MAKERERE UNIVERSITY

WE BUILD FOR THE FUTURE

ASSESSMENT OF THE APPLICATION OF THE MITIGATION HIERARCHY ON A ROAD IMPROVEMENT PROJECT IN WESTERN UGANDA

INGRID ARINGANIZA
2020/HD02/17922U

A DISSERTATION SUBMITTED TO THE DIRECTORATE OF RESEARCH AND GRADUATE TRAINING IN PARTIAL FULFILLMENT OF THE REQUIREMENTS FOR THE DEGREE OF MASTER OF SCIENCE IN ENVIRONMENT AND NATURAL RESOURCES OF MAKERERE UNIVERSITY

DECEMBER 2025

DECLARATION

I, **Ingrid Aringaniza**, declare that this dissertation represents my own original work and has never been submitted to any University or other Tertiary Educational Institution in any form for an academic award. The contribution of my supervisors to this dissertation was consistent with normal supervisory practice.

Signature .... Date.....19th December 2025..........

Ingrid Aringaniza

This dissertation is submitted with our approval as the Academic Supervisors.

Signature .... Date...22nd December 2025...

Assoc. Prof. Gerald Eilu

Department of Forestry, Biodiversity and Tourism

Makerere University

Signature .... Date...19th December 2025...

Dr. Enock Ssekuubwa

Department of Forestry, Biodiversity and Tourism

Makerere University

Digitisation and Self-Archiving Consent Agreement: Theses

Agreement between Makerere University & Students (Authors of Theses / Dissertations / Reports)

1. The author is a student of Makerere University and author of the thesis / dissertation entitled:

ASSESSMENT OF THE APPLICATION OF THE MITIGATION HIERARCHY ON A ROAD IMPROVEMENT PROJECT IN WESTERN UGANDA

2. The author grants to the University:
   a. The right to deposit the electronic version of the Thesis / Dissertation into Makerere University Institutional Repositories (Mak IR) or (Mak UD); and
   b. The right to store the thesis / dissertation in Mak IR / Mak UD and make it permanently available to the general public via the Internet at no cost to the general public after a grace period (if any is specified). Choose one of the two options below:
   c. The Author may opt for immediate open access to the public: OPTED FOR IMMEDIATE OPEN ACCESS
   d. Or Restrict access indefinitely or for the specified number of years : ……………………………………

3. The author warrants that to the best of the authors knowledge and belief:
   a. The thesis / dissertation is an original work;
   b. The author is the owner of all the intellectual property in the thesis / dissertation;
   or
   c. The Author is entitled to deal with the intellectual property in the thesis / dissertation by publishing it on the Internet
   d. The Author has the right, power and authority to enter into this Agreement and to grant the University the rights contained in this Agreement; and
   e. The University's use of the thesis / dissertation pursuant to this Agreement will not infringe the intellectual property rights of any third party.

4. The Author acknowledges and agrees that the University is not responsible or liable for any breach of the intellectual property rights in the thesis / dissertation, in particular any breach of copyright, as a result of the use of the thesis / dissertation pursuant to this Agreement.

5. The University acknowledges that the rights granted by the Creator in clause 2 of this Agreement, do not cause any transfer or assignment of any proprietary rights in the intellectual property in the article to the University.

Signed by the Author as confirmation that the Author has read and accepted the terms of this Agreement:

Name: INGRID ARINGANIZA College/School: CAES Department: DEPARTMENT OF

ENVIRONMENTAL MANAGEMENT

(Tick) Type of Degree: (Undergraduate / PGD / [✓ Masters] / PhD), Reg. No.: 2020/HD02/17922U

Tel No.: 0706557064 E-Mail: aringanizaingrid@gmail.com Signature: ________________ Date:22/12/2025

Supervisor's endorsement:

DEDICATION

To Cleus, my supportive husband, whose unwavering support and encouragement have been my constant source of strength throughout this academic journey; to my beloved children Claus, Manzi, Damian, Amber, and Zion, whose love and inspiration fuel my determination every day; to my parents and siblings, whose sacrifices form the bedrock of my educational journey; and to all the mothers who dare to pursue their dreams, may we continue to inspire one another.

ACKNOWLEDGEMENT

I thank God Almighty for the gift of life, strength, wisdom, and perseverance that have sustained me throughout this academic journey. I express my gratitude to the Wildlife Conservation Society (WCS) for financial and academic support. I am grateful to my supervisors, Assoc. Prof. Gerald Eilu and Dr. Enock Ssekuubwa, for guidance, mentorship, and scholarly input. I also like to acknowledge the lecturers at the Department of Environmental Management for their role in imparting knowledge and fostering a conducive learning environment, that has been critical to my academic growth and the development of this research. I thank Dr. Simon Nampindo, Dr. Grace Nangendo, Tom Igeme, Tobby Abura, Beatrice Kyasiimire, and Paul Hatanga at WCS for their valuable contributions, guidance, and support throughout the research process. I am also grateful to my husband Cleus Bamutura, the family of Mr. Nsiimenta Wilberforce and Dr. Twongirwe Calorine, my entire family and friends, for their unwavering love, understanding, and continuous encouragement, that enabled me to pursue my academic and research goals. I am indebted to Dr. Ritah Kigonya and Dr. Patrick Byakagaba for their help with the research methods, and to Dr. Tonny Kukeera for reviewing my drafts and helping me find useful research papers. Their academic support and friendship have greatly improved the quality of this study.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF ACRONYMS

ANOSIM	Analysis of Similarities
BBOP	Business and Biodiversity Offsets Programme
DBST	Double Bituminous Surface Treatment
EIA	Environmental Impact Assessment
KNP	Kibale National Park
LULC	Land Use and Land Cover
NEMA	National Environment Management Authority
NMDS	Non-metric Multidimensional Scaling
UNRA	Uganda National Roads Authority
UWA	Uganda Wildlife Authority

ABSTRACT

The mitigation hierarchy is vital for preventing and managing environmental impacts of infrastructure development, yet empirical evidence of its effectiveness in safeguarding biodiversity remains limited, particularly in developing countries. This study assessed the application of the mitigation hierarchy following upgrade of the 66 km Fort Portal-Kamwenge road, specifically the 13 km section traversing Kibale National Park in western Uganda. The objectives were to: 1) assess extent of mitigation hierarchy principle application; 2) analyze spatial-temporal land use and land cover changes along the road corridor; and 3) evaluate vegetation composition within the established road corridor. Data were collected using systematic vegetation surveys, multi-temporal LANDSAT satellite image analysis (2015, 2017, 2023), and comprehensive document review following PRISMA guidelines. Stratified random sampling established 36 plots (10m × 10m) at three distance strata (0m, 50m, 100m) from both road sides across six equal segments. ArcGIS was used to perform spatial-temporal LULC analysis within a 500m buffer zone of the road corridor within the national park. The Community Analysis Package was used for the Analysis of Similarity (ANOSIM) and Non-metric Multidimensional Scaling (NMDS). Vegetation community structure and diversity indices were analysed using R software. Environmental and Social Impact Assessment reports were reviewed to evaluate compliance with mitigation hierarchy requirements. Implementation was assessed using indicators derived from international standards (BBOP) and scored on a standardized numerical scale. Results showed differential implementation across mitigation hierarchy stages: complete avoidance and minimization (100% compliance), partial restoration (67%), and absent offset/compensation measures (0%), yielding 62% overall implementation. Land cover analysis revealed initial tree cover loss (-3.6%) during construction (2015-2017), followed by significant recovery (+4.9%) by 2023, with concurrent grassland expansion. Vegetation assessment documented 94 plant species from 45 families. There were no significant differences in species composition or woody stem density relative to road proximity, suggesting minimal road edge effects on forest structure eight years post-construction. These findings suggest that the mitigation hierarchy framework, when properly implemented, may contribute to biodiversity protection as indicated by the forest recovery and limited road edge effects on vegetation structure. We recommend prioritizing complete implementation of all hierarchy sequences, particularly establishing robust offset/compensation mechanisms and comprehensive restoration protocols, to achieve optimal environmental protection outcomes.

CHAPTER ONE

INTRODUCTION

1.1 Background to road development impacts and the mitigation hierarchy

Expansion of road networks in tropical regions is a threat to biodiversity-rich ecosystems, particularly rainforests (Laurance et al., 2014). Transportation corridors fragment habitats, facilitate human encroachment, increase wildlife mortality from vehicle collisions, alter hydrological regimes, and create edge effects that modify microclimate conditions up to several hundred meters into forest interiors (Trombulak & Frissell, 2000; Coffin, 2007). In response to the challenge, mitigation hierarchy has emerged as one of the frameworks for managing biodiversity risks associated with development projects (BBOP, 2012). This sequential approach prioritizes impact avoidance, followed by minimization, restoration, and biodiversity offsetting as a last resort (Bull et al., 2016). Avoidance and minimization of project impacts are believed to be the most effective measures because restoring impacted areas and offsetting impacts are usually costly and have lower chances of being successful (Bull et al., 2022; Larsen et al., 2018).

Mitigation hierarchy has gained global traction, being incorporated into national policies, financial institution guidelines, and international best practices (Arlidge et al., 2018). Countries such as Canada and Australia have integrated the principles into environmental assessment legislation (Canada, 2019; Australia, 2021). Financial institutions, including the World Bank and the International Finance Corporation (IFC) have developed guidelines requiring projects to adhere to the mitigation hierarchy (IFC, 2012). In Africa, the African Development Bank (AfDB) has institutionalized the mitigation hierarchy through its environmental and social safeguards framework (AfDB, 2012, 2013). Uganda has equally incorporated elements of the mitigation hierarchy into its National Environment Act of 2019 and the National Environment (Environmental and Social Assessment) Regulations of 2020, marking a significant shift in the approach to environmental management (GoU, 2019, 2020).

