured and examined for haemoparasites. Sixty-four individuals (64/200) were positive for haemoparasitic infections, yielding an overall prevalence of 32.00%.

Haemoparasite Types Identified

Microscopic examination of Giemsa-stained blood smears revealed four haemoparasite genera across the sampled small mammal populations. *Trypanosoma spp.* was the most frequently detected parasite, found in 25 of 200 animals (12.50%), followed by *Babesia spp.* (20/200; 10.00%), *Anaplasma spp.* (14/200; 7.00%), and *Plasmodium spp.* (5/200; 2.50%). A higher count of all parasite types was recorded from peridomestic animals compared to those from domestic settings (Table 1).

Table 1: Haemoparasite types identified in small mammals by habitat, Morogoro Municipality.

Haemoparasite Type	Domestic (n positive)	Peridomestic (n positive)	Total Positive (N=64)	Prevalence (% , N=200)
<i>Trypanosoma spp.</i>	9	16	25	12.50
<i>Babesia spp.</i>	7	13	20	10.00
<i>Anaplasma spp.</i>	4	10	14	7.00
<i>Plasmodium spp.</i>	1	4	5	2.50
Total	21	43	64	32.00

Prevalence by Host Species and Habitat

Prevalence varied between habitats across all species (Table 2). In each genus, peridomestic individuals exhibited higher infection rates than their domestic counterparts.

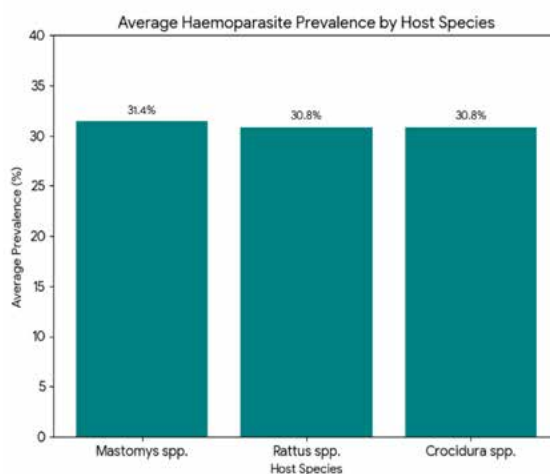
Table 2: Host species composition and haemoparasite prevalence by habitat, Morogoro Municipality.

Host Species	Habitat	Captured (n)	Infected (I)	Prevalence (%)
<i>Mastomys spp.</i>	Domestic	30	8	26.67
	Peridomestic	50	18	36.00
<i>Rattus spp.</i>	Domestic	30	8	26.67
	Peridomestic	40	14	35.00
<i>Crocidura spp.</i>	Domestic	20	5	25.00
	Peridomestic	30	11	36.67
Total	All habitats	200	64	32.00

The peridomestic habitat accounted for a pooled prevalence of 35.83% (43/120), which was higher than the domestic habitat pooled prevalence of 26.25% (21/80). This pattern was consistent across all three host genera, with peridomestic animals consistently recording higher infection rates.

Comparison of Average Prevalence by Host Species

Mastomys spp. exhibited the highest average prevalence at 31.4%. *Rattus spp.* and *Crocidura spp.* shared a nearly identical average prevalence of 30.8%, indicating broadly uniform reservoir competence across the three genera (Fig. 2).

**Figure 2:** Average haemoparasite prevalence by host species.

DISCUSSION

The present study demonstrates a diverse assemblage of haemoparasites circulating within small mammal populations in Morogoro Municipality, Tanzania, with the detection of *Trypanosoma*, *Babesia*, *Anaplasma*, and *Plasmodium* genera. This diversity is consistent with previous reports from other Tanzanian urban and peri-urban ecosystems (Bernad *et al.*, 2024; Deng *et al.*, 2024), reinforcing the role of small mammals as important reservoirs of vector-borne pathogens. Among the detected parasites, *Trypanosoma* spp. exhibited the highest prevalence (12.50%), aligning with findings from coastal regions such as Rufiji, where comparable infection rates have been reported in commensal rodents (Malele *et al.*, 2011). The epidemiological importance of *Trypanosoma* lies in its transmission through biting arthropods and its zoonotic potential, as certain species, including *T. lewisi*, have been implicated in human infections, thereby raising public health concerns (Malele *et al.*, 2011).

Babesia spp., the second most prevalent haemoparasite (10.00%), further underscores the importance of tick-borne transmission within the study area. The detection of this parasite in both domestic and peridomestic environments suggests widespread tick activity, with heightened transmission likely occurring in peridomestic settings. Similar observations have been reported in Arusha, where *Babesia* and *Theileria* were identified as dominant haemoparasites in small mammals (Bernad *et al.*, 2024). Likewise, *Anaplasma* spp. was detected at a prevalence of 7.00%, with higher occurrence in peridomestic habitats, supporting its known ecology as a tick-transmitted pathogen affecting both humans and livestock. Its widespread detection across Tanzanian regions, including Mwanza (Deng *et al.*, 2024), highlights its broad geographical distribution and epidemiological significance. Although *Plasmodium* spp. was detected at a lower prevalence (2.50%), its presence contributes to the overall complexity of haemoparasite communities and indicates the

coexistence of multiple transmission cycles within the same ecological setting.

A key finding of this study is the significantly higher prevalence of haemoparasites in peridomestic habitats (35.83%) compared to domestic environments (26.25%), supporting the concept of the human–wildlife interface as a critical zone for pathogen transmission (Ezenwa *et al.*, 2006). Peridomestic areas provide favorable microhabitats for arthropod vectors due to the accumulation of refuse, vegetation, and stored materials, which enhance vector breeding and survival (Giraudoux *et al.*, 2012). This is consistent with observations in the Morogoro region, where ectoparasite communities are dominated by ticks and mites (Mramba *et al.*, 2024). Furthermore, the peridomestic environment facilitates interactions between commensal species such as *Rattus* spp. and semi-wild species including *Mastomys* spp. and *Crocidura* spp., thereby promoting interspecies transmission and pathogen amplification (Mulungu *et al.*, 2013). In contrast, domestic environments are subject to human management practices that may reduce vector habitats, contributing to the relatively lower infection rates observed indoors (Gratz, 2005).

The relatively uniform haemoparasite prevalence across the three host genera, averaging approximately 31%, indicates that all groups function as competent reservoirs within this ecosystem (Makundi & Mwanjabe, 2005). This finding suggests that zoonotic risk is not restricted to a single host species but is instead distributed across the small mammal community. The ecological behavior of *Mastomys natalensis*, which frequently moves between peridomestic and indoor environments, enhances its role as a bridge host facilitating pathogen transmission between wild and human-associated settings (Gratz, 2005; Taylor, 2008). Additionally, the involvement of *Crocidura* spp. highlights the importance of including insectivores in disease surveillance, as they are often underrepresented despite their role in maintaining haemoparasite transmission cycles (Bernad *et al.*, 2024).

The overall haemoparasite prevalence of 32.00% observed in this study falls within the previously reported national range of 25–40% for small mammals in Tanzania (Makundi & Mwanjabe, 2005). Notably, the peridomestic prevalence closely matches values reported in multiple regions, including Arusha (Bernad *et al.*, 2024), Ilemela District in Mwanza (35.5%) (Deng *et al.*, 2024), Kilombero District (38.76%) (Rubanza *et al.*, 2019), and coastal districts where *Trypanosoma* prevalence reached 36.0% (Boulangé *et al.*, 2021; Malele *et al.*, 2011). This consistency across geographically diverse settings suggests that elevated haemoparasite transmission in peridomestic habitats represents a persistent and widespread ecological pattern rather than a localized anomaly. The comparatively lower prevalence in domestic environments is similarly consistent with previous studies and may reflect reduced vector exposure due to environmental management practices.

From a public health perspective, the findings of this study highlight several important risk factors for zoonotic transmission. The high overall infection rate indicates that small mammals constitute an active reservoir of haemoparasitic infections. The concentration of infections in peridomestic areas—where human activity is frequent—creates direct pathways for exposure through arthropod vectors. Moreover, the involvement of multiple host genera complicates control strategies, as interventions targeting a single species are unlikely to be effective. The observed alignment with patterns from other Tanzanian municipalities further suggests that these risks are structural and persistent, necessitating sustained surveillance and intervention efforts.

Based on these findings, targeted management strategies are essential to reduce transmission risk. Priority should be given to improving peridomestic environmental conditions through waste management, vegetation clearance, and reduction of vector breeding sites (Gratz, 2005; WHO, 2024). Surveillance systems should also be expanded to include detailed identification of haemoparasites using both microscopy

and molecular approaches, enabling more accurate detection of co-infections and cryptic species. Community education initiatives are equally important, as increasing awareness of zoonotic risks can promote behavioral changes that reduce human exposure to vectors and reservoir hosts (CDC, 2018). Additionally, future studies would benefit from incorporating spatial analyses through GIS-based mapping to better understand habitat-specific transmission dynamics.

In conclusion, this study reveals a high burden of haemoparasitic infections among small mammals in Morogoro Municipality, with *Trypanosoma* spp. being the most prevalent. The consistently higher infection rates observed in peridomestic habitats confirm this interface as a key transmission hotspot. The similar prevalence across *Mastomys*, *Rattus*, and *Crocidura* further establishes these taxa as important reservoir hosts of public health significance. Effective control and surveillance strategies must therefore prioritize integrated management of vectors and hosts within peridomestic environments to mitigate ongoing zoonotic risks in Morogoro and similar urban ecosystems across Tanzania.

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SUB-THEME: NATURAL RESOURCES GOVERNANCE AND CONSERVATION

CAN TANZANIA SUSTAIN INCLUSIVE GREEN GROWTH AND SHARED PROSPERITY? A THEORETICAL REVIEW

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ABSTRACT

This paper, employing a documentary review approach, seeks to shed light on how green economy (GE) maximizes value and growth across the entire economy while sustainably managing natural resource assets. Economy growth and wealth generation can be achieved simultaneously with reduction in emissions and environmental impact. Opportunities for green growth-including low-carbon emission and sustainable environmental goods and services can be facilitated while maintaining the global competitiveness of a country's industries. The concept of 'inclusive green growth' should be understood against the background of the prevailing non-green and non-inclusive growth patterns of the world economy. Climate change, ecosystem degradation and resource depletion are threatening the very foundations of life. The concept of inclusive green growth recognises the trade-offs between growth, green and inclusiveness, while emphasising that within the overarching objective of social welfare, there is potential for synergies. Green growth focuses on the welfare of future generations, whereas inclusive green growth addresses the welfare of current generations and the equitable distribution of welfare gains. Shared prosperity, according to World Bank Group's goal, aims to increase per capita real household income or consumption of the bottom 40 percent of each country's population. Since the goal is country specific, there is no explicit target set at the global level. To that effect, it is essential for policymakers to fully integrate all issues related to green growth, inclusive green growth and shared prosperity into national development goals.

Keywords: Green growth, Inclusive growth, Shared prosperity, Tanzania,, Welfare gain

INTRODUCTION

Economic growth can be achieved and wealth generated while simultaneously reducing emissions and environmental impact. Opportunities for green growth can be facilitated, including production of low-carbon, and environmental goods and services, while maintaining the global competitiveness of a country's industry (Bilame, 2023; Green Growth Leaders, 2011; Zhang *et al.*, 2025). A green economy (GE) maximises value and growth across the entire economy

while ensuring the sustainable management of natural resources assets (Zhang *et al.*, 2025). World Bank defines green growth as "a strategy for promoting economic growth while adding the ecological quality to existing economic processes and creating additional jobs and income opportunities with a minimal environmental burden". In addition, according to the OECD, green growth is "the fostering of growth and development while ensuring that natural assets continue to provide the environmental resources and services on which human well-being relies".

The UNEP (2011) and Acosta *et al.* (2020) report defines green economy as “an economy that results in improved human well-being and reduced inequalities over the long term, while not exposing future generations to significant environmental risks and ecological scarcities”.

The concept of ‘inclusive green growth’ should be understood against the prevailing context of the current non-green and non-inclusive growth of the world economy. Despite the significant contribution of global economic growth to increase prosperity and poverty reduction, inequality has risen, and the number of people experiencing hunger has remained largely unchanged (UNEP, 2011; World Bank, 2021). Although debate continues regarding the extent to which the depletion of natural environment can persist without major economic repercussions, it is evident that, at some point, climate change, ecosystem collapse, resource depletion or a combination thereof will impose substantial social and economic costs on humanity, thereby heightening the welfare risks associated with non-green and non-inclusive growth (Damania *et al.*, 2025, Ostryet *et al.*, 2014)

The concept of ‘inclusive green growth’ acknowledges the trade-offs between economic growth, environmental sustainability, and social inclusion, while emphasising that, within the overarching objective of social welfare, there remains scope for synergies. Production growth that is environmentally sustainable and socially inclusive is most conducive to enhancing welfare, as environmental degradation and increasing inequality diminishes the overall well-being (Economist, 2014). From a welfare economics perspective, ‘inclusive green growth’ can be understood as growth that improves the welfare of both current and future generations, recognising the social costs and benefits (including environmental costs) of growth and its distributional implications over both the short and long term (Bilame, 2023, Jetske and Ezra, 2015). In this context, the core meaning of the concept of ‘inclusive green growth’ can be simply defined as an

economic growth (measured by gross domestic product or GDP) that simultaneously achieves substantial environmental protection and encompasses all major sectors of the economy, thereby employ a large spectrum of the active working population.

Adopting a welfare-economics definition also facilitates the formulation of inclusive green growth policies. Questions regarding how sustainable growth could be stimulated and how the benefits of such growth should be allocated between current and future generations are extensively discussed in the economic literature, providing numerous insights relevant to the design of inclusive green growth policies (Bilame, 2023, Jetske and Ezra, 2015). For example, welfare theory underscores the role of well-functioning markets in maximising social welfare, yet, highlighting their limitations for public good resources (including environmental goods and services), where as property rights are hard to define and cannot be made exclusive, a prerequisite for efficient market operation.

To this end, the main objective of the study is to show how Tanzania can sustain inclusive green economic growth that promotes shared prosperity, reflected in the welfare of current and future generations. This welfare should also be manifested in both equity and quality. Equity denotes equality of opportunities, where all people are equal before the law, while equality refers to outcomes-that is, everybody should earn the same. In the current context of neoliberal market ideology, the emphasis is predominantly on equity, with equality traditionally considered a political objective. To achieve the intended objective, the study, employed a methodology based on a review of relevant studies and publications on the subject.

Growth and Green Growth

The main difference between ‘growth’ and ‘green growth’ is that the latter acknowledges the role of natural capital in growth and its important role in ensuring the welfare of future

generations. As noted earlier, capital stocks are essential for growth and development; to achieve sustainable development, current generations should ensure that capital stocks are at least maintained (Fitter, 2013; Omukuti *et al.*, 2025; United Nations, 2025.).

Natural capital constitutes a key component of a country's overall capital stock; consequently, degradation of ecosystems, deforestation and resource depletion reduces the welfare of future generations if the resulting resource rents are not reinvested in alternative forms of capital. When resource rents are reinvested in alternative forms of capital (e.g. human capital or other assets) future generations may inherit a comparable level of overall capital, thereby ensuring sustainable development. This approach is referred to as weak sustainability. By contrast, strong sustainability requires that future generations inherit not only a comparable quantity of capital, but also a similar composition of capital (Neumayer 2003; United Nations, 2025). This particularly applies to natural capital that is only partially substitutable and, therefore irreplaceable. Fitter (2013) assesses the substitutability of natural capital, concluding that most services provided by natural capital currently are non-substitutable, and where substitution is possible, it comes at a very high cost. Although this may change in the future (for example, a breakthrough innovation could replace soil formation processes), a precautionary approach is warranted to ensure that future generation inherit at least a minimum level of natural capital (Devi *et al.*, 2025, Fitter, 2013).

It is important to recognise that many of the poorest countries do not reinvest their resource rents, and that the over-exploitation of natural capital is often occurs through foreign direct investments (such as mining, plantations, fossil fuel extraction, agro-industry), facilitated by bilateral and international trade policies (Bilame, 2020; Duchaevea & Magomadov, 2023). Furthermore, a precautionary approach entails economic opportunity costs, which are particularly high for developing countries,

given their relatively large stocks of natural capital compared to other forms of capital. These opportunity costs must be taken into account when discussing the protection of global environmental resources, such as biodiversity and carbon stocks (Bilame 2020; Dercon, 2012).

'Green growth' typically involve investments in innovation and improvements of resource-use efficiency, aimed at reducing the rate of natural capital depletion and increasing the likelihood of identifying viable substitutes. Given that the scarcity value of environmental resources is not or not fully expressed in market prices, the incentive to invest in eco-innovation is limited. Win-win strategies may fail to materialise because trade-offs often emerge between short-term economic interests and long-term environmental benefits; for example, the immediate profits from exploiting cheap fossil fuel may outweigh the long-term environmental benefits of low carbon energy). Addressing these trade-offs is a central challenge for 'green growth' strategies, for instance, by creating appropriate incentives and reducing institutional barriers to sustainable resource use (Jetske and Ezra, 2015).

Green Growth and Re-thinking from the Current Approach

Critical natural resources-clean air, abundant and safe water, and healthy and nutrient rich soils- are fundamental to increasing living standards and reducing poverty (Bilame, 2020). Clean air is essential for human health and productivity; water is a critical input for agriculture, energy generation, and industrial processes; and fertile soils are the foundation for producing nutritious food and supporting ecosystem services.

According to the World Bank (2018), approximately 80% of people in low-income countries are exposed to all three major environmental hazards: land degradation, unsafe air pollution levels, and water stress.

By contrast, only 1 percent of people in high-income countries are exposed to all three environmental risks (Damania *et al.*, 2025; World Bank, 2024). However, it would be misleading to suggest that affluence automatically provides immunity from environmental risks. Environmental degradation remains a global issue, and approximately 90 percent of the world's population lives in areas where at least one of the three critical natural resources—land, air, or water—is degraded. The frequent co-occurrence of these environmental challenges, particularly in low-income countries, illustrates that environmental problems carry economic consequences that act as a headwind on development (Damania *et al.*, 2025; Omukuti, *et al.*, 2025). Addressing a single stressor in isolation may yield limited benefits if other overlapping risks persist.

These statistics challenge the traditional paradigm that environmental degradation is an inevitable consequence of industrialisation, and diminishes in post-industrial societies. In reality, many of the countries most affected by environmental degradation have yet to industrialise. They bear the dual burden of poverty compounded by a degraded environment, without having reaped the benefits of industrialisation. Consequently, achieving green growth with shared prosperity is unlikely under such circumstances (Devi *et al.*, 2025; World Bank, 2025.)

This demonstrates that in many low-income countries, particularly in Sub-Saharan Africa, large shares of the population are both poor and exposed to stressors. Inequality also runs deep within countries. Poor individuals and communities excluded from decision-making face greater environmental challenges: they are 75 percent more likely to live in areas where land is degraded and to experience deficits in essential services such as piped water, sanitation, and electricity (Damania *et al.*, 2025). Without access to these essential services, they remain at a higher risk of suffering from the negative effects of poor environmental conditions. Accordingly, achieving green economic growth remains a challenge for citizens.