Despite widespread adoption of the mitigation hierarchy, empirical research on its effectiveness, particularly in tropical contexts, remains scarce (zu Ermgassen et al., 2019; Arlidge et al., 2020). This gap in knowledge is especially critical given the rapid infrastructure development occurring in ecologically sensitive areas, including some biodiversity-rich regions (Kisange et al., 2019;

Ojija et al., 2023). For instance, transport infrastructure in Africa is being developed across 30 planned corridors, threatening critical habitats through habitat fragmentation, increased accessibility for exploitation, edge effects, wildlife mortality, and barriers to animal movement (Laurance et al., 2015; Collinson et al., 2019). The application of mitigation hierarchy in road projects in tropical rainforests presents unique challenges due to the relatively high species richness and complex ecological interactions (Gardner et al., 2009). While some mitigation strategies (e.g., route optimization to avoid sensitive areas, timing construction to minimize breeding season impacts, and incorporating wildlife connectivity needs in design) have shown promise, (Laurance et al., 2015; van der Ree et al., 2015), their effectiveness in tropical contexts remains unclear.

This knowledge gap is especially critical for Uganda, where infrastructure development is accelerating within and adjacent to protected areas. Uganda's road network has expanded significantly over the past decade, with several major projects traversing or bordering ecologically sensitive areas including national parks and forest reserves (Uganda National Roads Authority, 2020). Despite the recent integration of mitigation hierarchy principles into Uganda's National Environment Act of 2019, there exists limited empirical evidence on how these provisions are being implemented in practice, what ecological outcomes result from their application, and whether current regulatory frameworks adequately ensure biodiversity conservation alongside infrastructure development. Studies from neighbouring East African countries suggest that environmental impact assessment processes often emphasize compliance documentation over substantive biodiversity protection, with mitigation measures frequently designed on paper but inadequately implemented or monitored in the field (Bidandi & Williams, 2018; Kisange et al., 2019). Whether Uganda's updated legal framework has addressed these implementation challenges, and whether projects approved under this framework achieve measurable conservation outcomes, remains empirically unverified.

Furthermore, infrastructure projects often lead to land use and land cover (LULC) changes, which can serve as indicators of ecosystem pressure and degradation. Satellite-based analysis of LULC provides valuable spatial and temporal insights into how development reshapes landscapes, affecting habitat quality, connectivity, and carbon storage (Hettiarachchi et al., 2023; Thorn et al., 2022). Recent global assessments indicate that land use change has affected almost a third (32%) of the global land area in just six decades (1960-2019), with diverging patterns between the Global

North and South (Winkler et al., 2021). However, comprehensive studies linking mitigation efforts to observable LULC trends are still limited in tropical regions.

Infrastructure development may also alter vegetation composition, especially in edge zones or corridors adjacent to roads. These changes may result from microclimate alterations, invasive species introduction, pollution, or increased human activity (Kleinschroth et al., 2019; Kunert et al., 2020). Tropical species are particularly vulnerable to such infrastructure impacts because many are ecological specialists that "avoid even narrow (<30-m wide) clearings and forest edges," while others face increased mortality from roadkill or hunting pressure near roads (Laurence et al., 2009). Evaluating vegetation patterns is therefore critical to understanding the long-term ecological consequences of road projects and the effectiveness of mitigation measures.

This research aimed to address these critical knowledge gaps by evaluating the application and effectiveness of the mitigation hierarchy in Uganda's Fort Portal-Kamwenge road upgrade project through Kibale National Park, specifically assessing how mitigation measures were implemented, analyzing spatial-temporal land use changes, and evaluating vegetation composition in areas affected by road infrastructure development.

1.2 Problem statement

Rapid expansion of infrastructure in tropical regions threatens biodiversity-rich ecosystems, particularly in developing countries (Laurance et al., 2014). While the Mitigation hierarchy has emerged as a potential solution to balance development with conservation (BBOP, 2012), there remains a critical gap in understanding its practical effectiveness in conserving biodiversity within complex tropical ecosystems (zu Ermgassen et al., 2019; Arlidge et al., 2020). The unique challenges presented by high species richness and intricate ecological interactions in tropical rainforests demand a thorough evaluation of mitigation strategies in real-world scenarios (Gardner et al., 2009). While some approaches have shown promise, comprehensive assessments of the full Mitigation Hierarchy's impact on biodiversity conservation outcomes in tropical contexts are lacking (Sonter et al., 2020).

Despite Uganda's incorporation of mitigation hierarchy principles into the National Environment Act of 2019 and the Environmental and Social Assessment Regulations of 2020 (Government of Uganda, 2019, 2020), there exists no systematic empirical evidence on how these regulatory requirements are being translated into practice in infrastructure projects. While the legal framework mandates sequential application of avoidance, minimization, restoration, and offsetting measures, the extent to which project developers and regulatory authorities actually implement these provisions, the quality and rigor of implementation, and the factors that enable or constrain effective application remain undocumented (Cares & Franco, 2023; Bigard et al., 2021). This implementation gap means that Uganda's updated environmental governance framework operates without feedback mechanisms to identify weaknesses, refine regulatory requirements, or strengthen enforcement (Hickey & Mohan, 2005; Bidandi & Williams, 2018), potentially allowing infrastructure development to proceed with inadequate biodiversity safeguards despite seemingly robust legal provisions.

Without empirical evidence of what works and what fails in specific ecological and institutional context, infrastructure projects may continue to employ ineffective mitigation strategies that satisfy regulatory compliance requirements on paper while failing to achieve conservation outcomes in practice (Phalan et al., 2018; Maron et al., 2016). The implications extend beyond individual projects to cumulative landscape-level impacts, as protected area networks face increasing infrastructure pressure from multiple developments whose combined effects on biodiversity remain unquantified and potentially unsustainable (Plumptre et al., 2007; Hartter & Goldman, 2011). Addressing these gaps requires systematic empirical research that evaluates real-world mitigation implementation, measures actual ecological outcomes, and generates context-specific evidence to inform both policy refinement and practical conservation interventions in tropical infrastructure development.

1.3 Objectives

1.3.1 Main objective

The main objective was to assess how the mitigation hierarchy principle influences biodiversity in forested protected areas, evaluating its effectiveness in balancing infrastructure development with conservation.

1.3.2 Specific objectives

The specific objectives were:

i. To assess the extent to which the mitigation hierarchy principle was applied in the Fortportal-Kamwenge road upgrade.
ii. To analyze the spatial-temporal changes in land use and land cover (2015–2023), following upgrade of the road.
iii. To evaluate vegetation composition and structure adjacent to the road corridor.

1.4 Research questions

The following research questions guided the study:

a) To what extent was the mitigation hierarchy (avoidance, minimization, restoration, and offsetting) applied during the Fort Portal–Kamwenge road upgrade within Kibale National Park?
b) To what extent has land use and land cover (LULC) changed over time along the road corridor before, during, and after road construction?
c) What spatial and temporal patterns of forest cover change can be observed along different sections of the road corridor?
d) How does vegetation species composition and structure vary with proximity to road infrastructure?

1.5 Justification

This study responded to a significant literature gap identified by Sonter et al. (2020) and Arlidge et al. (2020), in terms of the scarcity of empirical studies evaluating the full mitigation hierarchy in tropical contexts. The focus on road infrastructure is particularly relevant given the projected global expansion of road networks, with estimates of 25 million kilometres of new roads by 2050, disproportionately concentrated in tropical regions that harbor the majority of global terrestrial biodiversity (Laurance et al., 2014; Ibisch et al., 2016). In Uganda specifically, road infrastructure development is accelerating rapidly as part of national and regional connectivity agendas. Uganda's road network expanded from approximately 70,000 km in 2000 to over 145,000 km by 2020, with the National Transport Master Plan (2018-2040) projecting an additional 4,000 km of paved roads and substantial upgrading of existing routes over the next two decades (Ministry of Works and

Transport, 2018; Uganda National Roads Authority, 2020). Critically, this expansion increasingly intersects with Uganda's protected area network, which covers approximately 16% of the country's land area and harbors significant biodiversity including globally important populations of mountain gorillas, chimpanzees, and numerous endemic species (Plumptre et al., 2007; NEMA, 2016). At least 12 major road projects traversing or bordering protected areas have been implemented or are planned across Uganda's national parks and wildlife reserves, creating urgent need for evidence-based mitigation strategies (UNRA, 2020; UWA, 2019). The Fort Portal-Kamwenge road represents one of several infrastructure developments linking regional economic centers while passing through critical conservation landscapes in the Albertine Rift, a global biodiversity hotspot recognized for exceptional species richness and endemism (Plumptre et al., 2007; Brooks et al., 2011).

In terms of policy implications, the study aligns with several international conventions and policy frameworks, including Target 3 of the Convention on Biological Diversity's Post-2020 Global Biodiversity Framework and multiple United Nations Sustainable Development Goals, particularly SDG 15 (Life on Land). At the national level, it is in line with Uganda's recent integration of the mitigation hierarchy into its National Environment Act, 2019, providing timely insights for policy implementation and refinement. Furthermore, this research provides practical insights into the implementation and outcomes of the IFC's Performance Standard 6 on Biodiversity Conservation, which mandates the application of the mitigation hierarchy on development projects within protected areas. This study sought to inform more effective policies and practices that reconcile development needs with biodiversity conservation in these ecologically valuable and vulnerable ecosystems.

1.6 Significance of the study

This research directly contributes to achieving environmental sustainability in road construction, a critical challenge as Uganda and other tropical countries pursue infrastructure expansion while attempting to conserve biodiversity. Sustainable road construction requires moving beyond the false dichotomy of "development versus conservation" toward integrated approaches where infrastructure needs are met through designs and practices that minimize ecological harm and compensate for unavoidable impacts (Kiesecker et al., 2010; Laurance et al., 2015).

The study contributes to this sustainability quest by documenting the actual ecological outcomes of a major road upgrade through providing empirical evidence on what "sustainability" looks like in practice versus on paper. This evidence enables more realistic assessment of whether current mitigation approaches achieve no net loss objectives or whether they fall short of genuine sustainability (Bull et al., 2013).

The findings of this study will be useful in three key ways. First, the ecological impact assessment provides quantifiable evidence on how effectively mitigation measures protect biodiversity, allowing for evidence-based improvements to future mitigation strategies. Second, the spatial analysis of land use changes offers a replicable methodology for monitoring infrastructure impacts over time. Third, the vegetation composition analysis provides baseline data for long term ecological monitoring of road impacts in protected areas.

These findings will be used by environmental regulators to improve assessment criteria for future road projects, by conservation organizations to advocate for more effective mitigation measures, and by international financial institutions to evaluate compliance with environmental safeguards. Protected area managers will use the results to develop targeted restoration interventions in degraded roadside zones, while environmental consultants will benefit from the methodological approaches for conducting more comprehensive impact assessments. Additionally, academic researchers can build upon these findings to develop more robust frameworks for evaluating mitigation effectiveness in various ecological contexts.

1.7 Conceptual framework

This study examines how the mitigation hierarchy framework, when applied to road development projects through tropical forests, influences biodiversity conservation outcomes. The conceptual framework presented in Figure 1 synthesizes the relationships between infrastructure development, mitigation interventions, and ecological responses, informed by established ecological theories and empirical observations from similar contexts.

The framework conceptualises road infrastructure development as the primary stimulus threatening biodiversity in tropical forests (independent variable). This conceptualization draws on extensive empirical evidence documenting how roads generate multiple impact pathways including direct habitat loss, landscape fragmentation, edge effects, facilitated human access, and wildlife mortality (Laurance et al., 2009; Trombulak & Frissell, 2000). The spatial pattern of these impacts follows principles derived from Forman's (1995) patch-corridor-matrix model, which predicts that linear infrastructure bisects continuous habitat matrices into discrete patches, increases edge-to-area ratios, and creates corridors of altered environmental conditions extending outward from the disturbance. Island biogeography theory (MacArthur & Wilson, 1967) further informs this conceptualization by predicting that habitat patches created through fragmentation will exhibit declining species richness as patch size decreases and isolation increases, while metapopulation theory (Levins, 1969; Hanski, 1998) emphasizes that connectivity between patches determines long-term population persistence across fragmented landscapes.

In response to these threats, the mitigation hierarchy components (avoidance, minimization, restoration, and offsetting) function as intervention strategies (mediating variables) designed to prevent, reduce, rehabilitate, or compensate for biodiversity impacts (BBOP, 2012; Bull et al., 2016). The framework presupposes a sequential logic where earlier interventions (avoidance, minimization) are more effective and certain than later ones (restoration, offsetting), reflecting the fundamental principle that preventing impacts is preferable to attempting to reverse them after they occur (Phalan et al., 2018). This hierarchical ordering is theoretically justified by ecological principles: maintaining intact large patches (avoidance) better supports area-sensitive species than attempting to restore small degraded patches (restoration), and reducing disturbance intensity (minimization) more reliably maintains ecological processes than trying to recreate them through offsetting (Lindenmayer & Fischer, 2006).

The framework identifies specific biodiversity outcomes as dependent variables measurable through empirical observation: changes in land use and land cover patterns, alterations in forest cover and spatial configuration, and shifts in vegetation species composition and structural attributes. These particular outcome variables were selected based on their theoretical importance and practical measurability. Land use and land cover patterns directly reflect the patch-corridor-matrix model's predictions about landscape configuration changes following infrastructure development (Forman, 1995; Turner et al., 2001). Forest cover changes at varying distances from roads test predictions about edge effect spatial extent derived from tropical forest edge studies (Laurance et al., 2002; Harper et al., 2005). Vegetation composition and structure serve as indicators of community-level responses predicted by island biogeography and metapopulation theories, where fragmentation-sensitive species decline while edge-adapted or generalist species increase (Ewers & Didham, 2006). The inclusion of structural attributes (stem density, tree abundance, canopy connectivity) acknowledges that biodiversity encompasses not only species richness but also functional diversity, ecosystem structure, and ecological processes that collectively determine ecosystem integrity (Díaz et al., 2007).

The framework presupposes several key linkages between variables, each grounded in theoretical expectations and empirical evidence from comparable contexts. First, the framework assumes that road development, if unmitigated, will generate negative biodiversity outcomes through multiple mechanisms. The spatial extent and intensity of these impacts depend on road characteristics (width, traffic volume, surface type) and surrounding ecological context (forest type, species composition, prior disturbance history). This assumption is theoretically justified by edge effect theory, which predicts that forest edges experience altered microclimate, increased light penetration, wind disturbance, and elevated tree mortality that cascade into compositional changes favoring pioneer and edge-adapted species over interior specialists (Chen et al., 1995; Murcia, 1995; Kapos, 1989).

Second, the framework presupposes that mitigation hierarchy interventions can modify these impact pathways, reducing or eliminating negative outcomes. However, the effectiveness of mitigation is not assumed to be uniform or automatic, rather, it depends on implementation quality, ecological appropriateness of measures, and contextual factors that either facilitate or constrain mitigation success. Avoidance measures that maintain large, contiguous forest patches are presupposed to be most effective because they preserve the intact matrix conditions that support diverse species assemblages and maintain ecological connectivity (Fahrig, 2003; Andrén, 1994). Minimization measures that reduce disturbance corridor width and intensity are expected to limit edge penetration distance, thereby protecting greater proportions of interior forest habitat (Laurance et al., 2009). Restoration measures that reestablish canopy connectivity or rehabilitate degraded zones are presupposed to facilitate wildlife movement and enable gradual recovery of forest structure, though full restoration of original species composition and ecological function may require decades and is not guaranteed (Chazdon, 2014; Meli et al., 2017). Offsetting measures are conceptualized as compensatory rather than restorative—they cannot reverse impacts at the project site but can theoretically maintain landscape-level biodiversity by protecting or restoring equivalent habitat elsewhere, though achieving genuine ecological equivalence remains theoretically and practically challenging (Maron et al., 2012; Bull et al., 2013).

Third, the framework explicitly recognizes that mitigation effectiveness is moderated by multiple contextual factors. Policy and governance structures determine whether mitigation is mandatory or voluntary, what standards must be met, and whether enforcement mechanisms ensure compliance (Phalan et al., 2018). Weak governance creates conditions where mitigation commitments remain on paper but are not implemented in practice, while strong regulatory frameworks with adequate enforcement capacity enhance the likelihood that planned measures are actually delivered (Bidandi & Williams, 2018). Stakeholder engagement influences whether affected communities and conservation organizations can meaningfully participate in mitigation planning and hold developers accountable, with more inclusive processes generally producing more comprehensive and sustained mitigation outcomes (Reed, 2008; Bull et al., 2013). Available funding and resources constrain the sophistication and comprehensiveness of mitigation that can be implemented, with budget limitations often resulting in preference for cheaper measures (offsetting on paper) over more expensive interventions (wildlife crossing structures, long-term restoration programs) regardless of their relative ecological effectiveness (Balmford & Whitten, 2003; Kiesecker et al., 2010). Technical capacity determines whether practitioners have the expertise to design effective species-specific mitigation, conduct rigorous baseline assessments, and implement adaptive management based on monitoring results (Ferraro & Pattanayak, 2006; Gullison et al., 2007).

Fourth, the framework presupposes that tropical forest contexts present specific ecological characteristics that may modulate biodiversity responses to both road impacts and mitigation interventions. High species richness and narrow ecological specialization mean that tropical forests contain many species with limited distributions and specific habitat requirements, making them potentially more vulnerable to fragmentation than temperate forests dominated by generalist species with broader tolerances (Gibson et al., 2011; Laurance et al., 2011). Complex trophic interactions including specialized pollination and seed dispersal mutualisms mean that impacts on one species or functional group can cascade through the ecosystem, potentially amplifying fragmentation effects beyond those predicted by simple species-area relationships (Terborgh et al., 2001). Deep edge effect penetration observed in some tropical forests, extending 300-500 meters into forest interiors, means that even roads with narrow disturbance footprints may affect much larger areas than anticipated from temperate-region studies where edge effects typically extend only 50-100 meters (Laurance et al., 2002; Broadbent et al., 2008). Socioeconomic conditions including poverty levels, land tenure insecurity, and livelihood dependencies on forest resources influence whether local communities comply with protected area regulations, engage in resource extraction that compounds road impacts, or participate in restoration activities that could enhance mitigation effectiveness (Adams et al., 2004; Naughton-Treves & Treves, 2005).

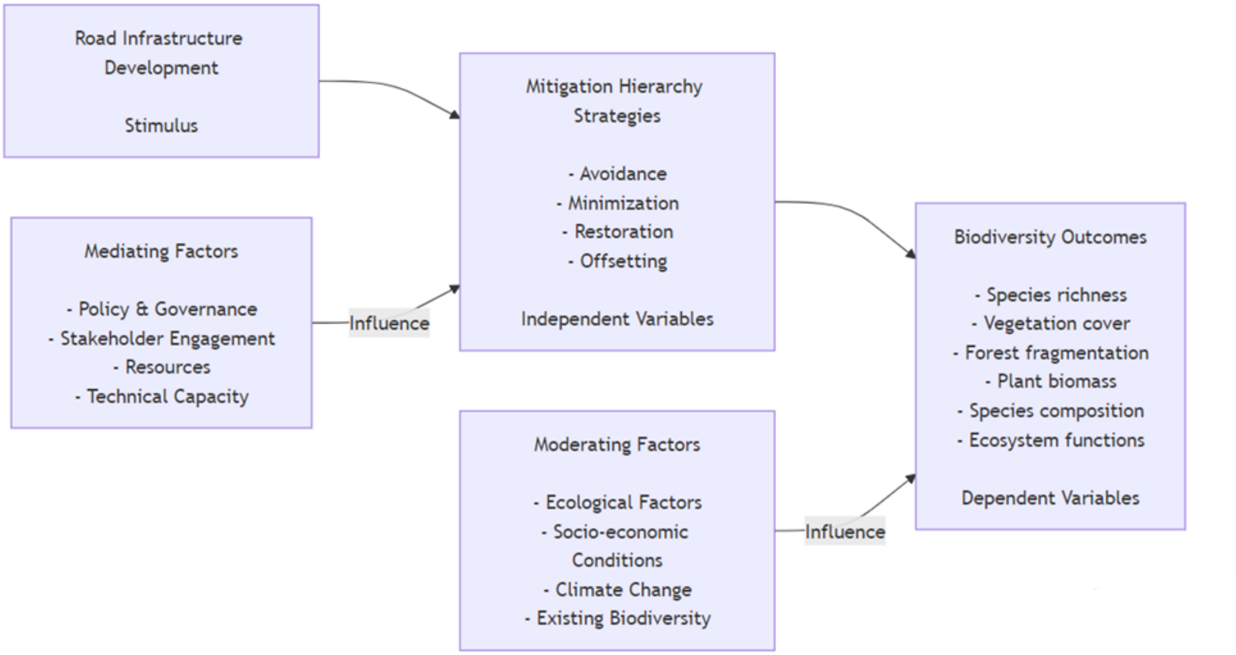


Figure 1: Conceptual framework illustrating the application of the mitigation hierarchy principle on biodiversity in tropical forests.