Green Economy Issues in the Context of Sub-Saharan Africa

Sub-Saharan African countries (SSA) are at a critical juncture in their development trajectories. Confronted with persistent poverty and unemployment, the resource base of many productive sectors is increasingly threatened by environmental degradation, including deforestation, soil erosion, desertification, loss of biodiversity, depletion of fish stocks and the impacts of climate change (World Bank Annual Report, 2025). At the same time, these challenges represent opportunities for change.

A green economy offers considerable opportunities to mobilise resources in support of a low-emission, climate-resilient development pathway (Bilame, 2020; European Union, 2017; UNEP, 2011). However, the combination of pursuit of poverty alleviation, accelerated growth and development, and climate change mitigation and adaptation is likely to involve trade-offs and policy choices between mutually supportive, but equally important priorities toward the improvement of welfare and quality of life for Africa's citizens. This is likely to entail increased consumption of food, energy and water. In this context, policies and investments aimed at sustaining and enhancing natural capital assets—such as soils, forests and fisheries upon which many poor communities depend for their livelihoods—can be instrumental (Bilame, 2023). Concurrently, the economic system must evolve to enable sub-Saharan African (SSA) to improve its terms of trade and expand its productive capacity. The transition to a green economy will require the mobilisation of new financial resources from both public and private sources. This necessitates a realignment of policy goals and priorities, alongside the corresponding allocation of capital, while simultaneously addressing immediate and pressing development challenges and short-term poverty alleviation objectives.



Figure 1: Global and Sub-Saharan Africa Forest Degradation
Source: World Bank (2022)

Deforestation rates are exceptionally high across SSA, with approximately 4 million hectares of forest (which is also another important source of livelihoods for a significant proportion of the population) lost annually (GIZ, 2013). This is illustrated in Fig 1, which highlights the extent of deforestation in SSA. The evidence underscores the urgent need for remedial measures. Given the accelerating rate of deforestation, there is a high call for all countries in SSA to adopt sustainable and inclusive measures. Green Growth cannot be achieved without first addressing deforestation concerns in SSA. Measures aimed at addressing deforestation must, therefore occupy a central position in strategies designed to mitigate forest loss. At this juncture, it is important to note that ‘Green Growth’ and ‘Green Economy’ are two sides of the same coin, but integral to the pursuit of sustainable afforestation. Furthermore, it should be emphasized that the sustainable inclusive conservation and enhancement of natural resources constitutes a vital source of income, livelihoods and employment for the majority of Africans, and represent a starting point for a transition towards ‘Green Growth’

and ‘Green Economy’. As shown in Fig 1, the continent of South America is experiencing a deforestation trend similar to that of that of SSA, underscoring the urgent need for remedial measures.

Shared Prosperity

The goal of shared prosperity recognizes that promoting the well-being of the most vulnerable society is a key development objective (Narayan *et al.*, 2013; Ofori, 2022). Thus, although the average income growth among the bottom 40 percent of the population has become the standard indicator of shared prosperity, the goal itself is much broader, aiming to sustainably enhance the well-being of the poorer segments of society. Embedded in this goal, therefore, are both inter-temporal and multidimensional objectives that defines shared prosperity as the well-being that must be equitably distributed amongst individuals over time. This multidimensional aspect of the goal points to the need to focus on non-income dimensions of prosperity such as education, health, nutrition, and access to

water and essential infrastructure, as well as on strengthening the voice and participation of all segments of society in economic, social and, political spheres (World Bank, 2013b). The broad focus of the shared prosperity goal aligns with the call for development objectives that extend beyond mere access to or ownership of material goods. Amartya Sen (1983, 1999)- a key proponent-has called for income to be viewed not as the sole end to development but rather as a gauge of what a person is able to do (capability) and manages to do (functioning). This broader perspective of development, further reinforced by Bilame (2020), has been influential in the literature on broad-based growth, leading initially to efforts to measure the non-income dimensions of development and subsequently on inclusive growth, which examines how growth trickles down to the poor. That work, in turn, has led, through the introduction of multidimensionality, to the notion of “inclusive development.”

The second goal of the World Bank Group articulates a commitment to promoting “shared prosperity,” and defines it as seeking to sustainably enhance the well-being of the poorer segments of society. The goal reflects a practical compromise between the single-minded pursuit of prosperity and an equity-oriented concern about the ability of the less well-off in society to improve their well-being through participation in a country’s prosperity (Bilame, 2023; Narayan *et al.*, 2013). The goal, therefore places grater emphasis on inclusive development and growth than has been traditionally been the case, while also paving the way for a focus on inequality (Damania *et al.*, 2025; Lakner *et al.*, 2014). The goal is measured by the pace of real income or consumption growth at the household level, on average and over time, for the bottom 40 percent of the income distribution in each country.

At this stage, one is in a position to comment as to whether the kind of Tanzania economic growth trend over the current decade aligns with the principle of “inclusive green growth”

and the pursuit of ‘shared prosperity’. From the earlier discussion in this chapter, ‘inclusive green growth’ has been pointed out to be nothing more than growth that improves the welfare of both current and future generations; that acknowledges the social costs and benefits (including environmental costs) of growth and its distributional implications in both the short and the long term (Bilame, 2023; Jetske and Ezra, 2015; World Bank, 2025). Furthermore, the core concept of ‘inclusive green growth’ can be simply defined as an economic growth (GDP growth) that simultaneously achieves a significant environmental protection and takes on board all major sectors of the economy that employ a large spectrum of the active working population. On the other hand, shared prosperity, as highlighted in this paper, recognises the pursuit of enhancing the well-being of the most vulnerable in a society (Narayan *et al.*, 2013; World Bank, 2024).

Green Economy in the Context of Tanzania

To date, Tanzania has no national definition of the term “green economy” in the context of sustainable development and poverty eradication. However, different national policies have elements that promote a green economy. Such elements include promotion of renewable energy sources (solar, hydropower, wind, geothermal, biogas), the use of energy efficient appliances and equipment, efficient mass transit systems, cleaner production initiatives, fuel switching to natural gas and other alternative energy sources, the use of energy efficient cooking stoves, and promotion of mini-hydro and co-generation projects (URT, 2010c). It is broadly considered by various sectors in the country as an economy that improves human well-being while managing the environment sustainably; one that ensures sustainable development and prevent environmental degradation, reduces greenhouse gas emission, integrates issues of social, environment and economic aspects into development process, and takes into consideration the sustainability of future generations.

Tanzania embraces the view that a defining feature of a green economy is that seeks to provide diverse opportunities for economic development and poverty alleviation without depleting country's natural assets. This is particularly necessary in low-income countries, where ecosystem goods and services constitute substantial component of the livelihoods of poor rural communities and serve as a vital safety net against natural disasters and economic shocks. The green economy should be regarded as a tool to facilitate the achievement of sustainable development and poverty eradication and not to replace them.

Where Does Tanzania Stand in terms of Green Growth and Shared Prosperity?

The extent to which Tanzania reflects principles of inclusive green growth and shared prosperity in her development endeavors remains somewhat uncertain. The recently developed National Development Vision 2050 remains silent on issues of green growth and shared prosperity. One would have expected this document to provide clear insights into the key issues regarding green growth and shared prosperity in the context of Tanzania. The Vision 2050 highlights broader issues of global warming and climate change, emphasising the need for Tanzania to prepared accordingly. Its silence on issues related to inclusive green growth and shared prosperity in development strategies may indicate that the country has not prioritised these objectives, raising questions about the direction of national development and the strategies been adopted to achieve inclusive green growth and shared prosperity.

CONCLUSION

This study has highlighted how green growth strategies often involve investments in innovation and resource use efficiency to reduce the rate of natural capital depletion and increase the likelihood of identifying substitutes. Given that the scarcity value of environmental resources is not or not fully expressed in market prices, the incentive to invest in eco-

innovation remains limited. Win-win strategies may not materialise because trade-offs often emerge between short-term economic interests and long-term environmental benefits. Overcoming these trade-offs is the challenge of green growth strategies.

The broad focus of the shared prosperity goal aligns with the call for development goals that extend beyond access to or ownership of material goods. This broader perspective of development has been influential in the literature on broad-based growth, and has led first, to efforts to measure the non-income dimensions of development and then to work on inclusive growth that examines how growth trickles down to the poor. That work, in turn, has led, through the introduction of multidimensionality, to the notion of inclusive development.

As to whether Tanzania mirrors in green growth and shared prosperity in her development endeavors remains somewhat a challenge. The recently developed National Development Vision 2050 remains silent on issues of green growth and shared prosperity. The Vision 2050 highlights general issues of global warming and climate change and that Tanzania should get prepared for the same. At this juncture, it is not too late for the Vision 2050 to integrate the issues of green growth and shared prosperity into its development framework. In the course of its review, issues of green growth and shared prosperity should be prioritised, with a view to achieving sustainable development-development that considered not only the benefits of the current generations but also the well-being of future generations.

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COMMUNITY-DRIVEN REGENERATION OF MIOMBO WOODLANDS AND CHIMPANZEE CORRIDORS IN WESTERN TANZANIA

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ABSTRACT

Miombo woodlands are amongst the most extensive and ecologically significant ecosystems in Tanzania, yet they are increasingly threatened by deforestation, recurrent wildfires, agricultural expansion, and unsustainable extraction of forest resources. In western Tanzania, degradation of miombo woodlands has contributed to habitat fragmentation and the weakening of ecological connectivity for chimpanzees (*Pan troglodytes schweinfurthii*) within the Greater Gombe Ecosystem. This paper documents a five-year, community-driven restoration initiative implemented in the villages of Kagongo, Mtanga, Chankabwimba, and Kiziba, where local communities, led by youth through the Jane Goodall Institute's TACARE and Roots & Shoots (R&S) models, have voluntarily allocated over 56 hectares of land for assisted natural regeneration (ANR). Using participatory restoration approaches, community fire management, and youth-led citizen science monitoring, the initiative has resulted in measurable vegetation recovery, increased biodiversity, and re-emergence of ecological functions. An observation from Buyombo Hill (Kagongo Village) recorded 27 native tree species and a Shannon–Wiener Diversity Index (H') of 2.57, indicating high species diversity and evenness in a recovering miombo ecosystem. Beyond ecological gains, the initiative has strengthened social cohesion, enhanced environmental literacy, and contributed to improved ecosystem services. The findings demonstrate that community ownership and youth leadership can provide a scalable, cost-effective model for the restoration of degraded miombo woodlands and strengthening chimpanzee corridors in Tanzania.

Keywords: assisted natural regeneration, community-based conservation, Miombo woodlands, wildlife corridor

INTRODUCTION

Miombo woodlands dominate much of southern and western Tanzania and support high levels of biodiversity, ecosystem services, and rural livelihoods (Frost, 1996; Chidumayo & Gumbo, 2010). These woodlands are characterized by tree species in the genera *Brachystegia*, *Julbernardia*, and *Isoberlinia*, and they play a critical role in carbon storage, soil fertility, hydrological regulation, and wildlife habitat. However, widespread degradation driven by charcoal production, shifting cultivation, overgrazing, and recurrent wildfires has resulted in forest fragmentation and declining ecological integrity (Bond & Parr, 2010).

In western Tanzania, degradation of miombo woodland poses a significant threat to chimpanzee populations that rely on forested landscapes and functional corridors for movement between habitat patches. The Greater Gombe Ecosystem historically supported extensive chimpanzee ranges, but increasing human pressures have reduced forest cover and habitat connectivity (Craigie *et al.*, 2010). The restoration of degraded miombo landscapes is therefore essential, not only for biodiversity conservation but also for maintaining ecosystem services and supporting local livelihoods.

Conventional restoration approaches in miombo ecosystems have often relied on

tree planting, which can be costly and ecologically mismatched to local conditions (Chidumayo & Gumbo, 2010). In contrast, assisted natural regeneration (ANR), which focuses on protecting natural regrowth, suppressing disturbances, and enabling ecological succession, has emerged as a more effective and context-appropriate strategy for miombo ecosystems (Chazdon & Guariguata, 2016). This paper presents evidence from a community-led ANR initiative facilitated by the Jane Goodall Institute Tanzania (JGITZ), highlighting the role of youth engagement, community ownership, and participatory monitoring in restoring degraded miombo woodlands and strengthening chimpanzee corridors.

MATERIAL AND METHODS

Study area

The study was conducted in western Tanzania, within the Greater Gombe Ecosystem, covering villages in Kigoma Region (Kagongo and Mtanga) and neighboring landscapes in Chankabwimba and Kiziba villages, where a community-led restoration initiative was implemented. These areas lie within the miombo woodland zone and experience a unimodal rainfall pattern with a pronounced dry season, during which wildfires are common (Frost, 1996). Buyombo Hill in Kagongo Village, one of the selected study sites, serves as a focal case study. Historically, the hill was severely degraded due to repeated burning, agricultural clearing, and charcoal production, pressures widely documented in miombo landscapes across southern and eastern Africa (Bond & Parr, 2010). Prior to intervention, vegetation cover was sparse, erosion was evident, and wildlife presence was minimal. Despite this degradation, remnant tree stumps and root systems remained, providing a foundation for natural regeneration.

Methods

Community Engagement and Governance

The restoration initiative was guided by the JGI TACARE (Take Care) model, which emphasises community empowerment, participatory decision-making, and integration of conservation with livelihoods. Community-based natural resource management approaches have been widely recognised as effective in improving conservation outcomes in rural African landscapes (Berkes, 2004). Community sensitisation meetings were conducted to raise awareness on forest conservation, wildlife protection, and fire management. Villagers voluntarily allocated land for regeneration, and R&S alumni volunteered to support the regeneration initiative at the hill site.

Youth Leadership and Roots & Shoots Involvement

Youth engagement was central to the initiative. Former Roots & Shoots students, particularly those who had completed secondary education, expressed interest in continuing conservation work beyond school. Youth-led environmental action has been shown to enhance long-term stewardship and intergenerational transfer of conservation values (Hartley *et al.*, 2015). These youth served as restoration champions, mobilising peers, conducting outreach at household level, and overseeing regeneration sites.

Assisted Natural Regeneration (ANR)

Instead of relying on traditional approach, which involve planting trees, the initiative adopted ANR as its primary restoration strategy. ANR is particularly well suited to miombo woodlands, where many tree species regenerate vigorously from existing rootstocks once fire and pressure from cutting are reduced (Chidumayo & Gumbo, 2010). Key activities of the ANR included the protection of existing tree stumps and natural seedlings, establishment of firebreaks around regeneration sites, installation of boundary markers, and community education on sustainable grazing, fire use, and tree cutting. This approach minimised costs while preserving local genetic diversity of miombo

tree species and maintaining overall ecosystem structure.

Biodiversity Monitoring and Citizen Science

Youth and students conducted participatory biodiversity monitoring using simple, replicable methods. At Buyombo Hill, a 6.25-hectare area was surveyed and data collected through species identification walks, plot observations, and ecological scavenger hunts. Data collected included tree species presence, signs of wildlife activity, and qualitative indicators of ecosystem health.

Species diversity was quantified using the Shannon–Wiener Diversity Index (H'), and evenness (E) was calculated to assess the distribution of individuals amongst species. Although data collection was community-led, results were reviewed with technical staff from Jane Goodall Institute Tanzania (JGITZ) to ensure consistency and scientific rigor.

RESULTS

Vegetation Recovery

At Buyombo Hill, 27 native tree species were identified within the monitored area. Dominant species included *Brachystegia spiciformis*, *Anissophylllea boehmii*, *Albizia antunesiana*, *Pterocarpus angolensis*, and *Uapaca kirkiana*. Many of these species are important food sources for wildlife, including chimpanzees, birds, and small mammals. In addition, vigorous sapling recruitment, increasing canopy cover, and reduced signs of erosion were observed, suggesting that the woodland is progressing along a natural recovery trajectory.

Species Diversity

The Shannon–Wiener Diversity Index calculated for Buyombo Hill in 2023 was $H' = 2.57$, with an evenness value E of 0.79. These values indicate relatively high species diversity and a balanced community structure for regenerating miombo woodland, comparable

to other recovering forest systems in the region.

Faunal Indicators and Ecosystem Functions

Evidence of increasing signs of faunal return, was documented, including butterflies, small mammals, birds, and edible mushrooms. Bird calls, nests, feathers, droppings, and animal footprints observed, suggested re-establishment of trophic interactions and pollination networks. Fifteen key ecological indicators were consistently recorded, including termite mounds (soil nutrient cycling), spider webs (predator–prey balance), blooming flowers (active regeneration), and wildlife droppings (foraging activity). Collectively, these indicators suggest a recovering and functioning ecosystem.

SOCIAL AND COMMUNITY IMPACTS

Fire Reduction and Improved Management

The creation of firelines and increased community awareness resulted to a notable decline in wildfire incidents within the regeneration sites. Community members reported improved control of dry-season fires and increased adherence to local fire management bylaws.

Livelihood and Ecosystem Service Benefits

As regeneration progressed, communities began to experience tangible benefits, such as improved soil fertility, better water retention, reduced erosion, and the return of non-timber forest products such as mushrooms and medicinal plants. Cooler microclimates were also noted, particularly during the dry season.

Education and Social Cohesion

The regenerated sites function as outdoor classrooms for Roots & Shoots clubs and local schools. Students participate in species identification, ecological monitoring, and group discussions, enhancing environmental literacy and conservation ethics. Collaboration

amongst youth, women's groups, elders, and village leaders strengthened social cohesion and collective responsibility for natural resource management.

DISCUSSION

The findings demonstrate that community-driven ANR can effectively restore degraded miombo woodlands while supporting biodiversity conservation and ecosystem connectivity for chimpanzees. Compared with tree planting, ANR proved to be more cost-effective, ecologically appropriate, and socially acceptable in the study area.

Youth leadership emerged as a critical driver of success. Through Roots & Shoots, youth played a pivotal role in driving behavioural change, knowledge dissemination, and long-term stewardship within the community. This aligns with broader evidence that inclusive, and participatory approaches enhance restoration outcomes and resilience.

The initiative also contributes to national and global commitments, including Tanzania's Bonn Challenge pledge to restore 5.2 million hectares of degraded land and the UN Decade on Ecosystem Restoration (2021–2030).

CHALLENGES AND LESSONS LEARNED

Despite positive outcomes, several challenges were identified. Limited financial and logistical resources constrained enforcement of bylaws and expansion of monitoring activities. Occasional political interference and continued dependence on forest products amongst vulnerable households posed risks to long-term sustainability. These challenges underscore the need for continued capacity building, livelihood diversification, and institutional support.

A key lesson is that restoration success is closely linked to community ownership. Where communities perceived direct benefits and maintained control over decision-making,

levels of compliance and stewardship were markedly higher.

Challenges remain:

- Limited resources for enforcing bylaws
- Occasional political interference
- Dependence on forest products amongst vulnerable households
- Need for more technical restoration capacity building

Yet, the enthusiasm of R&S youth, combined with community acceptance, demonstrates that lasting restoration is possible when local people are empowered.

CONCLUSION AND RECOMMENDATIONS

Community-driven restoration of miombo woodlands in western Tanzania has demonstrated measurable ecological recovery, strengthened chimpanzee corridors, and delivered multiple social benefits. Assisted natural regeneration, combined with youth engagement and participatory monitoring, offers a scalable model for landscape restoration. To sustain and expand these gains, we recommend: Strengthening VNRC capacity through training and consistent technical support; Enhancing community education on fire management and forest laws; Providing basic field equipment for biodiversity monitoring; Supporting youth in advanced citizen science techniques; Expanding ANR approaches to neighboring villages; and Documenting biodiversity recovery through regular, standardized assessments and disseminate the findings or achievements to the communities for inspiration and adoption purposes.

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SUPPORTING PHOTOS



Photo 1-Regeneration Progress (2020 → 2024): This is a regeneration progress for the Buyombo Hill in Kagongo village-Kigoma region



2021



2024



Photo 2-Mushrooms at Buyombo Hill: *Edible mushrooms collected, indicating revived soil biology and microhabitats.*



Photo 3-Student Conservation Excursion: *School groups learning ecological monitoring during a field-based class.*



Photo 4-Group Photo at Buyombo Hill: *Students, youth, and mentors after completing a forest hike at Buyombo regeneration hill.*



Photo 5-Fireline creation: *Community members involved in fire management at the regeneration site-Buyombo Hill, this is to protect the hill from fire resulting from the slash and burn practice usually practiced in dry seasons.*



Photo 6- Buyombo hill site overseers: *Few youths volunteering to oversee the regeneration site-Buyombo Hill crew-Kagongo village in Kigoma region.*

SUB-THEME: SUSTAINABLE TOURISM

CATEGORISING DOMESTIC TOURISTS IN TANZANIA'S NATIONAL PARKS: TYPES OF TOURISTS' THEORY APPROACH

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ABSTRACT

Innovation in tourism can also include building on theories that guide understanding of a phenomenon. In this paper, understanding domestic tourists in the context of Tanzania is crucial in order to improve domestic tourism as well as sustain the industry and thus have sustainable tourism. Application of the types of tourists' theory in tourism was used in past studies to determine the tourists' categories. Therefore, in order to expand literature and build on the types of tourists' theory, this paper's main objective is to examine the categorisation of domestic tourists in Tanzania's national parks using the types of tourists' theory. This study's location is Nyerere National Park in Tanzania. Data collected from 360 domestic tourists was analysed using descriptive statistics. The findings show that the majority of the domestic tourists were characterised as first-time visitors to Nyerere National Park (61.4%) and visited mostly through package tour (58.9%). Guided by Cohen's category of tourists, this implies that these domestic tourists are seeking familiarity of the park since they are unfamiliar with the park and its activities due to them being first time visitors. In addition, many domestic tourists also visit the park on package tours implying that they are visiting on pre-determined itinerary hence, these domestic tourists can be categorised as organised mass tourists who seek familiarity rather than novelty. Future research can explore longitudinal design to understand the patterns of domestic tourists in visiting national parks.

Keywords: Domestic tourists, national parks, types of tourists' theory, Tanzania

INTRODUCTION

National Parks (NPs) as protected areas are important for economies of many countries around the world. Africa is endowed with national parks ranging from the Serengeti National Park in Tanzania, Amboseli National Park in Kenya, Kruger National Park in South Africa to Murchison Falls National Park in Uganda (Tanzania National Parks, 2025a; Kenya Wildlife Service, 2025; Kruger National Park, 2025; Uganda Wildlife Authority, 2025). The sustainability of these endowments requires extensive research to understand the tourists that visit these national parks for various purposes like pleasure, enjoyment and even relaxing. For instance, studies such as Kara and Mkwizu (2020) noted that among

the factors that influence travel motivations for both local and international tourists visiting Tanzania are age and family size. Other scholars like Busagara *et al.* (2023) explored further on tourists in Tanzania and found that interactions of tourists exhibited strong positive mediating effect on service innovation.

However, this paper is of the opinion that innovation in tourism can also include building on theories that guide understanding of a phenomenon. Furthermore, understanding domestic tourists in the context of Tanzania is crucial in order to improve domestic tourism as well as sustain the industry and thus have sustainable tourism. Application of the types of tourists' theory in tourism was used in past studies to determine the tourists' categories.

Therefore, in order to expand literature and build on the types of tourists' theory, this paper's main objective is to examine the categorisation of domestic tourists in Tanzania's national parks using the types of tourists' theory.

Literature review

Domestic tourists

According to McCabe and Stokoe (2004), a tourist is generally identified in terms of consuming specific types of places. On the other hand, this study refers to domestic tourists as those tourists that are residents of a country and visit national parks within their country.

Theory Framing

The types of tourists' theory originating from Cohen (1972) is adopted in this study to guide the understanding of domestic tourists' categorisation. Cohen's type of tourists' theory assumes that the tourists' criterion is based on ability to experience and adapt to the unfamiliar but most importantly grouped into four types of tourists and these are organised mass tourists, individual mass tourists, explorers and drifters. The below Table 1 helps to summarise the definition of these four types of tourists according to Cohen (1972).

Table 1: Four types of tourists

Types of Tourists	Definition
Organised Mass	Tends to have highest familiarity, lowest novelty, follows a tour guide and itinerary fixed in advance.
Individual Mass	Tends to have higher familiarity, lower novelty, not in a group, somehow controlled by time and itinerary.
Explorers	Tend to have lower familiarity, higher novelty, travel alone, seeks comfortable accommodation and prefers transport which is reliable.
Drifters	Has a tendency of lowest familiarity, highest novelty, highly adventurous and living within the local community.

Source: Compiled from Cohen (1972)

In addition, the application of the types of tourists' theory in tourism has been used in past studies to determine the tourists' categories. A good example is the study by Elands and Lengkeek (2000) that used Cohen's types of tourists to create a typology of experiences with findings indicating that the modes of experience of the tourists do not form separate categories. Furthermore, the applicability and validity of the typology of experiences can be within the domain of recreation and tourism (Elands & Lengkeek, 2000). However, past studies like Sharpley (1999) criticised Cohen's types of tourists' theory of not accounting for

demographic factors which determine the different types of tourists such as age, gender and income. Due to the shortfall identified by Sharpley (1999) and the need to expand on the types of tourists' theory in the context of Tanzania, this paper considers the respondents demographic information in categorising domestic tourists and in particular, the domestic tourists visiting Nyerere National Park. Therefore, in order to expand literature and build on the types of tourists' theory, this paper's main objective is to examine the categorisation of domestic tourists in Tanzania's national park using Cohen's types of tourists' theory.

Categorisation of domestic tourists

Various studies in tourism exist that have categorised tourists based on study objectives. For example, the study by Pinto *et al.* (2025) categorised tourists from the perspective of their attitude towards sustainability and degrowth. Using quantitative online survey, the study found that tourists with sustainable travel practices show greater alignment with degrowth principles (Pinto *et al.*, 2025). Another example is a study from India by Rastogi *et al.* (2015) which examined domestic tourists by categorising them based on leisure value, household income and trip frequency segmentations. Using a logit analysis, the study found that most domestic tourists in selected 6 cities of India belonged to middle-income group. In Iran, for instance, the study by Ghaffari and Rezaei (2023) focused on describing domestic tourists in terms of motives to travel to Kerman city with findings showing that some of the motive reasons are desires to visit historical-cultural attractions, shopping, and group travel. In Tanzania, the study by Bakari (2021) conducted in Serengeti National Park by focusing on domestic tourism with findings revealing that most of the respondents' characteristics showed more males (51.3%) than females (48.7%) while majority had education level of 39% for college and 24.5% for university plus earning income of Tanzanian Shillings 300,000/= and above (71.3%). Similarly, Mkwizu (2018) indicated that most domestic tourists that visited Mikumi, Udzungwa, Ruaha and Kituo national parks were middle-aged males with secondary education. However, in Tanzania, there are no studies categorising domestic tourists visiting national parks guided by the types of tourists' theory. This paper's main objective is to examine the categorisation of domestic tourists in Tanzania's national parks using the types of tourists' theory.

METHODS

The research design is cross-sectional and uses a quantitative method. This study's location

is Nyerere National Park in Tanzania. The selection of Nyerere National Park which was established in 2019 is due to its tourism potential and it is also the largest national park in Africa (Tanzania National Parks, 2025b). A simple random sampling was used so that any individual in the population has the same chance of being selected. A survey questionnaire served as the data collection instrument from 360 domestic tourists. The collected primary data was analysed using descriptive statistics assisted by [Statistical Package for the Social Sciences](#) (SPSS) version 25 to provide frequencies and percentages.

RESULTS AND DISCUSSION

This study's demographic summary of the respondents is highlighted in Table 2. The findings show that most of the respondents are of the age between 26 to 35 years (40.8%). Also, the majority of these respondents are from Dar es Salaam (54.7%) earning income above 300,000 TZS (48.6%) and are males (60.8%) with university education (38.6%). This suggest that majority of domestic tourists visiting Nyerere National Park are middle-aged males from Dar es Salaam and earn income above 300,000 TZS with university education. These findings are similar to the study by Bakari (2021) that was conducted in Serengeti National Park located in Tanzania in terms of the findings for gender, and income but not for education. Therefore, the findings on education differ from past studies by Mkwizu (2018) and Bakari (2021) which showed majority of respondents with secondary education and college education respectively whilst this study showed most of the domestic tourists visiting Nyerere National Park have university education. The variations of results between this study and that of Bakari (2021) can be attributed to the methodological approach and also geographical locations as well as choice of national parks. By geographical location differences is that the Serengeti National Park is in Northern Tanzania while Nyerere National Park is in South Eastern Tanzania.

Table 2: Demographic summary of the study sample

Variable	Frequencies (n)	Percentage (%)
Age: 18-25	72	20
26-35	147	40.8
36-45	92	25.6
46-55	36	10
Above 55	13	3.6
Residence: Dar es Salaam	197	54.7
Dodoma	16	4.4
Lindi	1	3
Morogoro	50	13.9
Others	96	24
Income: No Income	57	15.8
< 300,000 TZS	128	35.6
>300,000 TZS	175	48.6
Tanzanian Shillings (TZS)		
Gender: Male	219	60.8
Female	141	39.2
Highest Education: Primary	5	1.4
Secondary	91	25.3
College	125	34.7
University	139	38.6

Source: Field Data

To further understand the domestic tourists, this study added to the demographic information three questions which are i) first-time or repeat visitor? ii) package or non-package tour? iii) which activities in the national park? The findings in Table 3 show that the majority of the domestic tourists were characterised as first-time visitors to Nyerere National Park (61.4%) and visited mostly through package tour (58.9%). Guided by Cohen's category of tourists, this implies that these domestic tourists are seeking familiarity of the park since they are unfamiliar with the park and its activities due to them being first time visitors.

Additionally, a large portion of domestic tourists visited the park using package tours which suggests that they are visiting on pre-determined itinerary. Further findings in Table 3 reveal that most of the respondents' activities in the national park was the wildlife safari activity (90%) and few did the walking safari (7.2%). Therefore, in general, the findings from Table 2 and Table 3 guided by Cohen's category definition of organised mass tourists shows that these domestic tourists that visited Nyerere National Park can be categorised as organised mass tourists who seek familiarity rather than novelty.

Table 3: Respondents type of visit/package tour/activities

Variable	Frequencies (n)	Percentage (%)
Your visit to the national park: First-time	221	61.4
Repeat	139	38.6
Type of tourist by package tour: Package tour	212	58.9
Non-package tour	148	41.1
Activities in the national park:		
Wildlife Safari	324	90
Non-wildlife safari	36	10
Walking Safari	26	7.2
Non-walking safari	334	92.8
Boat ride	12	3.3
Non-boat ride	348	96.7
Bird watching	10	2.8
Non-bird watching	350	97.2
Fly-camping	00	00
Non-fly camping	00	00

Source: Field Data

CONCLUSION

This paper's aim was to categorise domestic tourists in Tanzania's national parks using Cohen's types of tourists' theory. By adding the demographic information in Cohen's theory in categorising domestic tourists visiting Nyerere National Park in the context of Tanzania, the findings indicate that they are mostly middle-aged males from Dar es Salaam earning income above TZS 300,000 with university education. But also, first-time visitors on package tour and mainly preferring wildlife safari. These findings imply that these domestic tourists can be categorised as organised mass tourists who seek familiarity rather than novelty.

IMPLICATIONS OF THE STUDY

On the theoretical implication is that adding the demographic information to Cohen's types of

tourists' theory in the context of Tanzania, the types of domestic tourists are categorised as organised mass tourists based on age, gender, income, education, first time visitors, package tours and activities in the national park. The practical implications based on the paper's outcome that the organised mass domestic tourists that visited Nyerere National Park are mainly university educated middle-aged males with preference for wildlife safari hence the need for tourism stakeholders particularly national park authorities to also encourage more women to visit national parks. Another practical implication is for the national park authorities to encourage both males and females to engage in other activities within the park like boat rides on the largest river in Tanzania namely river Rufiji which is one of the main attractions of Nyerere National Park.

RECOMMENDATIONS

This paper recommends that policy and decision makers to consider regular surveys of domestic tourists to determine the patterns for the types of domestic tourists as well as promotion campaigns that advertise to the public but also include advertisements that attract more women to visit national parks. The promotion campaigns should also aim at encouraging domestic tourists to engage more on other activities within Nyerere National Park.

Limitations and directions for further studies

This paper's study location was done in one national park which is Nyerere National Park in Tanzania. In addition, this paper applied cross-sectional research design. Therefore, the direction for future research can be to explore longitudinal design to understand the patterns of domestic tourists' categorisation as well as conduct a similar study on other national parks within Tanzania.

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ASSESSING TOURIST VEHICLES IN ACCOMMODATING TOURISTS WITH DISABILITIES AT NGORONGORO CONSERVATION AREA; TOUR GUIDES' PERCEPTION

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ABSTRACT

The Ngorongoro Conservation Area (NCA) is observing a rise in tourism among individuals with various physical disabilities, needing an assessment of the suitability of existing services, such as vehicle accessibility and guide readiness. This study aimed to investigate the physical disabilities of tourists encountered by guides, assess the presence of disability-friendly features in tourist vehicles, and identify the challenges faced by tour guides when assisting tourists with disabilities. A cross-sectional strategy was employed to gather data from 150 tour guides through questionnaires and interviews conducted over two weeks. The SPSS software was utilized for quantitative data analysis, including descriptive statistics and Chi-square tests to investigate connections between variables. Findings indicated that guides often interact with tourists with sensory, mobility, and neurological disabilities; yet, many stated that cars inadequately fulfill accessibility requirements. Although 86.7% of guides recognized the existence of disability-friendly features in automobiles, hardly 30% deemed them entirely appropriate for all sorts of disabilities. Challenges like communication hurdles between disabled individuals and tour guides, inadequate specialized training programs for tour guides, and restricted vehicle accessibility were frequently observed, despite 74% of guides having received some training. The report underscores a disparity between training initiatives and the practical preparedness to adequately assist disabled travelers. These findings indicate a distinct necessity for enhanced and continuous specialized training programs for tour guides, superior vehicle adaptations adhering to universal design principles, and inclusive policies that engage those with disabilities in the planning process. Improving these aspects will foster a more accessible, polite, and pleasurable tourism experience for all visitors at NCA.

Keywords: Accessibility, Ngorongoro Conservation Area, specialized training, Tourism, impairments, tour guides, vehicle adaptations.

INTRODUCTION

Accessibility of tourist vehicles in protected areas like Ngorongoro Conservation Area (NCA) is essential to ensure equal tourism opportunities for persons with disabilities. (Kivuyo, 2021) Tourism contributes significantly to Tanzania's economy, about 17.2% of GDP and employs over 1.4 million people (Francis, 2010) but disabled tourists face challenges due to lack of accessible transport (Darcy and Buhalis, 2011). Globally, 16% of people live with disabilities, yet many tourist

sites and transport services are not adapted for their needs (Sarsak *et al.*, 2024). This problem is greater in developing countries due to poor infrastructure and policies (O'Neill, 2019). In sub-Saharan Africa, only 10-12% of tourism facilities meet accessibility standards (Gisore and Ogutu, 2015). At NCA, less than 5% of vehicles are modified for disabled tourists (Cavinato and Cuckovich, 1992). Lack of accessible vehicles and poor driver training restricts disabled tourists' participation, increasing inequality and hindering SDG 10 on reducing inequality (Kett, *et al.*, 2020).

Key features like wheelchair ramps and sensory aids are often missing (Chikuta, *et al.*, 2018a). Developed countries have laws like the Americans with Disabilities Act to improve access (Kruse and Schur, 2003) but many African countries, including Tanzania, have yet to fully apply inclusive transport policies, especially in rural and conservation areas (Nizar, 2011). Tanzania's National Disability Act (2010) supports inclusion, but implementation at sites like NCA remains weak (Nizar, 2011). Addressing these gaps requires upgrading vehicles, improving driver awareness, and enhancing infrastructure, which will support social inclusion, improve tourist experiences, and boost local economies (Yamanishi, *et al.*, 2024). This study will assess vehicle accessibility at Lodare Gate, types of disabilities encountered, available vehicle features, and challenges faced by tour guides, providing insights for policy and practice aligned with Tanzania's Tourism Development Master Plan 2021–2025 and international human rights standards (Duri *et al.*, 2018). The general objective of this study is to assess accessibility of tourist vehicles for tourists with disabilities at Lodare Entrance, Ngorongoro Conservation Area. Specifically, the study deals with: i) exploring types of physical disabilities encountered by guides, ii) assess the availability of disability-friendly features in tourist vehicles, and iii) identify challenges tour guides face when assisting tourists with disabilities.

MATERIALS AND METHODS

Study Area Description

The Ngorongoro Conservation Area (NCA), located in northern Tanzania (Fig. 1), approximately 180 kilometers west of Arusha (Qdmzlcnv *et al.*, 2011) is a UNESCO World Heritage Site spanning 8,300 square kilometers (Boylan 2008). The area consists of varied

ecosystems, grasslands, forests, wetlands, and highland plateaus that support rich biodiversity, including the Big Five: elephant, lion, leopard, buffalo, and rhinoceros (Daniel, 2023). It is also internationally significant due to key paleoanthropological sites such as Olduvai Gorge and the Laetoli Footprints, which contribute to understanding human evolution (Melita, 2015). These natural and cultural features make NCA a hub for tourism, conservation, and scientific research (Batinoluho, 2019). The climate is temperate, marked by a rainy season from November to May with 600–1,200 mm of rainfall, and a dry season from June to October with temperatures ranging between 15°C and 30° (Mwabumba *et al.*, 2022). These seasonal changes influence wildlife movement and tourist patterns, especially during the rainy season when roads become impassable, posing mobility challenges for people with disabilities (Mkiramweni *et al.*, 2016).