This conceptual framework guided the study design and analytical approach through specific operationalization decisions. The evaluation of mitigation hierarchy application (research question 1) directly assessed the independent variable by documenting which components were implemented and with what consistency, using established indicators from international standards (BBOP, 2012; NEMA, 2022). The analysis of land use and land cover changes (research question 2) and forest cover patterns (research question 3) measured dependent variables reflecting landscape-scale biodiversity outcomes, employing remote sensing methods to quantify patch configuration, edge density, and forest cover dynamics over time (Hansen et al., 2013; Turner et al., 2001). The assessment of vegetation composition and structure (research question 4) measured dependent variables reflecting community-level and structural responses, using distance gradients from the road edge as proxies for impact intensity and testing for patterns predicted by edge effect theory (Laurance et al., 2009; Harper et al., 2005).

The conceptual framework also informed interpretation of results by providing explicit expectations against which observed patterns could be evaluated. If observed outcomes align with predictions for rigorous full-hierarchy implementation, this would support the framework's presupposition that comprehensive mitigation enables infrastructure development compatible with biodiversity conservation. If observed outcomes instead align with predictions for partial or inadequate implementation, this would indicate that current approaches fail to achieve no net loss objectives and require strengthening in specific components identified through the comparative analysis.

CHAPTER TWO

LITERATURE REVIEW

2.1 The mitigation hierarchy principle

The mitigation hierarchy is a fundamental principle in environmental management, providing a structured approach to addressing potential impacts of development projects on biodiversity and ecosystems. This framework comprises four sequential steps: avoidance, minimization, restoration, and offsetting (BBOP, 2012). Avoidance involves preventing impacts from occurring, often through strategic project planning while minimization aims to reduce the duration, intensity, or extent of impacts that cannot be completely avoided. Restoration seeks to rehabilitate degraded ecosystems following exposure to impacts, and finally, offsetting compensates for any residual, significant adverse impacts that cannot be avoided, minimized, or restored (Bull et al., 2016).

The origin of concept of the mitigation hierarchy can be traced to the 1970s, evolving from environmental impact assessment frameworks (Tinker et al., 2005), and would later gain prominence in the 1990s and 2000s, becoming increasingly integrated into corporate policies, national legislation, and international guidelines (Kiesecker et al., 2010). Recent studies have highlighted both the potential and limitations of this principle. Phalan et al. (2018) emphasized the importance of strengthening the avoidance stage, arguing that it offers the most certain and cost-effective way to reduce impacts on biodiversity. However, Sonter et al. (2020) found that the effectiveness of the mitigation hierarchy in achieving No Net Loss of biodiversity varies significantly depending on local ecological conditions and policy design.

A key controversy in the application of the mitigation hierarchy is the relative emphasis placed on different steps. Critics argue that there is often a disproportionate focus on offsetting at the expense of avoidance and minimization (Apostolopoulou & Adams, 2017). This "compensatory approach" has been criticized for potentially justifying harmful development that could have been avoided or minimized (Spash, 2015). Ethical considerations in offsetting, including questions of ecological equivalence and the commodification of nature, remain subjects of debate (Ives & Bekessy, 2015). The trade-offs between development and conservation inherent in applying the Mitigation Hierarchy raise complex questions about values, rights, and responsibilities in environmental decision-making (Morrison-Saunders & Pope, 2013).

2.2 Application of the mitigation hierarchy in tropical rain forest contexts

Applying the mitigation hierarchy in tropical rainforests presents unique challenges due to these ecosystems' high biodiversity, complex ecological interactions, and often limited baseline data (Gardner et al., 2009). The high endemism in many tropical forests makes avoidance particularly crucial, as impacts on certain species or habitats may be irreplaceable. Tropical rainforests are characterized by their exceptional species richness, complex trophic interactions, and high levels of endemism (Corlett, 2016). These characteristics make them particularly vulnerable to disturbances and complicate efforts to predict and mitigate impacts. Barlow et al. (2016) demonstrated that even low levels of human disturbance can lead to significant biodiversity loss in tropical forests, with their Amazonian forests case study highlighting the importance of stringent avoidance measures.

The application of the mitigation hierarchy in tropical contexts has shown mixed results. In Brazil, for instance, strategic road planning in the Amazon has demonstrated the potential for effective avoidance of high-biodiversity areas (Laurance et al., 2015). The authors proposed a 'retraction and consolidation' strategy, prioritizing the upgrade of existing roads over new construction to minimize forest fragmentation. On the other hand, other studies argue that challenges in implementation and enforcement often undermine the effectiveness of mitigation measures in practice (Fearnside, 2015). Alamgir et al. (2017) found that many road projects in Southeast Asia failed to adequately implement mitigation measures, leading to significant unintended impacts on forests and wildlife.

Regarding restoration efforts, tropical forests present unique challenges due to their ecological complexity. Some innovative approaches show promise, but full ecological restoration often remains elusive. Chazdon and Guariguata (2016) reported that natural regeneration as a restoration tool in the tropics, can be cost-effective, and its success depends on local ecological and socio-economic conditions. This highlights the need for context-specific approaches in tropical forest restoration.

Lastly, biodiversity offsetting in tropical contexts emerges as a particularly controversial aspect of the mitigation hierarchy. Maron et al. (2012) highlighted the challenges of achieving genuine ecological equivalence in highly diverse and often poorly understood tropical ecosystems. Building on this, Bull et al. (2020) found that the effectiveness of biodiversity offsets varies widely depending on local ecological conditions and policy design, with particular challenges in high-biodiversity tropical contexts.

Despite widespread adoption of the mitigation hierarchy in policy frameworks and guidelines, empirical evidence of its effectiveness in conserving biodiversity within complex tropical ecosystems remains scarce (zu Ermgassen et al., 2019; Arlidge et al., 2020). Most applications have been documented in temperate regions, with significantly fewer comprehensive assessments in biodiversity-rich tropical forests where ecological complexity poses unique challenges for mitigation planning and implementation.

2.3 Road infrastructure development and biodiversity

Road infrastructure development poses significant threats to biodiversity through direct and indirect impacts. Direct impacts include habitat loss from road construction and wildlife mortality from vehicle collisions. Indirect impacts, often more pervasive, include habitat fragmentation, changes in species composition, and facilitation of human access to previously undisturbed areas (Trombulak & Frissell, 2000). These impacts are particularly concerning in tropical and subtropical regions with high biodiversity. Laurance et al. (2014) conducted a global analysis of planned road projects, finding that many of the most environmentally damaging roads are slated for construction in these areas. They emphasized the need for strategic planning to reconcile development needs with conservation imperatives.

One of the most significant indirect impacts of roads is habitat fragmentation, which can lead to reduced genetic diversity in populations, altered species interactions, and increased vulnerability to edge effects (Fahrig, 2003). The ecological footprint of roads often extends far beyond their immediate vicinity. While some studies have found edge effects penetrating up to 1-3 km into forest interiors in Amazonian forests (Laurance et al., 2009), significant impacts are often observed within much shorter distances. Broadbent et al. (2008) found that in tropical forests, the most intense edge effects typically occur within the first 100 meters from the forest edge. Similarly, Alignier and Deconchat (2011) observed that the majority of edge influence on plant communities in temperate forests occurred within 100 meters of the edge. However, it's important to note that not all species are negatively affected by roads, and context-specific assessments are crucial for effective mitigation planning. Van der Ree et al. (2015) emphasized this point, arguing for more nuanced approaches to understanding road impacts.

The far-reaching effects of roads are further illustrated by Barber et al. (2014), who used a spatially explicit model to demonstrate that roads have reduced the extent of the world's largest remaining tropical forests by 17%, highlighting the need for landscape-scale approaches to mitigation. Moreover, roads can facilitate human access to previously remote areas, leading to increased deforestation, hunting, and other forms of resource exploitation. Kleinschroth and Healey (2017) found that logging roads in Central African forests remained accessible for an average of 10 years after their construction, prolonging their impacts on wildlife and forest resources.

The application of the mitigation hierarchy to road projects in tropical forests has shown both successes and challenges. Avoidance measures, such as re-routing roads to avoid high-biodiversity areas, have proven effective in some cases. Caro et al. (2014) described how the realignment of a proposed road in Tanzania's Serengeti ecosystem could significantly reduce its impacts on wildlife migrations. This demonstrates the potential of careful planning in mitigating road impacts.

When avoidance is not possible, minimization measures come into play. Wildlife crossing structures, for instance, have shown promise in reducing direct wildlife mortality. However, their effectiveness can vary widely depending on species and local conditions. Goosem et al. (2021) reviewed the effectiveness of canopy bridges for arboreal mammals in tropical forests, finding that while they can be beneficial, their success depends on factors such as bridge design and location. This highlights the need for careful, context-specific design of minimization measures. Restoration of roadside habitats in tropical forests, the final step in the Mitigation Hierarchy before offsetting, presents significant challenges. While some innovative approaches have been developed, such as the use of artificial canopy bridges (Teixeira et al., 2013), full ecological restoration often remains difficult to achieve. The literature demonstrates a notable lack of longitudinal assessments that examine the long-term effectiveness of mitigation measures, particularly in road development contexts. Most evaluations focus on immediate implementation rather than tracking biodiversity outcomes over extended periods (Sonter et al., 2020). This gap is especially significant for understanding the temporal dynamics of edge effects and forest recovery following infrastructure development in sensitive ecosystems. This research addresses this gap through analyzing land use and land cover changes over time (1990-2019), the study offers a longitudinal assessment of biodiversity impacts before and after road development, addressing the need for temporal analyses of mitigation outcomes.