Culturally, the NCA is home to the Maasai community, whose pastoral traditions continue to shape the landscape through grazing and cultural practices (Millanga, 2009). The Hadzabe and Datooga also reside in the area, contributing to its socio-economic diversity. The Maasai participate actively in cultural tourism by offering homestead visits, dances, and storytelling, which support local livelihoods (Masong, 2017). Despite this potential, accessibility for persons with disabilities remains a challenge due to rugged terrain, lack of ramps, adapted vehicles, or inclusive infrastructure (Chikuta, *et al.*, 2018b). These barriers limit participation not only for tourists but also for local workers with impairments. As a result, assessing the inclusivity of tourism services, especially transportation, becomes essential for equitable and sustainable tourism development aligned with national and global disability rights frameworks (Charnley, 2005).

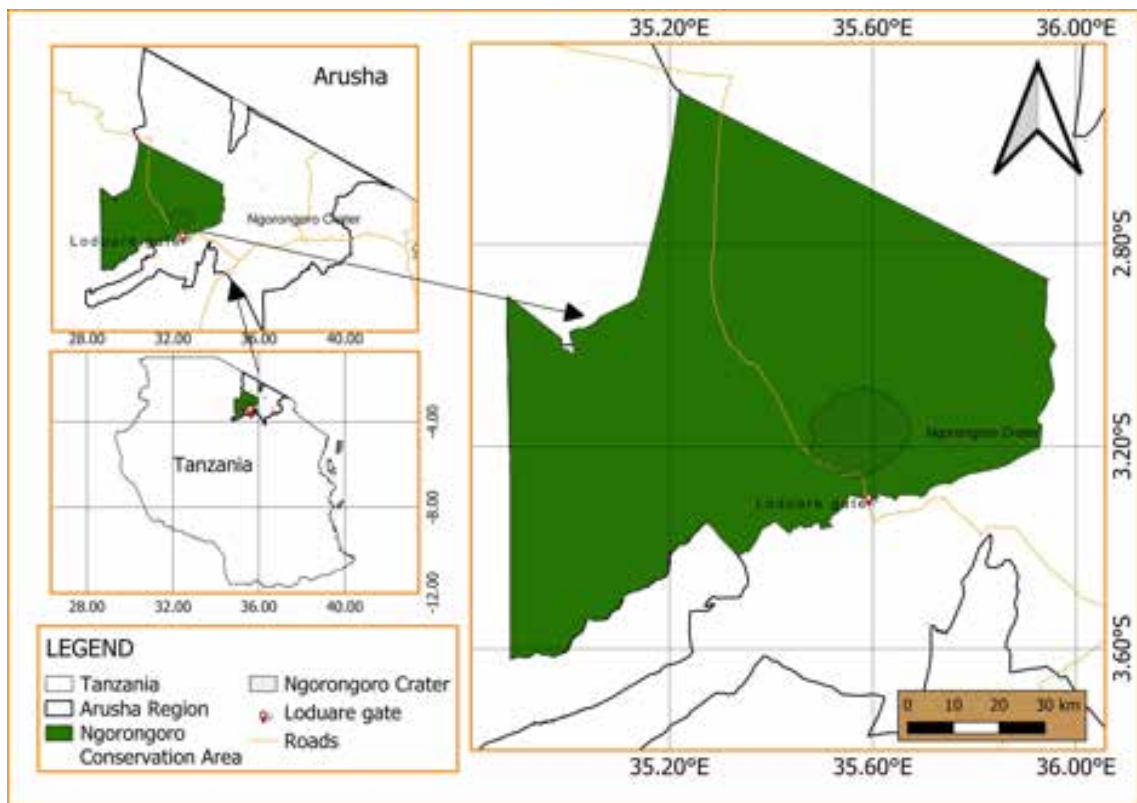


Figure 38:A Map of Ngorongoro Conservation Area, locating the main entrance gate (Lodua gate).
Source: QGIS Version 4.30.0

MATERIALS AND METHODS

The study used a mixed-methods approach combining quantitative and qualitative data to assess vehicle accessibility in Ngorongoro Conservation Area (NCA) (Levett *et al.*, 2016). Quantitative data were collected via structured questionnaires given to tour guides about accessibility features like wheelchair ramps and lifts. These were self-reported and not verified directly (Vounzoulaki 2024).

Qualitative data came from semi-structured interviews with stakeholders such as the Ngorongoro Tourism Department and Tanzania Association of Tour Operators. (Reichenberger, 2024). These explored infrastructure gaps, training needs, and challenges in vehicle modification, plus social barriers faced by tourists and providers (Kegamba *et al.*, 2024)

A cross-sectional design sampled 150 guides from about 400 vehicles, using purposive and convenience sampling at the entry points of Ngorongoro conservation area (Lodua gate) (Kothari, 2004)

Survey questionnaire included Likert-scale and open questions, with accessible formats such as verbal interviews for tour guides (Vounzoulaki 2024)

Quantitative data were analyzed using Chi-Square tests in SPSS to examine relationships between guide experience, disabilities encountered, and accessibility features (Vounzoulaki, 2024). Qualitative data underwent thematic analysis to identify key themes on vehicle suitability and policy gaps (Remillard *et al.*, 2022). Triangulation of data improved reliability and depth (Levett *et al.* 2016)

RESULTS AND DISCUSSION

Demographic Characteristics of Respondents

A total of 150 tour guides participated in the study. The majority were male (98%, n=147), with only 2% (n=3) female. Age distribution was fairly even: 26% (n=39) aged 20–29, 24%

(n=36) aged 30–39, 24% (n=36) aged 40–49, and 26% (n=39) aged 50 and above (Table 1). Regarding experience, 22.7% had less than one year, 22% had 1–3 years, 22.7% had 4–6 years, 20% had 7–10 years, and 12% had over 10 years of guiding experience. Most (82%, n=123) were employed by registered tour companies, while 18% (n=27) were self-employed.

Table 11: Demographic Information

Attributes	Category	Frequency	Percentage
Gender	Female	3	2.0
	Male	147	98.0
	Total	150	100.0
Age group	20–29	39	26.0
	30–39	36	24.0
	40–49	36	24.0
	50 and above	39	26.0
	Total	150	100.0
Experience	Less than 1 year	34	22.7
	1–3 years	33	22.0
	4–6 years	34	22.7
	7–10 years	30	20.0
	More than 10 years	19	12.0
	Total	150	100.0
Employment	Self-employed	27	18.0
	Tour company	123	82.0
	Total	150	100.0
Guided disabled tourist	No	31	20.7
	Yes	119	79.3
	Total	150	100.0

Source: Field Data 2025

Disabilities Encountered by Tour Guides

Nearly all guides (98%) reported encountering tourists with various disabilities. Sensory impairments were most common (35.3%), followed by mobility impairments (15.3%) and neurological or neuromuscular conditions

(12.7%) (Fig. 2). Less frequent were cognitive and speech impairments. Statistical analysis showed a significant association between guides' years of experience and the diversity of disabilities encountered ($\chi^2 = 172.314$, $df = 35$, $p < .001$), with more experienced guides reporting a broader range.

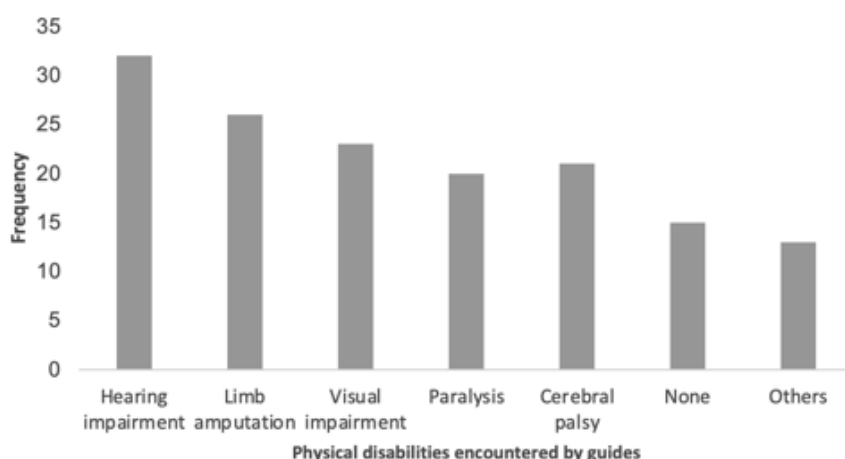


Figure 2: Tourist physical disabilities encountered by guides; Source: Field Data 2025

Availability of Disability-Friendly Vehicle Features

Most respondents (86.7%) indicated that some accessible features were present in tourist vehicles (Fig.3), but only 30% considered them fully suitable for disabled tourists. Moderate satisfaction was expressed for ease of entry/exit, mobility aid support, and adjustable seating.

However, features facilitating communication with visually or hearing-impaired tourists were rated poorly by over half of respondents. The type of disability significantly influenced perceptions of vehicle suitability ($\chi^2 = 155.558$, $p < .001$). Overall, vehicle accessibility in the Ngorongoro Conservation Area remains inadequate. Photo 1 and 2 indicates some accessibility to vehicles.



Figure 3: Disability-friendly features in tourist vehicles at NCA Source: Field Data 2025

Photo 1: Entry/Exit access in tourist vehicles **Photo 2:** Space for mobility device and Seat adjustment in tourist vehicles

Challenges Faced by Tour Guides

The majority of guides reported challenges in assisting tourists with disabilities. Key issues included communication barriers, lack of appropriate facilities, and limited availability of accessible vehicles. Overwork

and insufficient training were also cited. About 74% had received some training on disability awareness, and statistical testing revealed that those without training were significantly more likely to report difficulties ($\chi^2 = 158.009$, $p < .001$).

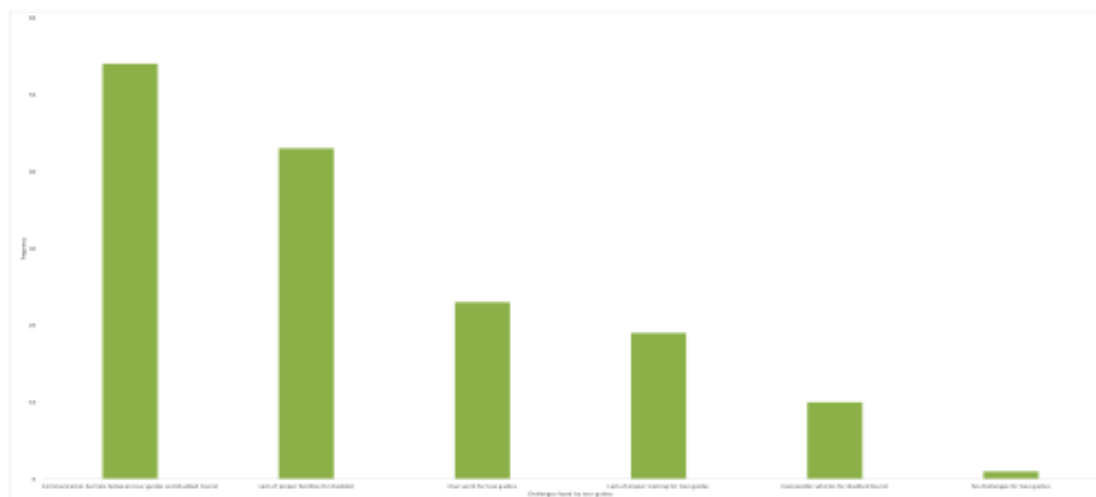


Figure 39: Challenges faced by tour guides in handling tourists with disabilities at NCA

Source: Field Data 2025

Thematic Interview

Table 12: Thematic analysis

Themes	Respondents	Quotations
Tourist Physical Disabilities	TATO, Tourism officer NCA	“Some tourists arrive with wheelchairs and hearing impairments, which require special attention.”
	Tourism Officer Karatu	“We’ve observed a growing number of tourists with visual and mobility impairments.”
	Tourism Officer NCA	“Handling tourists with cerebral palsy and amputations is common.”
Disability-Friendly Vehicle Features	TATO	“Only a few vehicles are modified with ramps or wider doors.”
	Tourism Officer Karatu	“Some tour companies have invested in adjustable seats and visibility tools.”
	Tourism Officer NCA	“Wheelchair space and special seat belts are limited.”
Challenges in Handling Tourists with Disabilities	TATO	“Lack of training for guides on disability issues.”
	Tourism Officer Karatu	“There is no standardized guideline for handling disabled tourists.”
	Tourism Officer NCA	“Communication with hearing-impaired tourists is difficult.”

Source: Field Data 2025

DISCUSSION:

Demographic Characteristics of Respondents

Tour guides at NCA are predominantly male, limiting gender-sensitive service delivery for disabled female tourists (Vermiglio *et al.* 2021). Experience, rather than age, influences a guide's competence in accessibility (Darcy and Buhalis 2011). Inexperienced guides lack the confidence to manage disability-related needs, unlike their more experienced counterparts in places like the Maasai Mara (Talle 1995). While many guides are affiliated with formal companies, their vehicles still lack adequate modifications (Loi and Kong 2017). Overall, gender imbalance, limited experience, and inadequate vehicle infrastructure hinder inclusive tourism at NCA (Darcy and Buhalis 2011).

Tourist physical disabilities encountered by guides while handling tourists at the Ngorongoro Conservation Area (NCA)

Tour guides frequently face a variety of tourist impairments, such as mobility, sensory, and neurological conditions. Although these encounters vary, they are a regular part of guiding duties, necessitating strong preparation. Experienced guides handle disabilities more effectively (Awan *et al.* 2022) but many guides lack training. Structured and ongoing training improves both technical and interpersonal competence (Lee and Studies 2016). To enhance inclusive service, NCA needs regular, targeted disability training and mentoring (Chikuta, Plessis, and Saayman 2018a).

Availability of disability-friendly features in tourist vehicles at NCA

Most tour guides report that accessibility features are present in tourist vehicles, but few find them fully functional or suitable. Key aspects like entry, seating, and mobility support are moderate, and features for sensory impairments are especially inadequate (Coetzee

and Town 2016). The lack of universal design and standardization weakens overall accessibility (Crews and Zavotka 2005). Feedback from users with diverse impairments should guide future designs (Putri 2024). A more integrated and regularly evaluated vehicle accessibility system is needed at NCA (Hutchinson *et al.* 2020).

Challenges faced by tour guides in handling tourists with disabilities at Ngorongoro Conservation Area (NCA)

Guides often lack advance information about tourists' disabilities, leading to unpreparedness and service breakdowns, especially with sensory impairments (Darcy and Buhalis 2011). The absence of accessible signage, brochures, or digital information restricts both guide readiness and tourist autonomy (Chikuta, Plessis, and Saayman 2018a). Even when training exists, it often excludes communication strategies. Communication is critical to quality service (Chikuta, Kabote, and Chikanya 2017). Solutions must include improved pre-tour coordination, accessible materials, and communication tools (Cloquet *et al.* 2018).

CONCLUSIONS AND RECOMMENDATIONS

Conclusion

Accessibility of tourist vehicles in Ngorongoro Conservation Area remains insufficient for tourists with disabilities, limiting their full participation in tourism activities. Challenges include lack of disability-friendly vehicle features, inadequate guide training, and poor awareness of disability needs, which undermine Tanzania's goals for inclusive and sustainable tourism. Tour guides face difficulties accommodating various disabilities due to limited experience and communication barriers, highlighting the need for universal design and comprehensive accessibility strategies. Sustainable improvements require combining infrastructure upgrades with capacity building,

attitudinal change, and inclusive policies involving persons with disabilities. Addressing these gaps will promote equitable and dignified tourism experiences in NCA (Persson *et al.*, 2015)

Recommendations

Based on the study's results and management implications, the following recommendations are made to advance accessible tourism at NCA:

- i. Implement mandatory, ongoing specialized training courses for all tour guides, emphasizing practical skills, communication strategies, disability protocol, and problem-solving to improve their confidence and ability in assisting impaired tourists.
- ii. Enhance and standardize tourist vehicles and facilities to comply with universal design principles in order to ensure comprehensive accessibility for tourists with diverse disabilities, including mobility, sensory, and cognitive impairments.
- iii. Build partnerships with disability organizations to involve individuals with disabilities actively in the planning, training, and evaluation processes to ensure that their lived experiences influence service design and delivery.
- iv. Foster an inclusive culture within NCA and partner organizations by integrating disability awareness into organizational values, policies, and regular operations, supported by leadership commitment and ongoing sensitization initiatives.
- v. Implement systematic monitoring and evaluation frameworks that regularly evaluate the efficiency of training, vehicle accessibility, and service quality, facilitating rapid modifications and continual improvements in accessible tourist delivery.
- vi. Encourage more studies utilizing qualitative and participative methodologies to directly examine the experiences of tourists with

disabilities. By implementing these ideas, NCA may advance towards creating profound understandings that will enhance the inclusive tourism strategy.

- vii. A premier model of accessible and inclusive tourism in Africa, according to international best practices, and guaranteeing that all guests, regardless of ability, are able to participate in a dignified and fulfilling experience.

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SUSTAINABLE TOURISM AND WOMEN IN HIGHER EDUCATION: PERSPECTIVES ON VISITS TO NATIONAL PARKS

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ABSTRACT

Sustainable tourism has been a growing phenomenon, and previous scholars have linked sustainability and national parks in the post-pandemic context. In addition, earlier studies have shown that fewer females than males visited national parks in Tanzania. To expand the literature on this topic, the main objective of this paper was to explore sustainable tourism and women in higher education from the perspective of visits to national parks. Guided by motivation theory, the specific objective was to conceptualize sustainable tourism and female academics in higher education, focusing on visits to national parks within the Tanzanian context. A literature review approach was applied as the methodology. The findings revealed that the existing literature consistently showed that fewer females visited national parks. The practical implication was that tourism stakeholders should encourage females, including female academics, to visit national parks more frequently as a means of relaxation as well as a potential retreat incentive. This paper is limited to a literature review approach; therefore, future studies may adopt mixed methods and collect primary data to further understand sustainable tourism and female academics in higher education in Tanzania.

Keywords: higher education, national parks, sustainable tourism, women

INTRODUCTION

Tourism as an industry has recovered worldwide after the pandemic. According to the United Nations (UN) Tourism, the international tourism increased up to 5% in the first half of 2025 despite the global challenges such as high transport and accommodation costs (UN Tourism, 2025). Africa as a continent including its tourism industry is also progressing. In fact, Africa recorded the strongest tourism performance in the first half of 2025 with an increase of 12% but also with double-digit growth of 14% for North Africa while for Sub Saharan Africa (SSA) showed 11% (UN Tourism, 2025). This implies that Africa is performing well and Tanzania is no exception. Tanzania's tourism earnings increased from USD 3.3 million in 2023 to USD 3.9 million in 2024 plus increased tourist arrivals 2,141,895 in 2024 from 1,808,205 in 2023 (National Bureau

of Statistics Tanzania, 2025). Sustaining the strong performance in tourism growth also requires sustainable tourism.