2.4 Effectiveness of the mitigation hierarchy

The effectiveness of mitigation hierarchy components varies widely. Avoidance strategies, such as alternative site selection or design modifications, are considered most effective but are often underutilized due to economic or political constraints (Phalan et al., 2018). Clare et al. (2011) conducted a review of avoidance measures in wetland mitigation and found that they were rarely implemented effectively, often due to a lack of early integration in the planning process. However, when properly applied, avoidance can yield significant conservation benefits. Kiesecker et al. (2010) proposed a "Development by Design" framework that uses systematic conservation planning tools to identify areas where development should be avoided, demonstrating the potential for data-driven approaches to enhance avoidance effectiveness. Laurance et al. (2015) demonstrated how strategic road planning in the Amazon could avoid impacts on high-biodiversity areas while still meeting development objectives.

Minimization techniques, including wildlife crossing structures and environmentally sensitive road design, have shown promise in reducing direct impacts on wildlife. However, their effectiveness can be species-specific and context-dependent (van der Ree et al., 2015). A meta-analysis by Rytwinski et al. (2016) found that wildlife crossing structures were generally effective at reducing wildlife-vehicle collisions, but their effectiveness varied by species and structure type. For example, underpasses were more effective for large mammals, while overpasses were more suitable for a wider range of species.

Restoration approaches in tropical contexts face significant challenges due to the complexity of these ecosystems. While some innovative techniques show potential, full ecological restoration often remains an elusive goal (Chazdon & Guariguata, 2016). Meli et al. (2017) conducted a global meta-analysis of tropical forest restoration outcomes and found that while restoration generally improved biodiversity and ecosystem services, restored forests rarely matched the composition and structure of undisturbed reference ecosystems. In a similar study, Crouzeilles et al. (2016) found that the effectiveness of forest restoration in enhancing biodiversity and vegetation structure varied widely depending on the previous land use, restoration method, and forest type.

Biodiversity offsetting, the last resort in the mitigation hierarchy, remains controversial. While it can provide opportunities for conservation gains, concerns persist about the equivalence of offsets, additionality, and long-term outcomes (Maron et al., 2012). Bull et al. (2013) reviewed biodiversity offsets in practice and found that while they have the potential to achieve No Net Loss of biodiversity, this was rarely demonstrated in practice due to challenges in measurement, equivalence, and long-term management. Sonter et al. (2020) found that the potential for offsets to achieve No Net Loss varied widely across different ecosystems and development contexts, highlighting the need for careful, context-specific offset design.

While individual components of the hierarchy have been studied separately, comprehensive assessments comparing the relative contribution and effectiveness of avoidance, minimization, restoration, and offsetting when implemented together remain limited (Phalan et al., 2018). This fragmented approach has hindered the development of integrated mitigation strategies that effectively leverage the complementary strengths of each component. Furthermore, the literature reveals a lack of standardized methodologies for measuring the effectiveness of mitigation measures, particularly for road projects in protected areas like national parks where conservation value is exceptionally high (van der Ree et al., 2015). This methodological gap hampers comparability across studies and limits the ability to develop evidence-based best practices. This research evaluates all four components of the mitigation hierarchy (avoidance, minimization, restoration, and offsetting) within a single road project, enabling comparative assessment of their implementation and relative effectiveness when applied together. As one of the first comprehensive evaluations of the mitigation hierarchy’s application in Uganda's road sector, this study provides valuable insights for the country's evolving environmental management framework and identifies factors influencing implementation effectiveness in this specific context. Through the assessment of vegetation-based indicators at varying distances from the road, the study develops a methodological approach for quantifying edge effects and evaluating mitigation effectiveness that can be applied in similar contexts. This methodological contribution helps address the need for standardized assessment approaches in road ecology research.

2.5 Land Use and Land Cover Change Detection in Road-Affected Landscapes

Understanding the spatial and temporal dynamics of land use and land cover (LULC) changes associated with road development is critical for assessing infrastructure impacts on tropical forest ecosystems. Remote sensing technologies combined with geographic information systems provide powerful tools for quantifying landscape transformations across multiple temporal and spatial scales, offering insights that ground-based surveys alone cannot achieve (Turner et al., 2003; Nagendra et al., 2013).

Recent global assessments demonstrate that land use change has affected nearly one-third of the global land area over just six decades (1960-2019), with particularly dramatic transformations in tropical regions where road development serves as a primary driver of deforestation and landscape fragmentation (Winkler et al., 2021; Hansen et al., 2013). Satellite-based monitoring has revealed that tropical forests are being lost at accelerating rates, with roads functioning as both direct agents of forest clearing and indirect facilitators of subsequent encroachment by providing access to previously remote areas (Barber et al., 2014; Laurance et al., 2014). The spatial patterns of this loss are not random but highly structured, with forest cover declining predictably as a function of distance from roads following exponential or power-law decay relationships (Chomitz & Gray, 1996; Pfaff, 1999).

Methodologically, LULC change detection employs various approaches depending on data availability, change types of interest, and analytical objectives. Post-classification comparison, which involves independent classification of multi-temporal images followed by pixel-by-pixel comparison, has proven particularly effective for infrastructure impact studies because it generates comprehensive transition matrices documenting specific land cover conversions such as forest to agriculture or forest to grassland (Lu et al., 2004; Mas, 1999). Alternative approaches including image differencing, principal components analysis, and change vector analysis offer advantages for detecting subtle gradual changes but provide less intuitive results for communicating impacts to non-technical audiences (Singh, 1989; Coppin et al., 2004). Recent advances in machine learning classification algorithms, particularly random forest and support vector machines, have improved classification accuracy in heterogeneous tropical landscapes where spectral confusion among vegetation types challenges traditional approaches (Pal, 2005; Rodriguez-Galiano et al., 2012).

Empirical studies across tropical regions have documented consistent patterns of LULC change associated with road development, though the magnitude and spatial extent of impacts vary considerably depending on contextual factors. In the Brazilian Amazon, Barber et al. (2014) demonstrated that paved roads generated deforestation extending far beyond the immediate road corridor, with detectable forest loss occurring up to 50 kilometers from major highways. Their spatially explicit modeling revealed that the Amazon's largest remaining wilderness areas have been reduced by 17% due to road network expansion, with further losses predicted as planned road projects are implemented. Similarly, Soares-Filho et al. (2006) projected that proposed Amazonian road network expansion would eliminate 40% of remaining forest cover by 2050 under business-as-usual scenarios, though these impacts could be substantially reduced through effective spatial planning and enforcement of protected area boundaries.

The temporal dynamics of road-induced LULC change typically follow predictable trajectories. Immediate construction-phase impacts include vegetation clearing for the road footprint itself, temporary work sites, equipment yards, and material extraction areas such as borrow pits (Kleinschroth et al., 2017). These direct impacts are generally limited in spatial extent but create initial landscape fragmentation. Post-construction impacts prove more extensive and persistent, as improved road access facilitates agricultural expansion, logging, hunting, and settlement encroachment along road corridors (Laurance et al., 2009; Arima et al., 2016). Empirical evidence from Southeast Asian tropical forests demonstrates that even unpaved logging roads remain accessible and continue generating forest loss for 5-15 years after their initial construction, extending the temporal window of road impacts well beyond the construction phase (Kleinschroth & Healey, 2017).

However, not all road impacts result in permanent forest loss. Some studies have documented forest recovery in road-affected areas, particularly in protected areas with effective enforcement or in landscapes where economic conditions change to reduce conversion pressures (Rudel et al., 2005; Chazdon, 2014). Distinguishing between temporary construction-phase disturbance followed by recovery versus permanent land use conversion requires multi-temporal analysis spanning sufficient years to detect recovery trajectories (Meli et al., 2017). The limited body of literature on post-construction recovery dynamics in tropical road corridors represents a significant knowledge gap, as most studies focus on documenting initial impacts rather than tracking long-term trajectories (Sonter et al., 2020).

Beyond simple land cover area changes, roads fundamentally alter landscape spatial configuration in ways that affect ecological processes. Landscape ecology provides theoretical frameworks and quantitative metrics for assessing these structural transformations (Forman, 1995; Turner et al., 2001). Roads increase landscape fragmentation by subdividing continuous habitat into smaller patches, increase edge density by creating linear boundaries between forest and non-forest cover types, and decrease connectivity by creating barriers to movement for species unable or unwilling to cross open areas (Fahrig, 2003; Fischer & Lindenmayer, 2007).

Quantitative assessment of these configuration changes employs metrics including mean patch size, patch shape complexity (perimeter-to-area ratios), edge density (total edge length per unit area), and connectivity indices measuring the degree to which landscape structure facilitates or impedes movement among habitat patches (McGarigal & Marks, 1995; Gustafson, 1998). Empirical applications demonstrate that landscape metrics provide sensitive indicators of infrastructure impacts that may not be apparent from simple land cover area calculations. For example, Taubert et al. (2018) found that tropical forest fragmentation reduced patch sizes below thresholds necessary for area-sensitive species even in landscapes where total forest cover remained relatively high, illustrating that spatial configuration matters as much as total habitat area.

Despite substantial progress in remote sensing methods and their application to road impact assessment, significant challenges and knowledge gaps remain. Distinguishing between natural forest dynamics including gap-phase regeneration and infrastructure-induced changes requires careful temporal analysis and understanding of background rates of change (Kennedy et al., 2010). Seasonal phenological variation in tropical forests can create apparent land cover changes that reflect leaf flush or senescence rather than actual land use conversion, necessitating consistent seasonal timing of image acquisition or phenological modeling approaches (Busetto et al., 2008). Cloud cover frequently obscures optical satellite imagery in humid tropical regions, limiting temporal resolution and sometimes preventing acquisition of cloud-free imagery during critical time periods (Asner, 2001).

Most critically, few studies have systematically evaluated LULC changes specifically in the context of mitigation hierarchy implementation. The existing literature documents road impacts extensively but rarely examines whether and how mitigation measures alter the spatial extent, temporal dynamics, or ultimate magnitude of land use change (van der Ree et al., 2015). This gap is particularly acute for tropical protected areas, where roads are increasingly being developed under regulatory frameworks requiring mitigation but empirical assessment of whether these requirements translate into reduced impacts remains scarce (Laurance et al., 2014). Understanding whether avoidance and minimization strategies constrain LULC change to narrower corridors, whether restoration measures enable recovery of disturbed areas, and whether patterns differ between mitigated and unmitigated roads would provide critical evidence for policy refinement and mitigation design.