Sustainable tourism is a trending phenomenon. For instance, past scholars in East Africa like Kimeto and Mkwizu (2023) have connected sustainability and national parks in the post pandemic. The study emphasized on the need to retrain on issues of customer care in order to have sustainable tourism in terms of services after the pandemic. In addition, past studies in Tanzania by Mariki *et al.* (2011), Mkwizu (2023a, 2023b, 2023c) and Mkwizu *et al.* (2023) have shown that there are less females compared to males that visit national parks in Tanzania. For instance, the study by Mariki *et al.* (2011) revealed that in the northern tourist circuit there were more men visiting protected areas compared to women. Similarly, Mkwizu (2023a) indicated that there were 60.8% males

compared to 39.2% females that visited Nyerere National Park. Therefore, there is a challenge of few women visiting national parks. To expand literature on this topic, the main objective of this paper is to explore sustainable tourism and women in higher education from the perspective of visits to national parks. Guided by the motivation theory, the specific objective is to conceptualize sustainable tourism and female academics in higher education focusing on visits to national parks from the context of Tanzania.

Higher Education, according to UNESCO (2025), is a cultural and scientific asset comprising post-secondary institutions that enable personal development, promote economic and social change, and equip students with skills for labor markets. This definition highlights inclusive access and skills building, which is particularly relevant for women in developing contexts such as Tanzania. Further, Alemu (2018) explains that higher education encompasses a wide range of institutions that offer advanced learning, training, and research opportunities, including universities, polytechnics, and professional colleges. It is characterized not only by the transmission of knowledge but also by the production and dissemination of new knowledge through research. Furthermore, Tilak (2025) highlights that higher education institutions are centers of innovation and societal transformation, developing knowledge and skills critical to economic and social progress. Smith and Brown (2025), define higher education as inclusive platforms that promote critical citizenship, lifelong learning, and professional competencies. In the context of this study, the women in higher education are those female academics with college and university education.

Eagles *et al* (2025) define national parks as natural areas managed primarily for ecosystem preservation and heritage, integrated with environmentally responsible tourism. In the same vein Parker and Daniels (2025) describe national parks as legally protected landscapes

with significant ecological and cultural values, requiring management that supports conservation alongside compatible recreational use. Additionally, Manning (2025) defines national parks as government-regulated lands established to conserve native species and habitats, support sustainable visitor experiences, and foster environmental stewardship through outreach and education. Moreover, Ncube and Mwasha (2024), highlight national parks as treasured environmental and cultural commons that aim to balance biodiversity conservation with local community benefits and sustainable tourism development. Likewise, Mkwizu (2023a) described national parks as attractions that contribute to tourism revenues for nations worldwide. This study adopts the definition from Eagles *et al.* (2025), as it explicitly integrates ecosystem protection, cultural heritage, and education within sustainable tourism frameworks, aligning closely with the focus on Tanzania's national parks as sites for educational and conservation-related visits by women in higher education.

Weaver (2025) defines sustainable tourism as tourism development that supports the long-term ecological balance, cultural integrity, economic viability, and social equity of host communities. It emphasises minimising environmental impacts while maximising community benefits and visitor satisfaction. Similarly, Gomez and Li (2025) explain sustainable tourism as a process that integrates environmental conservation with cultural preservation and economic empowerment, ensuring tourism activities do not degrade the natural or social fabric but provide ongoing benefits to local people. Further, Honey (2024) describes sustainable tourism as practices that maintain biological diversity and ecosystem health, foster environmental education, and promote ethical engagement between tourists and local cultures. Furthermore, Kamau and Mutiso (2024) define sustainable tourism as tourism that balances conservation goals with socio-economic development, engaging local communities in planning and benefits sharing to build resilient tourism systems. This

study adopts Weaver's (2025) comprehensive definition for sustainable tourism, as it aligns with the multidimensional sustainability goals relevant to Tanzania's national parks, where tourism intersects with ecological conservation and social empowerment through women's educational participation.

METHODOLOGY

This study adopts the motivation theory as a guide in exploring sustainable tourism and female academics in higher education visiting national parks. The motivation theory selected for this study is the Maslow's hierarchy of needs. The Maslow's hierarchy of needs theory was developed by Abraham Maslow in 1940 which assumes that the needs of humans starts with basic needs to complex needs (Bandhu *et al.*, 2024). The hierarchy starts with physiological needs, safety needs, belonging needs, esteem needs and self-actualization needs (Bandhu *et al.*, 2024). On the other hand, According to Bandhu *et al.* (2024), motivation is described as a yearning to accomplish a task. This study believes that Maslow's hierarchy of needs has left out the act of visiting national parks which is crucial for relaxing and enjoying nature as well as the sustainability of national parks hence sustainable tourism.

Literature reviews as methods have been used in research designs (Kraus *et al.* 2022) Likewise, this study adopts a literature method by selecting the Integrative Literature Review (ILR) as the methodology approach. In addition, Lubbe *et al.* (2020) stated that the strength of an ILR approach lies in summarising the findings of multiple types of evidence addressing a problem. The identified problem in this study is that of few women visiting national parks.

Therefore, in this particular paper, ILR is ideal for purposes of exploring sustainable tourism and women in higher education from the perspective of visits to national parks. This study follows the five steps of ILR by Torraco (2005) which are problem formulation, literature search, evaluation of data, data analysis, interpretation and presentation of results.

The first step involved formulating the problem question emanating from the specific objective which is to conceptualise sustainable tourism and female academics in higher education focusing on visits to national parks from the context of Tanzania. The question is "What is the conceptualisation of sustainable tourism and the distribution of female academics in higher education visiting national parks in Tanzania? The second step involves literature search and this study selected the university repository of The Open University of Tanzania (OUT) for the period of 10 years from 2015 to 2025 for PhD theses in tourism under the Department of Tourism and Hospitality (<https://repository.out.ac.tz/view/divisions/dep=5Fth/>).

Hence, literature search inclusion and exclusion criteria consider those studies not listed in the OUT repository as excluded from this study. Furthermore, only PhD theses conducted in Tanzania are included. The reason for selecting only the repository at OUT serves as the limitation of this study so that future research can explore other sources of data and expand this study. A total of 5 studies are selected and listed as per Table 1 from the OUT repository which are evaluated and analysed using content analysis. The information in Table 1 is also interpreted and further discussed in the findings and discussion section.

Table 1: OUT repository PhD theses in tourism (2015 to 2025)

N	Topic and Source	Method	Findings
1	Assessment of status and prospect of domestic tourism in schools in Tanzania Batinoluho (2017)	Mixed methods (qualitative and quantitative)	The school children, teachers, parents, guardians and key informants with college and university education are 192. Out these, the study showed for parents and guardians (66.5%) were unaware of domestic tourism issues.
2	Tanzania citizens' participation in domestic tourism: A social exchange perspective. Macha (2017)	Quantitative)	Relationship between gender and level of education indicates that participation in domestic tourism was significant and data analysis indicates medium effects.
3	The influence of television advertising on domestic tourism: A case study of southern tourist attractions in Tanzania. Mkwizu (2018)	Mixed methods (quantitative and qualitative)	Most of domestic tourists (92.2%) watched TV programs on national parks. There were 36% females compared 64% males who visited Mikumi, Udzungwa, Ruaha and Kitulo national parks. Education distribution of these domestic tourists showed university education (28.8%) and then college education (22.8%).
4	Assessment of Rural Tourism Initiatives as Strategy for Poverty Reduction in Tanzania: A Case of Bagamoyo and Kisarawe Districts. Mungure (2024)	Mixed methods (quantitative and qualitative)	Bagamoyo and Kisarawe respondents perceived rural tourism development as a multiplier effect. Concluded that the implementing rural tourism initiatives is a strategy to a new market opportunity bringing sustainability of poverty reduction.
5	The relationship between airport service performance and international tourists' satisfaction. Mwageni, (2024)	Quantitative	Findings indicate that processes of arrival, departure and departure non-processing domains have positive path coefficients with direct and significant relationships toward international tourists' satisfaction.

RESULTS AND DISCUSSION

The findings indicate a persistent gender disparity in national park visitation, with significantly fewer females compared to males. For instance, Rachel Mkwizu (2018) reported that only 36% of visitors to Mikumi National Park, Udzungwa Mountains National Park, Ruaha National Park, and Kitulo National Park were female, compared to 64% male visitors. These results align with earlier studies (Martha Mariki *et al.*, 2011; Rachel Mkwizu, 2023a), suggesting a consistent pattern of low female engagement in national park tourism. While research such as George Batinoluho (2017) examined visitor participation broadly, there remains a notable gap in understanding the behavior of female academics in higher education, as highlighted by Macha (2017), Mungure (2024), and Mwageni (2024). From the perspective of Motivation Theory, this low participation underscores the need for targeted interventions, including incentive programmes, awareness campaigns, and training that emphasize the personal and professional benefits of visiting national parks. Such strategies may enhance female engagement while fostering broader tourism development. Moreover, although previous studies did not explicitly address sustainable tourism, this study situates female academics' visitation within David Weaver's (2025) comprehensive conceptualization, which emphasizes multidimensional sustainability goals. Integrating ecological conservation and social empowerment, this perspective positions increased female participation as both a pathway to inclusive tourism and a contribution to the sustainable management of Tanzania's national parks.

CONCLUSION, IMPLICATIONS AND SUGGESTIONS

This paper concludes that there is evidence of less females than males visiting national parks which implies that there are few female academics in higher education visiting national park. The study has also found that more

research is needed to avail the distribution of female academics in higher education in terms of those with college and university education visiting national parks. On the other hand, conceptualising sustainable tourism can be adopted from Weaver (2025) due to the multidimensional sustainability goals particularly ecological conservation and social empowerment which may also be relevant to Tanzania's national parks to consider efforts in encouraging more women to visit national parks. Based on the outcome of this paper, the practical implication is for the tourism stakeholders to encourage females including female academics in higher education to visit national parks more often as a way for relaxing but also as a retreat incentive. This paper is limited to the integrative literature review approach only and the use of OUT repository as the literature search. Therefore, future studies may use other literature search sources as well as mixed methods and arrange for primary data collection in order to further understand sustainable tourism and female academics in higher education within Tanzania.

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THE LONG-TERM MONITORING OF LIVESTOCK PREDATION BY LARGE CARNIVORES IN THE MKOMAZI SUB-LANDSCAPE, TANZANIA

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ABSTRACT

Human–wildlife conflict (HWC), particularly livestock predation by large carnivores, remains a major challenge to wildlife conservation and rural livelihoods in Tanzania. This study aimed to assess the trends, economic impacts, and effectiveness of mitigation measures for reducing human–large carnivore conflicts across selected areas managed in collaboration with the Tanzania Wildlife Management Authority (TAWA), Tanzania National Parks Authority (TANAPA), and Tanzania Forest Services Agency (TFS). Data on livestock predation incidents were collected from 2020 to 2025. Incident reporting was conducted through manual human compensation forms completed by Village Game Scouts (VGS) from 2020 to 2023 and later transitioned to digital data collection using mobile-based platforms (Miombo and Open Data Kit) from 2024 to 2025. A total of 79 livestock predation incidents were recorded during the study period. Results indicate that 1,155 livestock were killed, resulting in an estimated economic loss of approximately TSh 712 million (~USD 285,000). Incidents occurred almost equally in open fields (49%) and livestock enclosures (51%), with the majority (76%) taking place during the dry season. Hyenas accounted for most attacks (58%), followed by lions (23%), leopards (17%), and other carnivores. Various mitigation interventions were implemented, including predator-proof bomas (chain-link fencing), solar lights and torches, guarding with dogs, and traditional deterrents such as fire, sticks, spears, and arrows. Overall, the applied mitigation measures were associated with a 93% reduction in livestock predation. The study concludes that integrated community-based monitoring and improved husbandry practices significantly reduce HWC impacts, highlighting the importance of strengthening early warning systems and scaling up effective predator deterrent technologies.

Keywords: Human-wildlife conflict, Human–larger carnivore conflict, livestock predation, coexistence, and Mkomazi ecosystem

INTRODUCTION

Human–carnivore conflict is a major conservation and livelihood challenge in landscapes adjacent to protected areas, threatening large carnivore conservation and imposing substantial economic costs on local communities through livestock depredation, crop damage, property loss, and occasional human injury (Nyahongo, 2007; Dickman, 2008; Bauer et al., 2015a). Livestock depredation is the most common form of human–wildlife conflict and often triggers retaliatory killing of carnivores, particularly

in pastoral systems where livestock are central to household livelihoods and cultural identity (Holmern et al., 2007; Inskip & Zimmerman, 2009). In the Mkomazi sub-landscape, major livestock predators include spotted hyenas, lions, leopards, cheetahs, and African wild dogs, with conflicts exacerbated by habitat fragmentation, free-ranging grazing, and inadequate livestock protection (Kissui, 2008; Mbise et al., 2018). Although mitigation measures such as predator-proof bomas, improved livestock husbandry, and guarding practices are widely promoted, their long-term effectiveness remains poorly understood. This study therefore assessed long-term livestock

predation trends, predator species, affected livestock, spatial and temporal patterns, economic losses, and the effectiveness of mitigation measures to inform evidence-based strategies for improving human–carnivore coexistence in the Mkomazi sub-landscape

MATERIALS AND METHODS

Study Area

The study was conducted in the Mkomazi sub-landscape (Fig.1), northeastern Tanzania (4°17'58"S, 38°23'22"E), covering approximately 11,954 km². The area comprises diverse ecosystems, including savannahs, bushlands, woodlands, forests, and wetlands, supporting rich biodiversity. Key wildlife

includes the African elephant (*Loxodonta africana*), black rhinoceros (*Diceros bicornis*), giraffe (*Giraffa camelopardalis*), plains zebra (*Equus quagga*), African buffalo (*Syncerus caffer*), and large carnivores such as Lion (*Panthera leo*), Leopard (*Panthera pardus*), Cheetah (*Acinonyx jubatus*), and African wild dog (*Lycaon pictus*), as well as over 450 bird species. The sub-landscape has a semi-arid climate with bimodal rainfall and distinct dry seasons that influence wildlife movements and human–wildlife interactions. According to the 2022 Population and Housing Census, approximately 1.22 million people live in the area, with livelihoods primarily based on livestock keeping, crop farming, beekeeping, fishing, small-scale enterprises, natural resource use, and tourism.

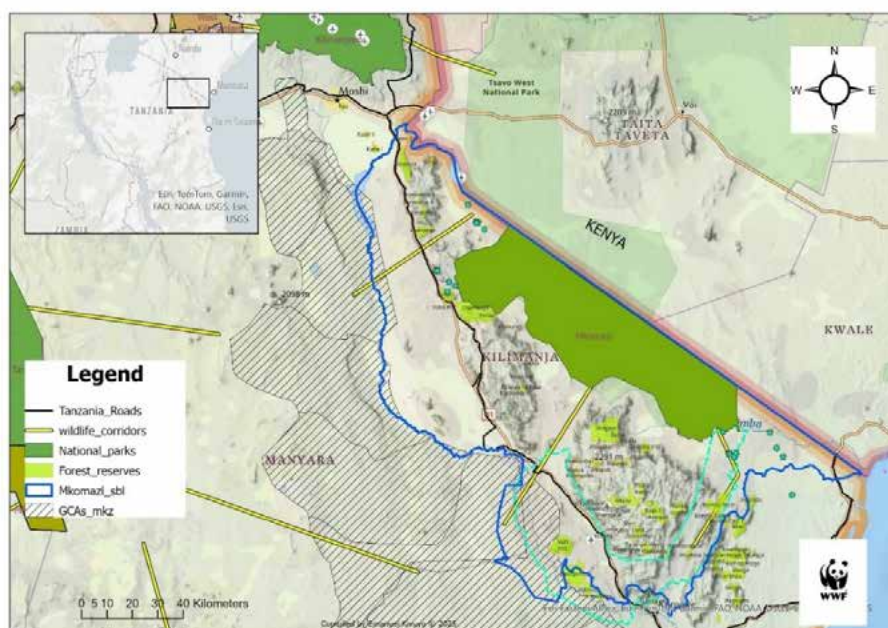


Figure 1. Map of the Study Area (Mkomazi sub-landscape)

Data Collection Techniques

Human–wildlife conflict (HWC) data were collected from 2020–2025 across 26 hotspot villages. From 2020–2023, incidents were recorded manually by Village Game Scouts (VGS) under District Game Officers (DGOs)

using standardized forms and verified by relevant authorities before entry into a centralized compensation database. From 2024–2025, data collection shifted to a digital system using a customized Open Data Kit (ODK) mobile application (Miombo), allowing

real-time entry by trained staff from TAWA, TANAPA, and TFS, with data synchronized to a central server and routinely validated for accuracy

Data Analysis

The cleaned and validated dataset was analyzed in Microsoft Excel using descriptive statistics, trend analysis, and visualizations to examine livestock depredation patterns, spatial distribution, and carnivore-specific attacks. Inferential analyses, including Chi-square tests, binomial tests, Pearson correlation, and one-way ANOVA, were performed to assess differences and relationships in predation patterns across carnivore species, livestock types, and time at appropriate significance levels.

RESULTS

Trend of livestock depredation by wild carnivores

Human–large carnivore conflict (HLC) incidents decreased from 2020 to 2022 and then rose from 2023 to 2025 (Fig.2). The early decline is likely reflected in underreporting and limited field monitoring, while the later increase is associated with improved reporting systems, strengthened Village Game Scout capacity, and digital data collection. Statistical tests confirmed significant temporal variation ($\chi^2 = 699.7$, $df = 5$, $p < 0.001$; $F(5, 30) = 3.17$, $p = 0.021$), with peak incidents recorded in 2024–2025.