2.6 Vegetation Responses to Road Infrastructure in Tropical Forests

Roads generate multiple direct and indirect effects on tropical forest vegetation, operating through mechanisms including edge effects, altered disturbance regimes, modified dispersal patterns, and facilitated human exploitation. Understanding these responses requires examining changes in species composition, community structure, functional diversity, and ecological processes across spatial gradients extending from road edges into forest interiors.

The creation of forest edges through road construction fundamentally alters abiotic conditions including light availability, temperature, humidity, and wind exposure (Chen et al., 1995; Kapos, 1989). These microclimatic changes propagate into forest interiors following exponential or sigmoidal decay functions, with impact intensity declining as distance from the edge increases but detectable effects potentially extending hundreds of meters (Laurance et al., 2002; Harper et al., 2005). Empirical measurements in Amazonian forests documented that edge-induced increases in air temperature and vapor pressure deficit can penetrate 100-200 meters into forest interiors, creating desiccation stress that elevates tree mortality rates particularly among canopy emergent and shade-tolerant understory species adapted to humid interior conditions (Kapos, 1989; Briant et al., 2010).

The spatial extent of vegetation responses to these edge effects varies considerably among studies and appears to depend on edge age, surrounding matrix characteristics, and regional climate. In Brazilian Amazonian forests, Laurance et al. (2002) documented elevated tree mortality and altered species composition extending up to 300 meters from forest edges created by roads and clearings, with effects intensifying over time rather than stabilizing. Similarly, Broadbent et al. (2008) found that forest structure changes including reduced basal area and altered size class distributions persisted to 400 meters from edges in some landscapes. However, other studies have reported more limited penetration distances, with most intense effects confined to within 50-100 meters of edges (Harper et al., 2005; Alignier & Deconchat, 2011).

This variation in observed edge effect distances reflects genuine heterogeneity in how different forest types and species assemblages respond to edge creation, as well as methodological differences in what variables are measured and what statistical thresholds define "significant" effects (Ries et al., 2004; Ewers & Didham, 2006). Critically, different ecological variables show edge effects at different spatial scales. Physical microclimate changes may penetrate 100-200 meters, structural attributes including tree density and size distributions may show effects at 50-150 meters, while species compositional shifts toward edge-adapted taxa may require 200-300 meters or more to dissipate (Harper et al., 2005). This multi-scalar nature of edge effects means that single-distance assessments may miss important impacts operating at scales beyond measurement extent or may detect effects for some variables while missing others operating at different scales.

Road-induced edge effects drive predictable changes in forest species composition, with interior specialist species declining in abundance or disappearing from edge zones while pioneer species, generalists, and disturbance-adapted taxa increase (Laurance et al., 2006; Magnago et al., 2015). This compositional turnover reflects differential species responses to altered environmental conditions and competitive dynamics. Shade-tolerant species adapted to stable humid understory conditions experience elevated mortality in desiccation-prone edge zones, while light-demanding pioneer species typically restricted to treefall gaps find suitable establishment conditions in the altered microclimate and increased light availability near edges (Laurance et al., 1998).

Structural changes accompanying compositional shifts include reductions in tree density and basal area near edges due to elevated mortality, particularly of large canopy trees that suffer disproportionate wind damage and desiccation stress (D'Angelo et al., 2004; Laurance et al., 2000). Paradoxically, some studies have reported increased stem density in edge zones due to dense regeneration of pioneer species and lianas exploiting newly available light and nutrients (Laurance et al., 2001). This apparent increase in density should not be interpreted as ecological recovery or mitigation success, as it reflects compensatory growth of disturbance-adapted taxa rather than maintenance of original forest structure and composition (Barlow et al., 2016).

The vertical structure of tropical forests also changes near road edges, with simplified stratification as canopy height decreases, mid-story density increases, and understory light penetration intensifies (Malcolm, 1994). These structural alterations cascade through trophic levels, affecting fauna dependent on specific vegetation strata. Epiphyte diversity and abundance decline near edges as microclimatic conditions become unsuitable for moisture-dependent species, reducing habitat complexity and food resources for arboreal fauna (Laurance et al., 2002; Cardelús et al., 2006).

Beyond taxonomic composition, roads affect functional diversity—the variety of ecological strategies and trait combinations represented in plant communities. Functional traits including wood density, leaf mass per area, maximum height, and seed dispersal mode show systematic variation across edge-to-interior gradients, with trait distributions shifting toward acquisitive "fast-growth" strategies near edges and conservative "stress-tolerant" strategies in interior zones (Laurance et al., 2006; Arroyo-Rodríguez et al., 2017). This functional homogenization toward disturbance-adapted traits may compromise ecosystem resilience to additional stressors and reduce redundancy in trait space that buffers communities against environmental change (Laliberté et al., 2010).

Ecosystem processes including decomposition rates, nutrient cycling, carbon storage, and hydrological regulation also respond to road-induced vegetation changes. Elevated tree mortality near edges releases large pulses of coarse woody debris that temporarily increase carbon stocks but eventually decompose, releasing carbon dioxide and contributing to net forest carbon losses (Laurance et al., 2000). Altered microclimatic conditions accelerate litter decomposition rates near edges, potentially increasing nutrient losses through leaching while simultaneously enhancing nutrient availability for plant uptake in a complex feedback (Didham, 1998). The net effect on ecosystem-level carbon balance and nutrient cycling depends on the relative magnitudes of these competing processes, which remain poorly quantified for most tropical forest road contexts (Pütz et al., 2014).

Roads facilitate biological invasions through multiple mechanisms including transport of propagules on vehicles, creation of disturbed edge habitats suitable for colonization, and altered competitive dynamics favoring invasive over native species (Mortensen et al., 2009; Gelbard & Belnap, 2003). Empirical studies have documented that invasive plant abundance and species richness decline exponentially with distance from roads, with effects detectable at 100-300 meters but most pronounced within 50 meters of road edges (Mortensen et al., 2009; Joly et al., 2011). Once established in road edge zones, invasive species may progressively spread into forest interiors through seed dispersal by wind, animals, or water, creating an expanding zone of invaded habitat that extends well beyond the direct influence of road edge effects (Flory & Clay, 2009).

The interaction between road-induced disturbance and invasive species establishment creates positive feedbacks that can drive ecosystem transformation. Invasive species often possess traits including rapid growth, prolific seed production, and tolerance of disturbed conditions that enable them to dominate edge zones, potentially excluding native species and creating novel communities with altered structure and function (D'Antonio & Vitousek, 1992). In some tropical forest contexts, invasive vines and lianas have proliferated in road edge zones, suppressing tree regeneration and creating persistent arrested succession dominated by disturbance-adapted vegetation fundamentally different from original forest (Laurance et al., 2001; Schnitzer et al., 2012).

The temporal dynamics of vegetation response to road disturbance remain poorly understood due to the scarcity of long-term monitoring studies. Most research examines impacts at a single time point or over short intervals, providing snapshots of current conditions but limited insight into whether observed patterns represent transient disturbance followed by recovery or stable altered states (Chazdon, 2014; Meli et al., 2017). The few longitudinal studies available suggest complex non-linear trajectories where some impacts intensify over time as edge effects penetrate progressively deeper, while other impacts may stabilize or partially reverse if disturbance pressures decline (Laurance et al., 2007).

Recovery potential depends critically on the intensity and spatial extent of initial disturbance, propagule availability from nearby undisturbed forest, presence or absence of ongoing disturbances including fire or continued human exploitation, and whether active restoration interventions are implemented (Chazdon & Guariguata, 2016; Crouzeilles et al., 2016). Natural regeneration in road edge zones may eventually re-establish forest cover, but whether this secondary growth converges compositionally and functionally with undisturbed forest or follows alternative successional trajectories toward novel communities remains context-dependent and difficult to predict (Guariguata & Ostertag, 2001). Evidence from some contexts suggests that edge effects may be partially reversible if surrounding matrix returns to forest cover, reducing the environmental contrast between forest and matrix (Mesquita et al., 1999), while other studies indicate persistent legacy effects where edge-induced changes prove irreversible over decadal timescales (Laurance et al., 2007).

Assessing vegetation responses to roads presents multiple methodological challenges that influence interpretation and comparability across studies. Sampling design decisions including plot size, distance intervals, number of replicates, and taxonomic scope substantially affect detection power and the spatial scale of impacts that can be resolved (van der Ree et al., 2015). Many studies employ relatively small plots suitable for tree surveys but inadequate for detecting landscape-scale patterns or assessing rare species disproportionately affected by fragmentation (Phillips et al., 2003). Distance gradient approaches assume linear or monotonic responses with distance, potentially missing threshold effects or non-monotonic patterns where impacts peak at intermediate distances (Harper et al., 2005).

Most critically, the literature on vegetation responses to roads in tropical forests contains significant geographical and ecological biases. The vast majority of empirical studies come from Neotropical forests, particularly the Brazilian Amazon, with far fewer assessments from African and Asian tropical forests that may exhibit different response patterns due to contrasting evolutionary histories, species pools, and human land use contexts (Gibson et al., 2011). Within studied regions, research concentrates on readily accessible sites near research stations, potentially missing more remote or heavily impacted areas (Laurance et al., 2012). Very few studies have examined vegetation responses specifically in protected areas where legal restrictions may constrain some impact pathways while roads still generate edge effects and facilitate illegal exploitation (Laurance et al., 2014).

The intersection between vegetation assessment and mitigation evaluation represents a particularly acute knowledge gap. While substantial literature documents vegetation impacts of roads, remarkably few studies have explicitly evaluated whether mitigation measures succeed in preventing, reducing, or reversing these impacts (Sonter et al., 2020). Understanding whether avoidance measures that route roads through less sensitive areas reduce vegetation impacts, whether minimization measures including vegetated buffers or reduced corridor widths limit edge effect penetration, whether restoration plantings in disturbed zones accelerate recovery, and whether these interventions produce outcomes distinguishable from unmitigated roads would provide essential evidence for improving mitigation practice. The scarcity of such comparative mitigation-effectiveness studies reflects both the challenges of establishing appropriate counterfactuals (roads without mitigation for comparison) and the limited tradition of post-project monitoring that characterizes infrastructure development globally (Morrison-Saunders et al., 2007).

CHAPTER THREE

STUDY AREA AND METHODS

3.1 Study area

3.1.1 Location and infrastructure

Kibale National Park (Figure 2) is situated in western Uganda, spanning approximately 795 km² between 0°13' to 0°41'N and 30°19' to 30°32'E (Struhsaker, 1997). The park was established in 1993, upgrading its status from a logged Forest Reserve gazetted in 1932 (Chapman et al., 2005). The Fort Portal-Kamwenge road, a critical infrastructure project, traverses 13.3 km of the park's northern section. This 66.2 km road connects Kamwenge Town Council to Fort Portal city, passing through diverse rural landscapes (NEMA, 2011). In response to regional development needs, an upgrade project commenced in 2015 to transform the gravel road into a Class II bitumen standard. The improved road features a 30-meter reserve in rural areas and 20 meters in urban zones. Notably, the project, completed in 2017, maintained the original width of the 13.3 km section through Kibale National Park as a key mitigation measure to minimize biodiversity impacts (NEMA, 2011). This careful consideration of ecological concerns during infrastructure development reflects the ongoing challenge of balancing conservation with economic progress in the region.

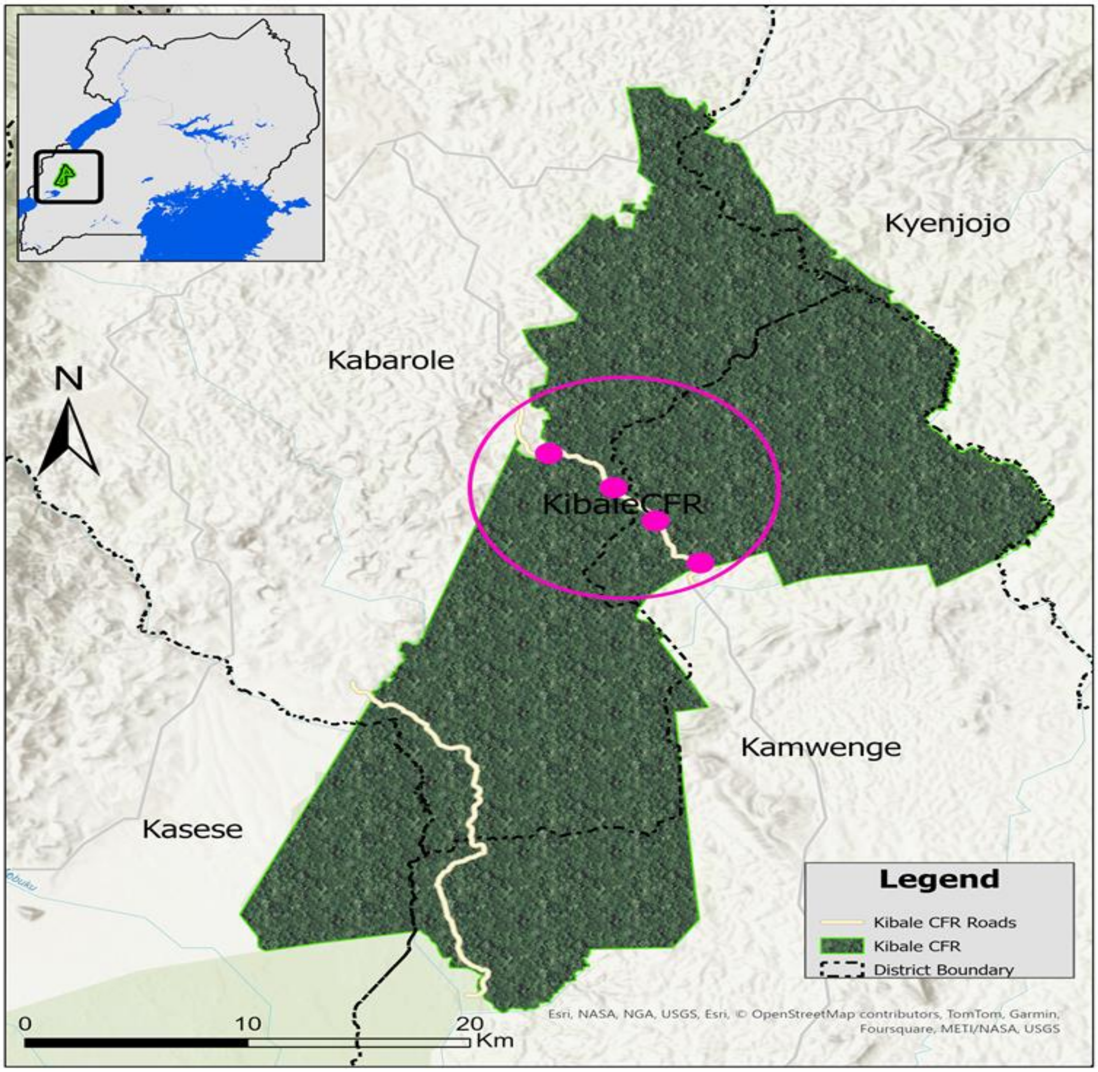


Figure 2: Geographical Location of Kibale National Park (Uganda) showing the study area in the Albertine Rift Region, western Uganda.

3.1.2 Climate and topography

Kibale National Park experiences a tropical climate characterized by bimodal rainfall patterns. Annual precipitation ranges from 1,100 to 1,700 mm, with higher rainfall in the north compared to the south. Peak rainy seasons occur from March to May and September to November (Stampone et al., 2011). Temperatures vary minimally, ranging from 14-15°C to 26-27°C throughout the year.

Complementing its climatic conditions, the park's topography is remarkably diverse. Altitudes range from 1,110 meters above sea level in the south to 1,590 meters in the north. This variation creates an undulating terrain across the Ugandan plateau, contributing significantly to the park's ecological diversity (UWA, 2015). The interplay between climate and topography has shaped Kibale's rich biodiversity, making it a critical conservation area in East Africa.

3.1.3 Flora and fauna

The climatic and topographic diversity of Kibale National Park supports an exceptional array of flora and fauna. The forest is classified as moist evergreen in the north, transitioning to moist semi-deciduous in the south, with a mixed composition in the central region. This variation in forest types hosts over 351 documented tree species, including significant species such as *Chrysophyllum gorungosanum* and the endangered *Milicia excelsa* (Struhsaker, 1997; UWA, 2015). Recent biodiversity assessments have documented the park's continued ecological importance, with the Kibale Biodiversity Assessment Survey (KBAS) confirming high species richness across multiple taxonomic groups and identifying priority conservation areas within the park (Plumptre et al., 2019; Kirkby et al., 2018).

The park's rich vegetation provides habitat for an impressive diversity of wildlife. Kibale hosts 13 primate species, making it one of the most important primate conservation sites in East Africa (Chapman et al., 2005). The park contains the largest population of eastern chimpanzees (*Pan troglodytes schweinfurthii*) in Uganda, with recent surveys by the Wildlife Conservation Society (WCS) estimating a population of approximately 1,450-1,500 individuals distributed across the park (Plumptre et al., 2010; WCS, 2018; Hickey et al., 2019). This population is of global conservation significance given the species' Endangered status on the IUCN Red List (Humle et al., 2016). Other notable primate species include red colobus (*Piliocolobus tephrosceles*), L'Hoest's monkey (*Allochrocebus lhoesti*), red-tailed monkey (*Cercopithecus ascanius*), blue monkey (*Cercopithecus mitis*), grey-cheeked mangabey (*Lophocebus albigena*), black-and-white colobus (*Colobus guereza*), and olive baboon (*Papio anubis*) (Struhsaker, 1997; Chapman & Lambert, 2000).

Beyond primates, the park supports forest elephants (*Loxodonta cyclotis*), estimated at approximately 200-250 individuals based on recent surveys (Plumptre et al., 2019), as well as forest buffalo (*Syncerus caffer*), various duiker species, and significant populations of carnivores including leopard (*Panthera pardus*), golden cat (*Caracal aurata*), and several mongoose and genet species (Struhsaker, 1997). Bird diversity exceeds 375 species, including several Albertine

Rift endemics and species of conservation concern (Pomeroy & Dranzoa, 1997; Plumptre et al., 2019). The park also harbors diverse reptile and amphibian communities, though these remain less comprehensively surveyed than mammals and birds (UWA, 2015).

Of particular relevance to this study, the Fort Portal-Kamwenge road passes through the central region of the park, an area characterized by high biodiversity and the presence of economically significant non-timber forest products, such as wild Robusta coffee (*Coffea canephora*) (Naughton-Treves et al., 1998). The road corridor intersects important chimpanzee ranging areas and primate movement pathways, with WCS surveys documenting regular chimpanzee presence on both sides of the road (WCS, 2018; Hickey et al., 2019). The road's proximity to these valuable ecological resources and critical wildlife habitats underscores the importance of effective mitigation measures to maintain ecological connectivity and reduce infrastructure-related impacts on this globally significant biodiversity.

3.1.4 Human pressures

Despite its protected status and rich biodiversity, Kibale National Park faces anthropogenic pressures that challenge its conservation efforts. The dominant land use types in the surrounding areas include subsistence agriculture, livestock grazing, and settlements. These activities have led to habitat fragmentation and increased human-wildlife conflict along the park's boundaries (Hartter, 2010). The Fort Portal-Kamwenge road, while providing crucial connectivity for local communities, introduces additional complexities to the park's management. The road poses threats to wildlife through habitat fragmentation and increased vehicular traffic. However, recognizing these risks, the road upgrade project implemented specific mitigation measures, including maintaining the original road width within the park boundaries (NEMA, 2011). This approach demonstrates an attempt to balance development needs with conservation imperatives. Beyond the immediate impacts of infrastructure, the park continues to grapple with illegal activities such as poaching and unauthorized resource extraction. These ongoing challenges necessitate sustained conservation efforts and innovative community engagement initiatives to ensure the long-term integrity of the park’s ecosystems (MacKenzie et al., 2017). The complex interplay between human needs and conservation goals in Kibale National Park exemplifies the broader challenges facing protected areas in developing regions.