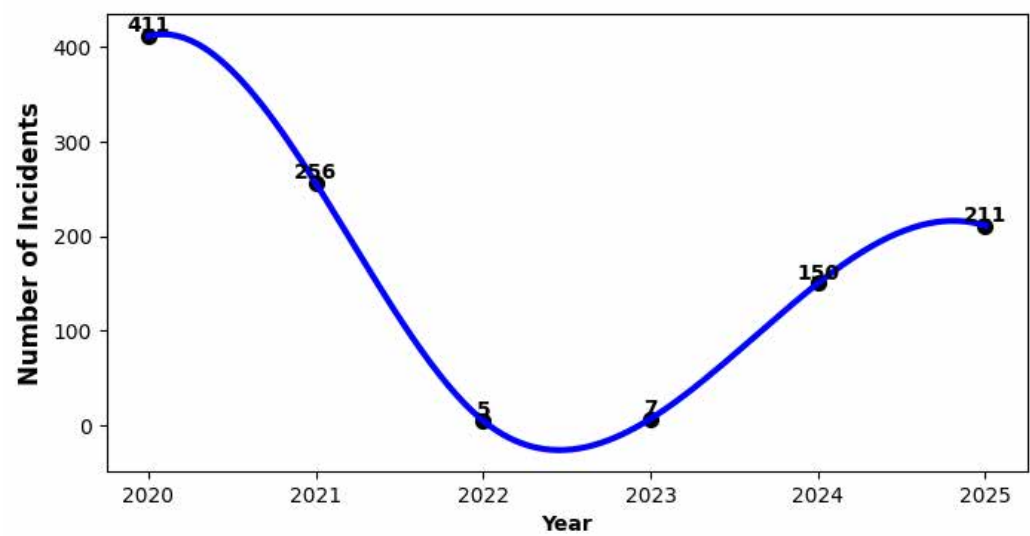


Figure 2. Graph showing the number of livestock predations in the Mkomazi sub-landscape

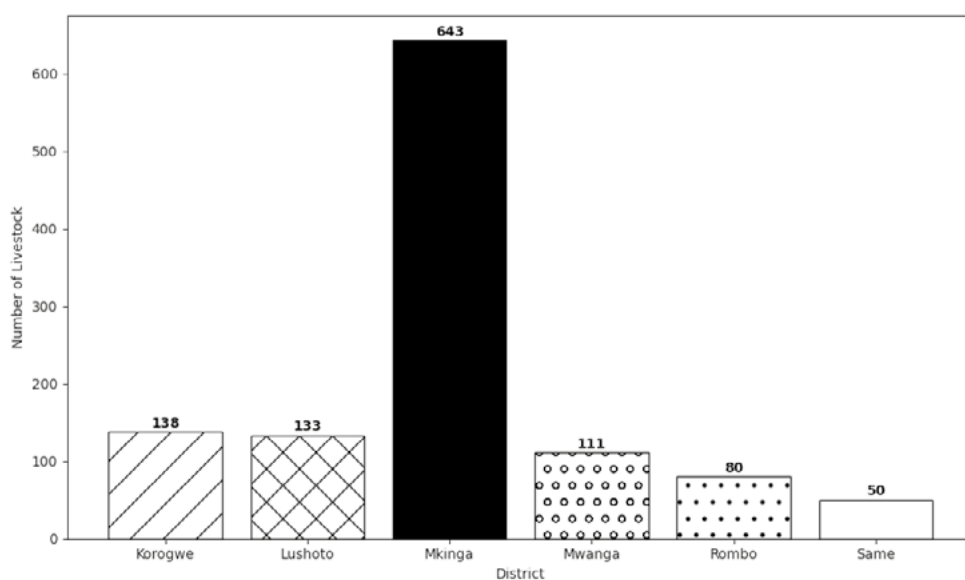
Spatial Distribution of Livestock Predation

A total of 1,155 livestock were killed by wild carnivores across six districts between 2020 and 2025 (Table 1), with marked spatial and temporal variation. Predation was highest in 2020–2021, declined sharply in 2022–2023, and increased again in 2024–2025. Mkinga recorded the highest losses, followed by

Korogwe and Lushoto (Fig. 3). The early peak may be associated with drought-related pressures, whereas the decline likely reflects underreporting, improved livestock protection measures, and reduced predator activity. The subsequent increase may be attributed to improved reporting and changing predator–livestock interactions.

Table 1. Spatial Distribution of Livestock Predation in Mkomazi from 2020 to 2025

Districts/Years	2020	2021	2022	2023	2024	2025	Grand Total
Korogwe					69	69	138
Lushoto				13	40	80	133
Mkinga	383	189			7	64	643
Mwanga	93	15				3	111
Rombo	1	51			28		80
Same	9	1	5		22	13	50
Grand Total	486	256	5	13	166	229	1155

**Figure 3.** Graph showing the number of livestock predations in the six districts

The heatmap and trend analyses revealed significant spatial and temporal variation in livestock depredation across the six districts (Fig 4). Predation peaked in 2020–2021, particularly in Mkinga, declined markedly in 2022–2023, and increased again in 2024–2025, especially in Mkinga, Korogwe, and Lushoto (Fig.4). A Chi-square test confirmed significant

differences in depredation across districts and monitoring periods (χ^2 , $df = 10$, $p < 0.001$), indicating that human–wildlife conflict in the Mkomazi landscape is dynamic and influenced by ecological, climatic, and human factors.

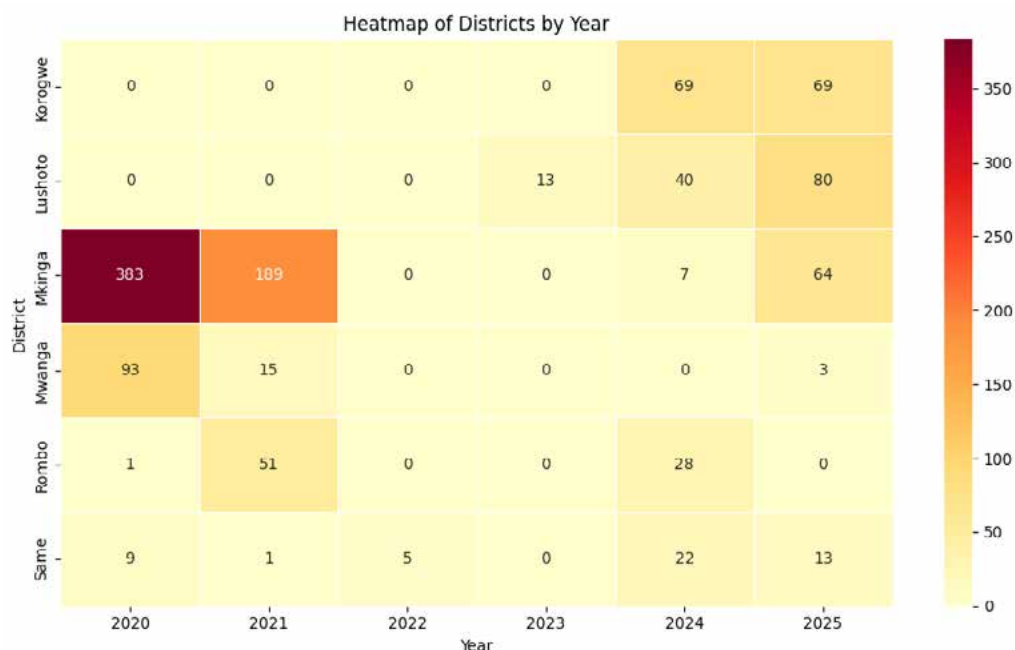


Figure 4. The heatmap showing livestock depredation incidents in Mkomazi

Livestock Species Affected by Wild Animals

Table 2. Wild animals involved in livestock depredation (N=7)

Wild Carnivores	Baboon	Cheetah/ jackal	Hyena	Jackal	Leopard	Lion	Others
Number of incidents	2	2	244	3	70	98	1
Proportion (%)	0%	0%	58%	1%	17%	23%	1%

Table 2 shows that spotted hyenas were responsible for most livestock predation incidents (58%, $n = 244$), followed by lions (23%, $n = 98$) and leopards (17%, $n = 70$), while other carnivores contributed minimally (0–2%). The Chi-square goodness-of-fit test confirmed significant differences in predation among carnivore species ($\chi^2 = 814.3$, $df = 6$, $p < 0.001$), highlighting spotted hyenas as the primary livestock predator.

Carnivore Species Responsible for Predation

A total of 1,155 livestock were attacked by wild carnivores, with goats (43%) and cattle (41%) being the most affected, followed by sheep–goat combinations (11%) and sheep (5%) (Fig. 5). The Chi-square test showed significant differences in predation among livestock types ($\chi^2 (3, N = 1155) = 543.3$, $p < 0.001$), indicating that goats and cattle were significantly more vulnerable to predation than sheep

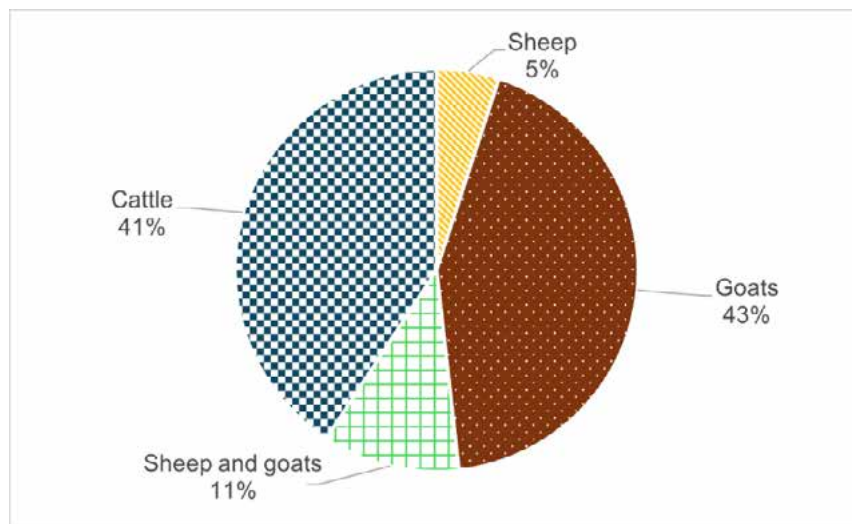


Figure 5. Proportion of livestock attacks

The Season for the livestock predation incidents

Most livestock predation incidents occurred during the dry season (76%), compared to the wet season (24%). A binomial test confirmed that this seasonal difference is statistically significant ($p < 0.001$), indicating higher predation rates during dry periods.

Location of the Livestock Depredation by Wild Animals

About 51% of livestock depredation incidents occurred inside enclosures (kraals) and 41%

outside in grazing areas. However, the difference was not statistically significant (binomial test, $p = 0.92$), indicating that predation occurred equally in both locations.

Economic Impact of Livestock Depredation

The total estimated economic loss from livestock predation was Tanzania Shillings (TZS) 712,080,750 (~ US\$284,832). Although goats were most frequently affected, cattle caused the highest financial losses (TZS. 481,775,000; $\approx$ US\$192,710) due to their higher market value, making them the main driver of economic impact (Fig 6).

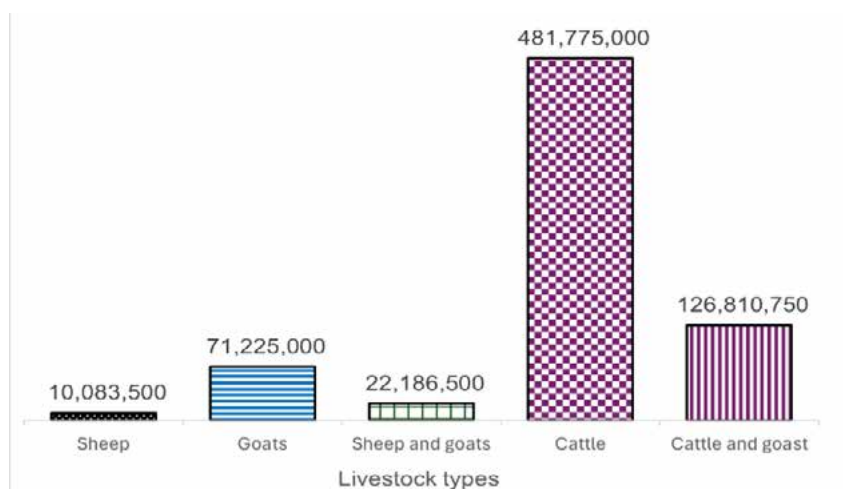


Figure 6. Estimated economic cost for livestock predation in Tanza.

The Pearson correlation analysis revealed a very weak, non-significant relationship between the number of livestock lost and the estimated economic cost ($r = 0.094$, $p = 0.881$). This indicates that economic losses are determined primarily by livestock type and market value rather than by the number of animals killed, with cattle incurring the highest costs despite fewer losses. The scatter plot (Fig. 7) further supports this weak relationship.

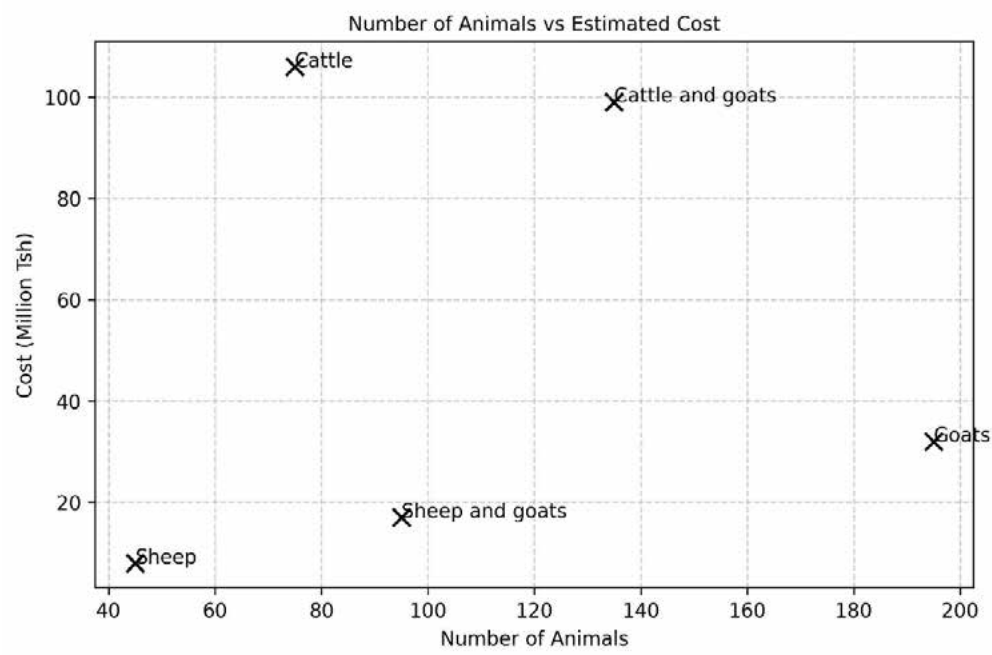


Figure 7. Scatter Plot: Number vs Cost

Mitigation Measures Employed by Local Communities for Livestock Prevention

Local communities in the Mkomazi sub-landscape use various measures to reduce livestock depredation, mainly torches and solar lights (24%), followed by torches, dogs, and fire (20%) (Fig.8). Livestock are also actively guarded during grazing and kept in vegetation-built bomas at homesteads. Since 2022, WWF has supported households with predator-proof bomas (PPBs) and improved lighting systems to enhance livestock protection.

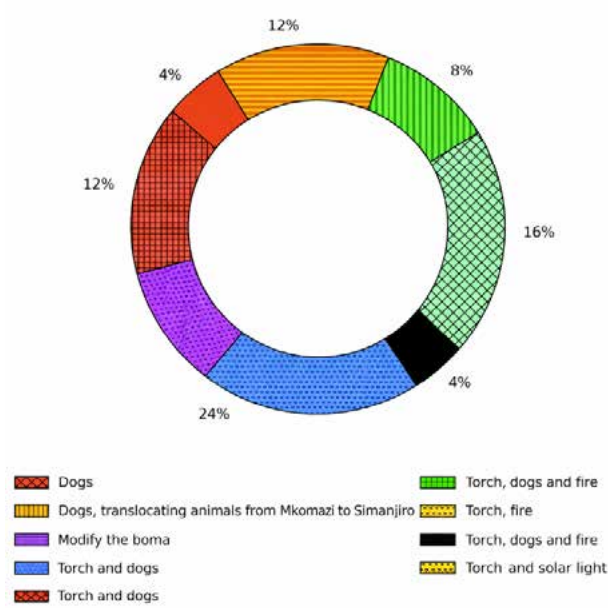


Figure 8. HLC mitigation measures employed by the communities

Effectiveness of the HLC mitigation measures in reducing livestock depredation

An assessment of 40 supported HLC mitigation sites found that 87% experienced wildlife incursions, primarily by hyenas (60%), elephants (20%), and wild dogs (20%). However, only 7% of incursions resulted in livestock losses, giving the mitigation measures an estimated 93% effectiveness in reducing predation. Only one predator-proof boma was damaged by elephants in Pangaro Village, Mwanga District.

DISCUSSION

Livestock Depredation

Goats were the most frequently preyed livestock, followed by cattle and sheep, likely due to their greater abundance and availability to wild carnivores. This finding is consistent with previous studies (e.g., Woodroffe & Frank, 2005; Bagchi & Mishra, 2006; Mbise et al., 2018; Kissui et al., 2019; Adhola et al., 2022), although it contrasts with Asaye et al. (2024), who reported no significant differences in predation among livestock types.

Wild Animals Involved in Livestock Depredation

Spotted hyenas were responsible for most livestock predation and the greatest economic losses, followed by lions and leopards. Predation differed significantly among carnivore species, indicating that a few large carnivores account for most livestock losses. These findings are consistent with studies conducted across East Africa (e.g., Ogada et al., 2003; Mwakatobe et al., 2013; Mbise et al., 2018; Kissui et al., 2019; Adhola et al., 2022; Wakoli et al., 2023; Mbise, 2024).

Measures Taken by Local Communities for Reducing Livestock Depredation

The most common mitigation measures included livestock enclosures (bomas), herding

dogs, spears and arrows, and the use of solar or battery-powered lights. Similar strategies have been reported in the Serengeti ecosystem and Samburu, while reinforced predator-proof bomas have been shown to effectively reduce livestock losses, particularly from spotted hyenas (Okello et al., 2014; Kissui et al., 2019).

The Season for the livestock predation incidents

Livestock predation was significantly higher during the dry season (76%) than the wet season (24%), likely due to reduced wild prey availability, increased livestock movement into high-risk grazing areas, and greater overlap between livestock and carnivores. This finding is consistent with Mwakatobe et al. (2013) and Mbise et al. (2017), but contrasts with Kissui et al. (2019) and Asaye et al. (2024), who reported higher predation during the wet season.

Location of Livestock predation by Wild Animals

Livestock predation occurred both inside enclosures (51%) and outside during grazing (41%), showing that risk is present in both settings. This suggests that traditional bomas alone are insufficient and should be complemented with improved husbandry and deterrent measures during both confinement and grazing. Similar patterns have been reported in the Maasai Steppe, where spotted hyenas also attacked livestock within kraals (Kissui et al., 2019).

Economic Loss

Livestock predation resulted in substantial economic losses, with cattle accounting for the highest financial impact despite fewer animals being killed, reflecting their higher market value. This finding is consistent with Adhola et al. (2022), who also reported cattle as the greatest source of economic loss, but differs from Gichohi et al. (2014), who found that goats and sheep (shoats) incurred the highest predation costs.

CONCLUSIONS AND RECOMMENDATIONS

Conclusions

Long-term monitoring shows that human–carnivore conflict remains a major conservation and livelihood challenge in the Mkomazi sub-landscape, with livestock predation recorded in all six districts and hotspots in Mkinga, Korogwe, and Lushoto. The results highlight the role of weak livestock protection and grazing practices but indicate that conflict can be substantially reduced through improved husbandry, predator-proof measures, digital monitoring, and stronger community engagement.

Recommendations

To reduce livestock predation and promote human–wildlife coexistence, efforts should focus on expanding predator-proof bomas and solar deterrent lights, strengthening rangeland and grazing management, and enhancing the capacity of Village Game Scouts through digital monitoring tools, transport, and incentives. Continued community awareness on livestock husbandry and coexistence, improved coordination among wildlife management institutions and local governments, and benefit-sharing from conservation and tourism initiatives are also recommended. Finally, sustained monitoring and further research on carnivore ecology, livestock management, and the effectiveness of mitigation interventions are essential for adaptive human–carnivore conflict management.

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CONFRONTING HUMAN-WILDLIFE CONFLICT IN THE MKOMAZI ECOSYSTEM, TANZANIA

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ABSTRACT

The Mkomazi sub-landscape, located in northeastern Tanzania across Lushoto, Korogwe, Mwanga, Same, Mkinga, and Rombo districts, forms part of the greater Mkomazi–Tsavo ecosystem that links Tanzania and Kenya. This ecologically rich area supports keystone species such as elephants, buffaloes, and leopards, yet local communities bordering Mkomazi National Park experience persistent human–wildlife conflicts (HWC). This study analyzed monitoring data collected between 2020 and 2025 by Village Game Scouts and District Game Officers, using both standardized, government-tailored HWC data-collection tools and the Miombo/ODK mobile application. The purpose of analyzing data is to transform raw monitoring records into actionable insights that guide conservation decisions and improve Human-Wildlife Conflict (HWC) management. The results showed that 3198 HWC incidents were recorded, with elephants being responsible for about 99% of crop damage and 96% of human attack incidents. Approximately 26,000 ha of crops, mainly cereals, beans, and tubers, were destroyed, and 48 people were attacked, nearly half fatally. The highest number of incidents occurred in Lushoto, Korogwe, and Mwanga districts, peaking at the end of the rainy season (May), when crops reached maturity. These findings confirm elephants as the main conflict species and highlight the spatial and seasonal nature of HWC in the Mkomazi ecosystem. Strengthening the capacity of wildlife officers, improving rapid response and data-driven management, and promoting unpalatable or less preferred crops could reduce losses and enhance coexistence between communities and wildlife in northern Tanzania.

Keywords: Crop damage, human–wildlife conflict, human attacks, Miombo app

INTRODUCTION

Human–wildlife conflict (HWC), defined as interactions between humans and wildlife that result in negative impacts on either party, is a major conservation and development challenge worldwide. In Africa, rapid human population growth, agricultural expansion, habitat fragmentation, and climate change have intensified interactions between people and wildlife, leading to crop damage, livestock predation, human injuries, and retaliatory killing of wildlife (Redpath et al., 2013; Nyhus, 2016; IUCN, 2020; WWF, 2021). Tanzania is among the most affected countries because of its extensive protected area network and abundant elephant and large carnivore populations. Between 2012 and 2019, HWC

caused over 1,000 human deaths and destroyed more than 41,000 ha of crops (TAWIRI, 2020).

Despite national interventions, including the Human–Wildlife Conflict Management Strategy, the Wildlife Conservation Act, compensation schemes, and Wildlife Management Areas, HWC remains persistent, particularly in communities adjacent to protected areas. The Mkomazi ecosystem, a transboundary landscape linking Mkomazi National Park (Tanzania) and Tsavo National Park (Kenya), experiences frequent elephant crop raiding and occasional carnivore attacks on livestock and people. However, limited spatially explicit information on conflict patterns constrains effective mitigation and management.

This study assessed the extent and spatial distribution of HWC incidents in the Mkomazi sub-landscape, identified the principal wildlife species responsible for damage, and proposed management strategies to reduce future conflicts. The findings provide evidence to support data-driven HWC management, strengthen community participation, and promote sustainable coexistence between people and wildlife in the Mkomazi ecosystem and similar landscapes in Tanzania.

METHODS

Study Area

The Mkomazi sub-landscape (Fig. 1) is in northeastern Tanzania ($4^{\circ}17'58''\text{S}$, $38^{\circ}23'22''\text{E}$) and covers approximately 11,954 km² (1.19 million ha). It forms part of the transboundary Mkomazi–Tsavo ecosystem linking Mkomazi National Park in Tanzania with Tsavo National Park in Kenya. The landscape experiences a semi-arid climate, receiving 400–700 mm of annual rainfall with mean temperatures ranging from 24–30°C (Homewood & Brockington, 1999).

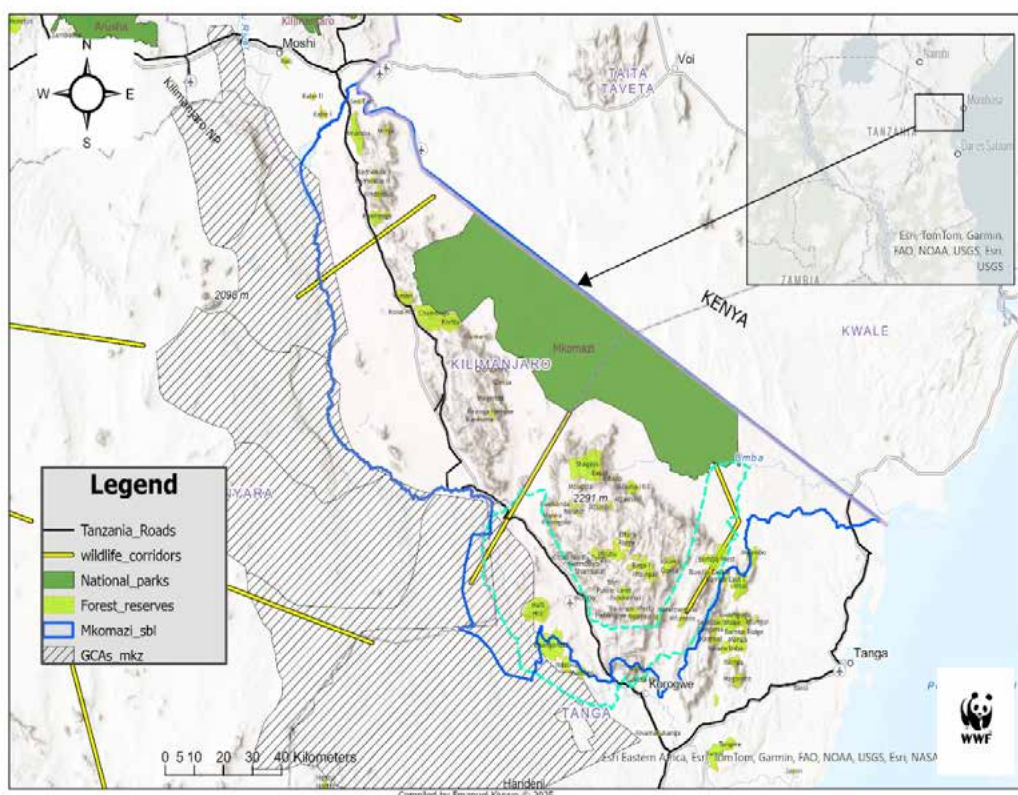


Figure 1. Map of study area (Mkomazi sub landscape)

The landscape comprises Acacia–Commiphora bushland, open savanna, dry woodland, montane forest, and wetlands that support diverse wildlife, including elephants (*Loxodonta africana*), buffaloes (*Syncerus caffer*), giraffes (*Giraffa camelopardalis*), lions (*Panthera leo*),

leopards (*Panthera pardus*), and other large mammals. Mkomazi National Park and the adjoining Uimba Game Controlled Area constitute the principal protected areas and serve as important wildlife corridors linking Tanzania and Kenya (Lovett, 2000).

Approximately 1.2 million people live within the six districts of Rombo, Mwanga, Same, Korogwe, Lushoto, and Mkinga (NBS, 2022). Most communities depend on small-scale crop farming and livestock keeping adjacent to protected areas, making the landscape one of Tanzania's major human–wildlife conflict hotspots. The close interface between wildlife habitats and agricultural land provides an appropriate setting for investigating the spatial and temporal patterns of human–wildlife conflict.

Data Collection Techniques

Human–wildlife conflict (HWC) data were collected between 2020 and 2025 from 69 villages across six districts (Same, Mwanga, Rombo, Korogwe, Mkinga, and Lushoto) within the Mkomazi sub-landscape. Between 2020 and 2023, Village Game Scouts (VGS) and District Game Officers (DGOs) recorded HWC incidents using standardized government reporting forms. All reported incidents were verified by village leaders and DGOs before submission.

From 2024 onwards, data collection was digitized using the Miombo/ODK mobile application installed on smartphones issued to VGS. The application enabled real-time digital recording and transmission of HWC incidents, improving the timeliness, completeness, accuracy, and spatial referencing of data compared with the previous manual paper-based reporting system. Information collected included the date, geographic coordinates, wildlife species involved, type of damage (crop, livestock,

property, or infrastructure), extent of damage, estimated losses, and photographs where applicable. Additional field verification and consultations with local leaders were undertaken to validate reported incidents and improve data quality.

Data Analysis

Verified HWC data were compiled and analysed using Microsoft Excel. Descriptive statistics and additional software such as the Python language and QGIS were used to summarize the frequency and temporal trends of HWC incidents, wildlife species involved, and the extent of crop damage. Graphs and charts were generated to illustrate spatial and temporal patterns of conflict across the study area.

RESULTS

Between 2020 and 2025, a total of 3,198 human–wildlife conflict (HWC) incidents were recorded in the Mkomazi sub-landscape. Elephants were responsible for approximately 98% of crop damage incidents and nearly 48 human attacks, about half of which were fatal. Conflict hotspots were concentrated in Lushoto, Korogwe, Mkinga, Same, and Rombo districts, with incidents peaking towards the end of the rainy season (May), coinciding with crop maturity (Fig. 2). Baseline assessments conducted before implementation of the elephant conservation project identified 124 villages affected by HWC, of which 68, 20, and 36 villages were classified as high-, medium-, and low-conflict areas, respectively. Historical district-level HWC trends established during the baseline informed subsequent monitoring and mitigation interventions.

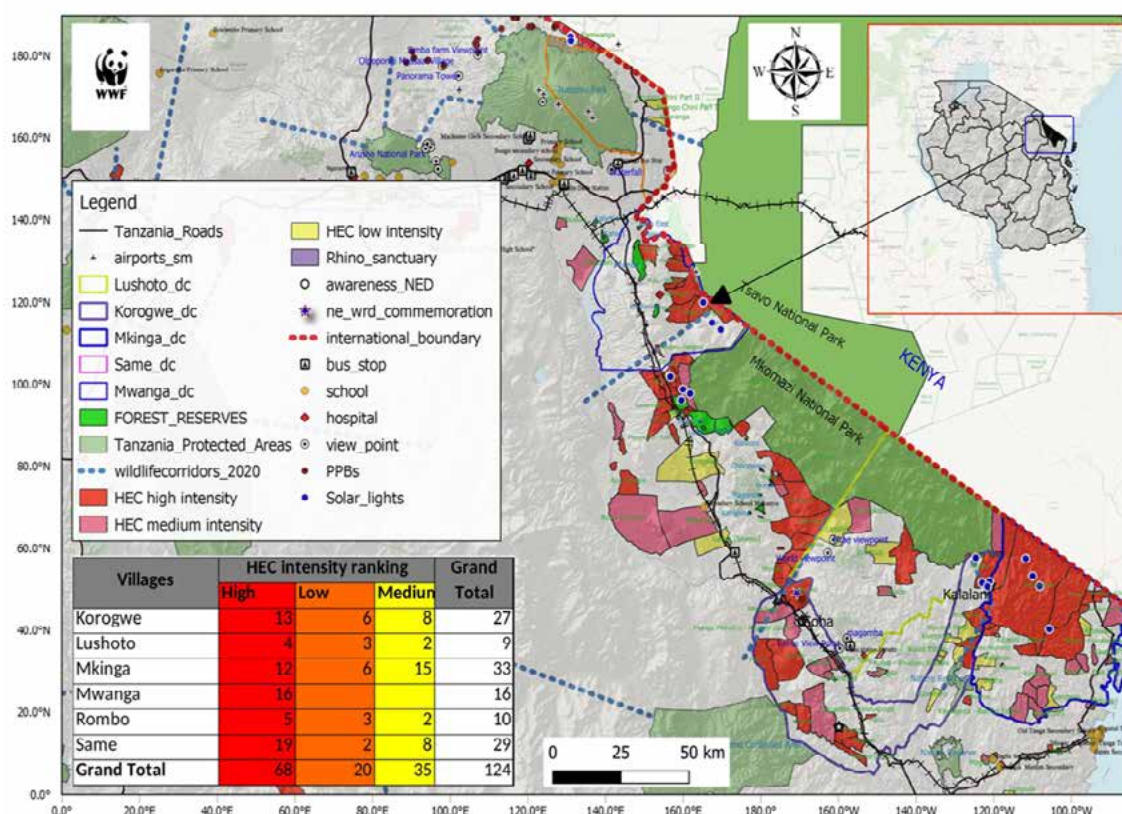


Figure 2. Map of Mkomazi sub-landscape showing HWC hotspot areas

Trend of HWC incidents

Fig. 3 illustrates the temporal trend of human–wildlife conflict (HWC) incidents in the Mkomazi sub-landscape between 2020 and July 2025. Reported incidents increased substantially from 27 in 2020 to 35 in 2021, before rising sharply to 825 in 2022 and peaking at 1,544 incidents in 2023. Thereafter, reported incidents declined to 921 in 2024 and 643 by July 2025.

A total of 3,198 HWC incidents were recorded during the study period, comprising 3,141 crop damage incidents (98.2%), 45 human attacks (1.4%), and 12 infrastructure damage incidents (0.4%). Overall, HWC incidents generally increased across the six districts (Fig.4) over the monitoring period, with crop damage constituting the predominant form of conflict (Figs. 2 and 3).

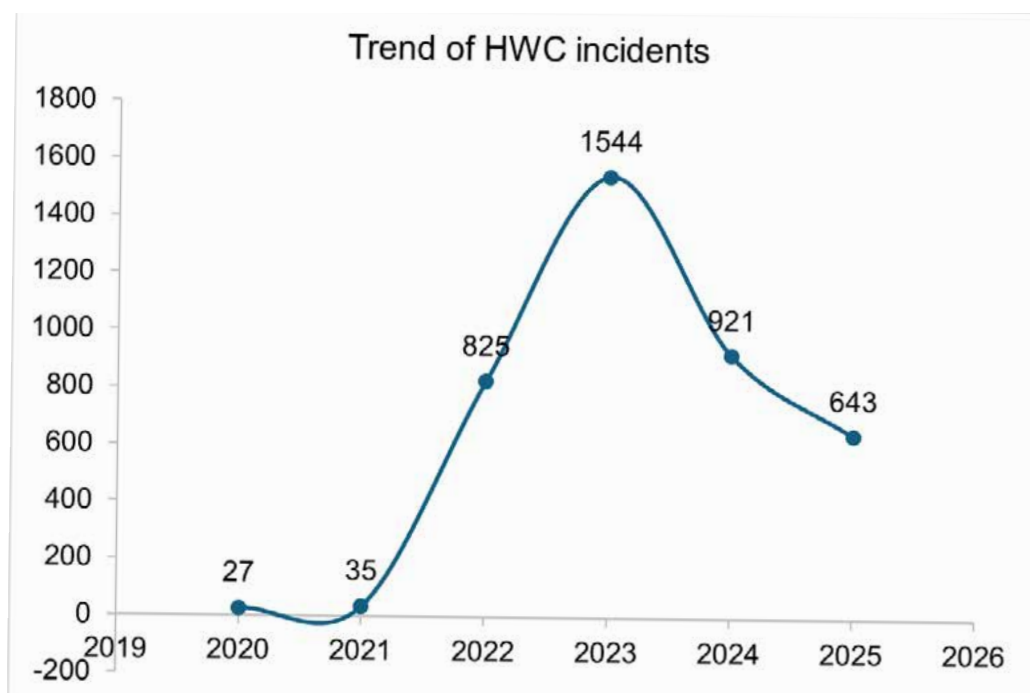
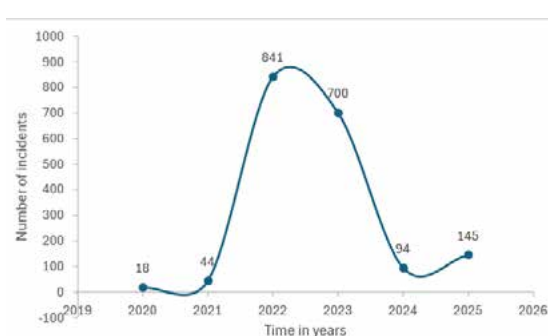
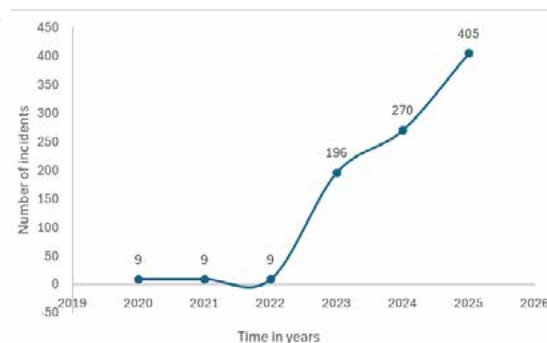


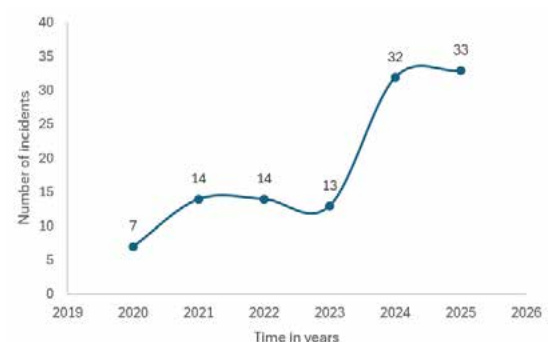
Figure 3. Trends of HWC in the Mkomazi Sub-landscape



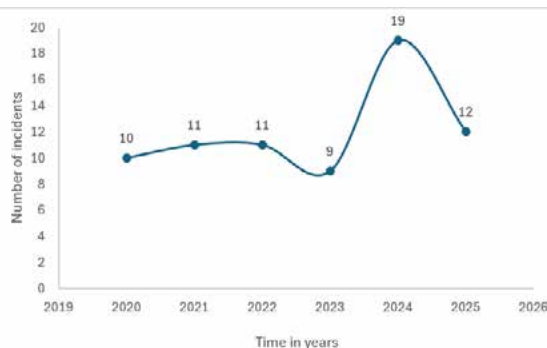
Korogwe



Lushoto



Mkinga



Rombo

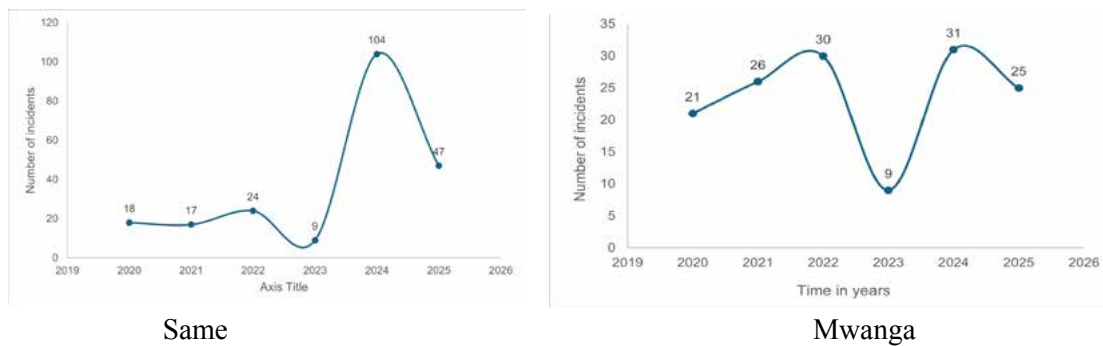


Figure 4. Trends of HWC in the Mkomazi sub-landscape districts

Trends of crop damage (Ha) in Mkomazi

Fig. 5 shows the annual trend in crop damage from 2020 to 2025. A total of 26,126 ha of crops were damaged during the study period. Crop damage rose from 2,935.04 ha in 2020

to a peak of 6,679.45 ha in 2022, then declined to 6,067.70 ha in 2023 and to 1,747.95 ha by July 2025. Overall, the results indicate an initial increase in crop damage, followed by a decline in subsequent years.

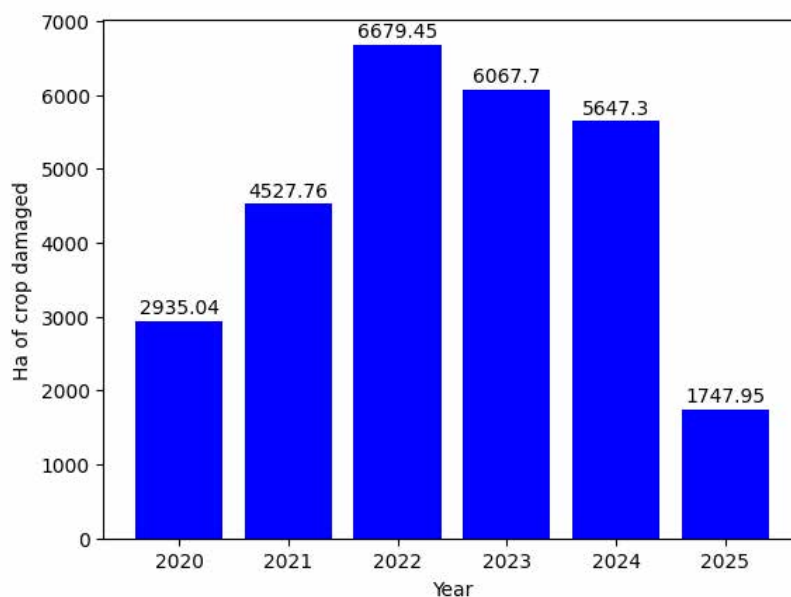


Figure 5. Trends of crops (hectares) damaged in the Mkomazi sub-landscape

Crop damage per district for six years

There has been marked spatial variation in crop damage across the Mkomazi sub-landscape. Korogwe (8,716 ha) and Mwanga (8,496 ha) recorded the largest affected areas, whereas

Lushoto (1,541 ha), Rombo (1,517 ha), and Mkinga (1,512 ha) experienced comparatively lower levels of damage. Same District recorded intermediate losses (4,344 ha), indicating that crop damage was concentrated in a few districts (Fig. 6).

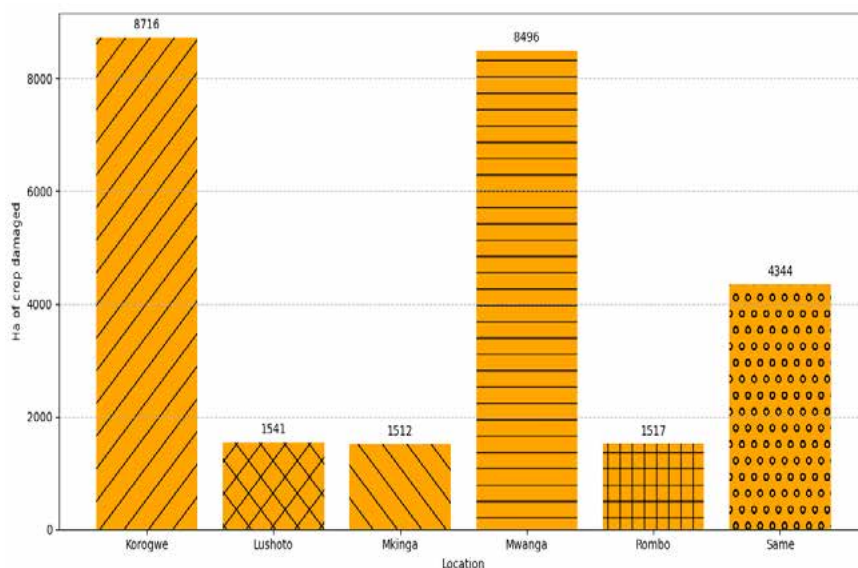


Figure 6. Crop damage (hectares) per district

Types of crops damaged

Fig. 7 shows that cereal crops experienced the greatest wildlife herbivore damage, affecting 13,659 ha, followed by mixed crops (11,694 ha). Beans recorded moderate damage (1,584 ha), while vegetables (378 ha), tubers (741 ha), and fruits (656 ha) were least affected.

The extensive damage to cereals is likely due to their widespread cultivation as staple food crops in Mkomazi and their attractiveness to herbivores, particularly elephants. Lower damage levels in vegetables, tubers, and fruits may reflect their smaller cultivation areas and/or better protection measures

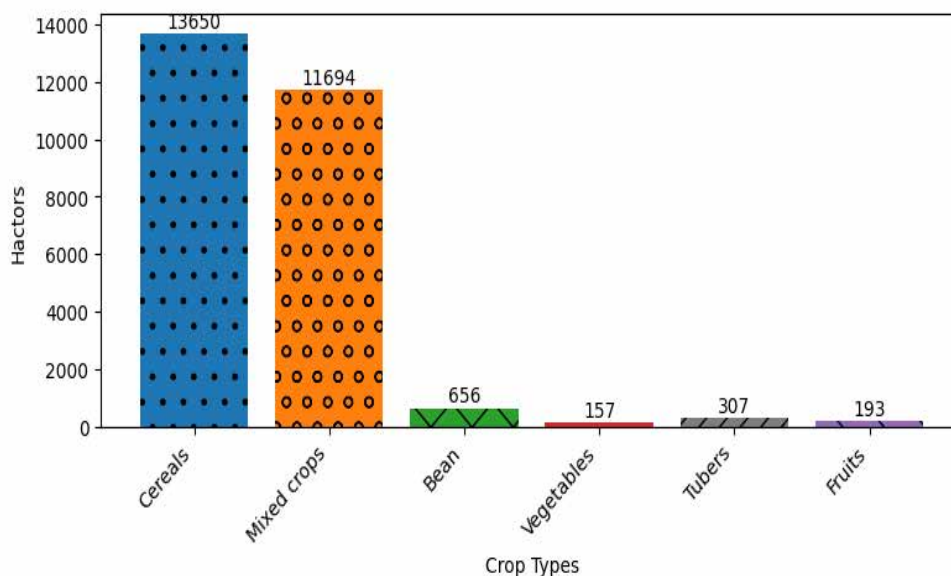


Figure 7. Types of crops (hectares) damaged by wild animals

Wild animals involved in crop damage

Elephants were overwhelmingly responsible for crop damage (Fig.8), accounting for 99% of incidents (n = 3,537). In contrast, other wildlife

species, including Hippos, Buffaloes, Baboons, and others, contributed only 1% of incidents (n = 29). This highlights elephants as the dominant wildlife species involved in crop-raiding and agricultural damage in the study area

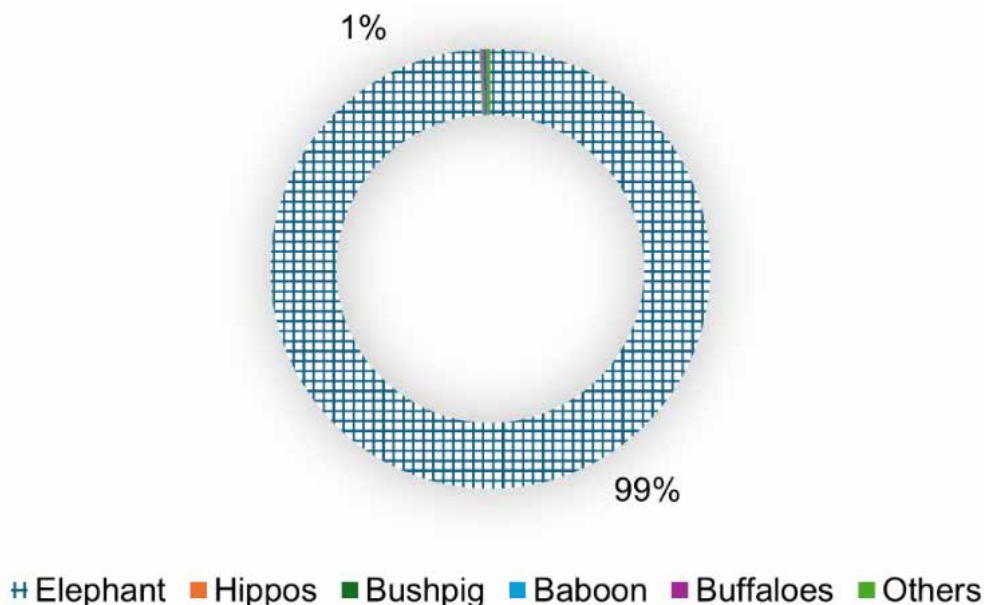


Figure 8. Wild Animals involved in Crop damage

Human attack incidents

Between 2020 and 2025, a total of 48 human attack incidents were recorded across the six districts of the Mkomazi sub-landscape. The number of victims increased markedly from 2 in 2020 to 20 in 2025, indicating a rising trend in human-wildlife conflict impacts. The highest numbers of attacks occurred in 2022 (15 cases) and 2025 (20 cases) (Fig. 9). In 2025, Korogwe (8 cases), Rombo (6 cases), and Lushoto (4 cases) recorded the most incidents. District-level patterns showed peaks in Lushoto during 2023 and 2025, Same and Mwanga in 2022

and again in 2025, while Mkinga reported incidents only between 2020 and 2022. Rombo first recorded attacks in 2024 and experienced a sharp increase in 2025. The elevated number of attacks in 2022 corresponds with the broader human-wildlife conflict trend associated with prolonged drought conditions. However, the reasons for the substantial increase in 2025 remain unclear despite ongoing mitigation interventions. In terms of severity, 54% of incidents resulted in injuries, while 46% resulted in fatalities, underscoring the serious nature of the conflict and the need for strengthened mitigation and community safety measures.

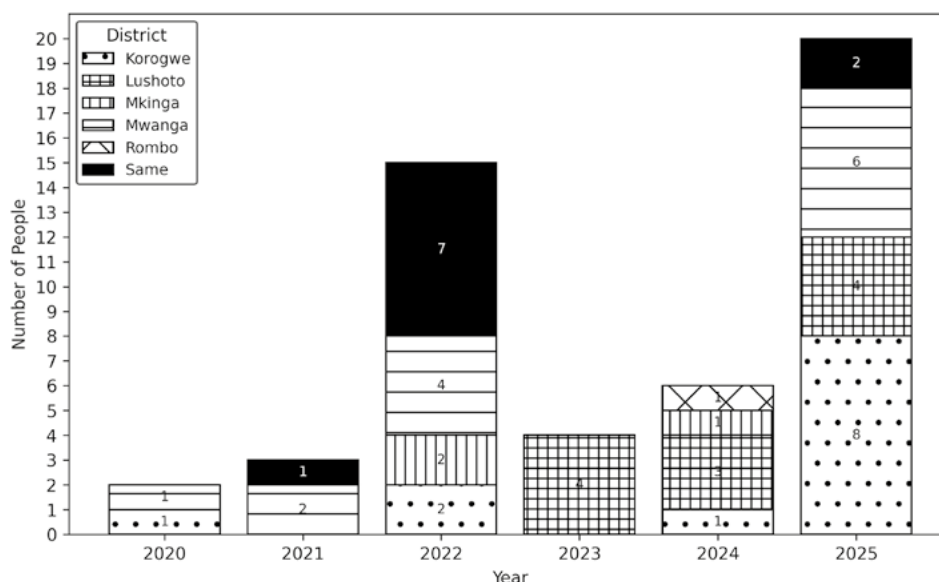


Figure 9. Human attack incidents by wild animals per year in six districts within the Mkomazi sub-landscape

Wildlife species responsible for human attacks

The frequency of human attacks varied significantly among wildlife species (χ^2 (2, N = 48) = 84.13, $p < .001$). Elephants accounted for 96% (n = 46) of all recorded attacks (Fig.10), while leopard and crocodile attacks

were infrequent, each representing 2% (n = 1). Among the 48 documented incidents, 54% (n = 27) resulted in injuries and 46% (n = 21) resulted in fatalities, indicating that elephants are the principal contributor to human injury and mortality associated with human–wildlife conflict in the Mkomazi sub-landscape

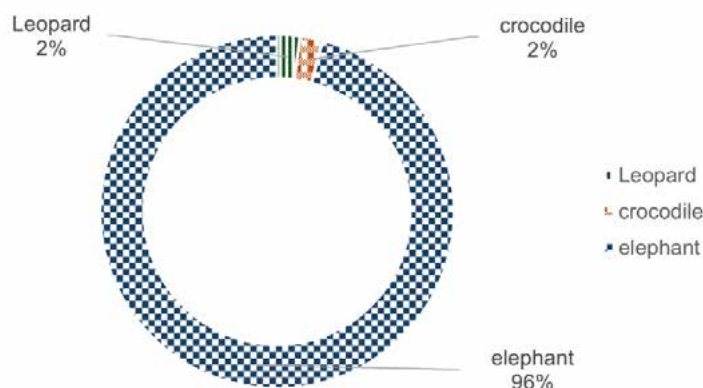


Figure 10. Wildlife involved in human attack incidents

HWC mitigation measures

Government wildlife authorities and conservation partners have implemented a range of human–wildlife conflict mitigation measures in the Mkomazi sub-landscape, including community awareness programmes, rapid response systems, deterrents (e.g., thunder flashes, chilli and beehive fences), and capacity building for Village Game Scouts. WWF complemented these efforts by providing mitigation equipment, mobility support, and technical training, investing approximately USD 200,000 between 2022 and 2026. Collectively, these interventions have strengthened HWC monitoring and response while promoting coexistence between local communities and wildlife

DISCUSSION

The findings confirm that human–wildlife conflict (HWC) remains a major conservation and livelihood challenge in the Mkomazi sub-landscape. Between 2020 and 2025, over 3,000 HWC incidents were recorded, with elephants accounting for more than 98% of crop damage and most human attacks. The extensive loss of cropland and frequent human casualties underscore the significant socioeconomic impacts of HWC on communities neighbouring protected areas.

The increase in reported HWC incidents between 2022 and 2023 is likely attributable to improved community awareness, enhanced reporting through the Miombo/ODK digital platform, and strengthened mitigation efforts. The subsequent decline in incidents and crop damage during 2024–2025 may reflect the effectiveness of these interventions, although the 2025 data cover only the period up to July.

The observed conflict patterns are consistent with previous studies in Tanzania and East Africa, which identify elephant crop raiding as the dominant form of HWC in landscapes adjacent to protected areas and wildlife corridors (Songhurst et al., 2015; King et al.,

2017; Msoffe et al., 2019). The concentration of crop damage in Korogwe, Mwanga, and Same districts likely reflects their proximity to protected areas, key elephant movement corridors, and extensive agricultural activities. Continued investment in digital HWC monitoring, community participation, and locally adapted mitigation measures will be essential for reducing conflict and promoting long-term coexistence between people and wildlife.

Species-specific patterns of conflict

Elephants were the predominant species involved in human–wildlife conflict, consistent with findings from Tanzania and elsewhere in East Africa (Kioko et al., 2006; Hoare, 2015). Their preference for cereal crops, particularly during the late rainy season when crops mature, explains the high incidence of crop raiding and associated livelihood losses (Graham et al., 2009; Chiyo et al., 2012). In contrast, other wildlife species contributed relatively few conflict incidents, although occasional leopard attacks highlight the continued need for targeted carnivore mitigation in livestock-keeping communities (Dickman, 2010; Chang’a et al., 2016). These findings suggest that HWC mitigation efforts and resource allocation in the Mkomazi sub-landscape should primarily focus on reducing human–elephant conflict while maintaining targeted interventions for other conflict-prone species

Spatial distribution of incidents

Human–wildlife conflict showed clear spatial variation across the Mkomazi sub-landscape, with Lushoto, Korogwe, Mkinga, and Mwanga recording the highest incidences of human attacks. This pattern is consistent with studies demonstrating that proximity to protected areas, wildlife corridors, and agricultural land increases conflict risk (Kioko et al., 2006; URT, 2020; Kideghesho et al., 2021). Although Same and Rombo reported fewer incidents, communities in these districts remain vulnerable due to continued wildlife

movements through dispersal corridors. Such spatial differences likely reflect variations in landscape characteristics, human activities, and opportunities for human–wildlife interactions. Previous studies have also highlighted the importance of community participation, conservation awareness, and locally adapted mitigation measures in promoting coexistence between people and wildlife (Mmbaga, 2024).

Implications for conservation and community support

Human–elephant conflict in areas surrounding Mkomazi National Park poses significant risks to both food security and conservation outcomes. Repeated elephant crop raids often trigger retaliatory killings, which threaten elephant population viability and weaken local support for conservation efforts (Treves & Bruskotter, 2014). Strengthening coexistence strategies is therefore critical to protect both community livelihoods and ecological integrity.

Various mitigation measures have been implemented to address HWC. WWF Tanzania’s Elephant Project supported Village Game Scouts in Korogwe and Mkinga with motorcycles and water tanks, improving rapid response to elephant incursions (WWF Tanzania, 2024). Community-based conservation and education initiatives have enhanced awareness and tolerance toward wildlife (Mmbaga, 2024), while electric fencing and buffer zones remain important tools for reducing crop and livestock losses (Runyoro et al., 2019).

Towards effective mitigation and management

Our findings highlight four key priority areas for action. First, interventions should be intensified at the end of the rainy season, when crops are near harvest and raiding peaks, through targeted ranger patrols and community-based deterrents. Second, mobile-based monitoring (Miombo/ODK) has improved reporting accuracy and timeliness,

supporting real-time response and adaptive management. Third, livelihood strategies such as promoting less palatable crops (e.g., chilli, ginger, safflower) offer potential to reduce crop raiding, though they require strong extension support and market linkages (Songhurst et al., 2015; King et al., 2017). Finally, strengthening the capacity of District Game Officers, rangers, and community scouts, alongside improved coordination among TAWA, TFS, and TANAPA, is essential for effective and sustainable HWC management

CONCLUSIONS AND RECOMMENDATIONS

Human–wildlife conflict in the Mkomazi sub-landscape is mainly driven by elephants, with incidents peaking during crop maturity periods. The decline observed since 2022 indicates that ongoing mitigation efforts are effective, highlighting the importance of sustaining community engagement, habitat restoration, and conflict monitoring to strengthen long-term coexistence in northern Tanzania.

These findings underscore the need for integrated, multi-sectoral approaches to HWC management. Priority actions include strengthening rapid response systems through expanded Village Game Scouts with improved mobility and communication, enhancing physical barriers such as electric fencing, predator-proof bomas, buffer zones, and chilli deterrents, and restoring wildlife corridors through improved land-use planning. In addition, improving water security through community tanks can reduce seasonal competition between humans and wildlife. Scaling up community education initiatives like Warriors for Wildlife is essential to sustain coexistence attitudes, while establishing a centralized HWC database will support real-time monitoring, evaluation, and adaptive management

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