Uganda offers a compelling case study in this context. While the country recently integrated the mitigation hierarchy into its environmental legislation (GoU, 2019, 2020), the framework had already been applied in practice through international lender requirements. The World Bank-funded Fort Portal-Kamwenge road upgrade project exemplifies this earlier implementation, employing the mitigation hierarchy to protect the biodiversity-rich park (NEMA, 2011). This project presents a unique opportunity to assess the practical effectiveness of the mitigation hierarchy in balancing development needs with biodiversity conservation. By focusing on the Fort Portal-Kamwenge road upgrade project (NEMA, 2011), this study sought to provide empirical evidence on how the application of avoidance, minimization, restoration, and offset strategies influences biodiversity conservation outcomes in an ecologically sensitive area (Laurance et al., 2015; van der Ree et al., 2015).

3.2 Methods

3.2.1 Research design

This study employed a mixed-methods research design combining quantitative spatial analysis with qualitative assessment of policy implementation (Creswell & Plano Clark, 2017). The design integrated three complementary approaches following an embedded mixed-methods framework (Johnson & Onwuegbuzie, 2004). Particularly, document analysis was conducted to evaluate mitigation hierarchy application, remote sensing analysis to quantify land use and land cover changes, and field-based ecological surveys to assess vegetation composition and structure. This triangulated approach enabled both retrospective evaluation of mitigation implementation and prospective assessment of ecological outcomes, providing comprehensive evidence of mitigation effectiveness across multiple scales and indicators (Jick, 1979; Patton, 2002). The integration of quantitative ecological measurements with qualitative policy analysis addresses the multi-dimensional nature of mitigation effectiveness, recognizing that successful conservation outcomes depend not only on ecological responses but also on institutional implementation processes (Ferraro & Pattanayak, 2006; Naidoo et al., 2008).

Evaluating mitigation hierarchy implementation requires understanding both what was intended (documented in policy and planning documents) and what was achieved (measurable ecological outcomes). Qualitative document analysis enables systematic assessment of mitigation commitments, regulatory requirements, and stated intentions, while quantitative ecological measurements provide empirical evidence of actual biodiversity outcomes (Johnson & Onwuegbuzie, 2004). Neither approach alone would adequately address the research questions, qualitative analysis reveals implementation processes and policy gaps, while quantitative analysis demonstrates whether these translate into conservation success or failure.

The complexity of infrastructure-biodiversity interactions demands multiple lines of evidence that can be triangulated to strengthen validity (Jick, 1979; Patton, 2002). Mitigation effectiveness cannot be determined solely from documentation review (which may overstate implementation) or solely from ecological measurements (which cannot explain implementation gaps without policy context). Through integrating both approaches, the study generated more robust conclusions than either method independently could provide. For instance, documented commitments to maintain canopy connectivity gain meaning only when field observations reveal whether these commitments were actually implemented, while observed terrestrial primate crossing behavior gains explanatory power when linked to documented absence of connectivity restoration measures.

Within this mixed-methods framework, the study followed a retrospective quasi-experimental design evaluating outcomes of a "natural experiment" where mitigation hierarchy was applied (treatment) to road development through a protected area, with outcomes compared against theoretical predictions and documented intentions (Shadish et al., 2002). This design was necessary because random assignment of infrastructure projects to different mitigation regimes is neither ethical nor feasible.

To operationalize this quasi-experimental approach, a before-during-after temporal framework was employed to capture the full trajectory of impacts and recovery. The 2015 baseline represents pre-construction conditions, the 2017 assessment captures peak construction-phase impacts, and the 2023 assessment evaluates medium-term post-construction outcomes including any recovery or degradation trajectories. This temporal structure enables detection of transient construction-phase impacts versus persistent long-term changes, a distinction critical for evaluating mitigation effectiveness (Morrison-Saunders et al., 2007).

Spatially, the study focused specifically on the 13-kilometer road section traversing Kibale National Park because this segment represents the direct interface between infrastructure development and protected forest ecosystem. The analytical extent was defined as 500 meters perpendicular from the road edge, consistent with empirical evidence indicating that landscape-level alterations in tropical forests intersected by roads typically occur within this range, although edge effects for certain species may extend further (Laurance et al., 2014; Broadbent et al., 2008).The stratified sampling design (six segments along road length, three distance intervals within each segment, both sides of road) enabled detection of spatial heterogeneity in impacts and assessment of whether edge effects follow predicted distance-decay patterns (Harper et al., 2005).

3.2.2 Review of mitigation hierarchy application documents

Subjective sampling was used during the literature review process. Documents on the Fort Portal-Kamwenge road project were identified and selected through established academic databases and institutional repositories based on predefined inclusion criteria, including relevance to mitigation hierarchy application, road infrastructure, and biodiversity impacts. The targeted documents contained detailed information on mitigation measures, with emphasis on those produced by key stakeholders including Uganda National Roads Authority (UNRA), National Environment Management Authority (NEMA), and Uganda Wildlife Authority (UWA). Field observations helped to verify the implementation of mitigation measures. Permission was obtained from relevant authorities such as Uganda Wildlife Authority to ensure ethical conduct. A standardized evaluation framework based on Business and Biodiversity Offsets Programme indicators (BBOP,2012) was applied to assess the implementation of each mitigation hierarchy component.

3.2.3 Satellite image acquisition and preprocessing

A stratified random sampling approach was employed for training data collection. Following the formula N = Classes (5) × Pixels (500), a total of 2,500 training pixels were sampled across five identified land cover classes. The stratification ensured proportional representation of each land cover type within the 500-meter buffer zone along the 13.3 km road section traversing Kibale National Park (Olofsson et al., 2014; Congalton & Green, 2019).

Temporal sampling was conducted by selecting satellite imagery from three distinct time periods: 2015, 2017, and 2023, representing conditions before, during, and after the road upgrade project respectively. These specific years were selected for several reasons. First, 2015 represents the pre-construction baseline, as the road upgrade commenced in early 2016 according to project documentation. Satellite imagery from 2015 therefore captures landscape conditions immediately before infrastructure disturbance, providing the critical baseline against which subsequent changes can be measured. Earlier years were not selected because they would not capture conditions immediately preceding the intervention, potentially missing important pre-construction trends or changes.

Second, 2017 captures the peak construction phase, enabling assessment of immediate land cover changes associated with construction activities. The 2017 imagery therefore documents the maximum extent of construction-related disturbance including temporary work sites, equipment yards, borrow pits, and vegetation clearing beyond the permanent road footprint. This temporal snapshot enables quantification of total land area affected during construction, including areas where impacts may be temporary if restoration measures are subsequently implemented.

Third, 2023 provides sufficient temporal distance (six years post-completion) to evaluate medium-term impacts and potential recovery or degradation trajectories following project completion. This temporal framework aligns with recommendations for infrastructure impact assessments, which suggest that meaningful ecological changes become detectable within 5-7 years post-disturbance in tropical forest systems (Laurance et al., 2009; Barlow et al., 2016). LANDSAT satellite imagery was acquired for consistent time periods for all the years, to maintain phenological consistency across the temporal series (Zhu, 2017). Cloud-free or minimal cloud cover scenes (less than 10% cloud cover) were prioritized to ensure data quality (Roy et al., 2014). Geographic Information System software was utilized to perform spatial-temporal LULC analysis within the 500-meter buffer zone along the road corridor.

To quantify transitions between land cover classes during the study period, change detection techniques were applied. Specifically, post-classification comparison was employed as the primary change detection technique to quantify land use and land cover transitions between the three time periods (2015-2017, 2017-2023, 2015-2023) (Lu et al., 2004; Singh, 1989). This approach involves independent classification of images from each time period followed by pixel-by-pixel comparison to identify locations where land cover class changed from one category to another (Coppin et al., 2004). Post-classification comparison was selected over alternative change detection methods (e.g., image differencing, principal components analysis, change vector analysis) for several reasons (Jensen, 2016).

Unlike other change detection methods, post-classification comparison generates a comprehensive "from-to" change matrix that specifies not only where change occurred but the specific type of transition (e.g., forest to grassland, grassland to forest, forest to agriculture), enabling analysis of change processes rather than simply detecting change presence (Mas, 1999). Second, this approach is less sensitive to differences in atmospheric conditions, sun angle, and sensor calibration between dates compared to methods that directly compare spectral values, making it more robust for multi-temporal analysis spanning eight years (Coppin et al., 2004). Third, post-classification comparison produces results that are intuitive to interpret and can be readily validated through field verification, supporting communication with non-technical stakeholders including park managers and policymakers (Lu et al., 2004).

The change detection analysis generated transition matrices documenting the area (in hectares) that converted from each land cover class in the earlier time period to each class in the later period. Net change for each class was calculated as gains minus losses, while total change was calculated as the sum of all losses (or gains) to quantify landscape dynamics (Pontius et al., 2004). Spatial patterns of change were visualized through change maps highlighting locations of different transition types, enabling identification of whether changes were concentrated near the road corridor or distributed more broadly across the study area (Hansen et al., 2013).

3.2.4 Vegetation field surveys

A systematic stratified sampling design was implemented for vegetation assessment following established protocols for road ecology studies in tropical forests (Laurance et al., 2009; Coffin, 2007). The 13-kilometer road section through Kibale National Park was divided into six equal segments of approximately 2.17 km each to account for potential spatial variation in forest structure and road impacts along the corridor. Within each segment, square plots measuring 10 m × 10 m for trees, with nested 5 m × 5 m subplots for saplings, 2 m × 2 m for shrubs, and 1 m × 1 m for herbs were established at three specific distances (0 m, 50 m, and 100 m) from both sides of the road edge, resulting in six sampling plots per segment.

The selection of these specific distances was informed by previous research on road edge effects in tropical forests. Studies have documented that the most intense ecological impacts of roads typically occur within the first 100 meters from the road edge, with effects including altered microclimate, increased light penetration, wind disturbance, and elevated mortality of canopy trees (Laurance et al., 2006; Broadbent et al., 2008). While some studies have detected edge effects extending up to 300-500 meters into tropical forests (Laurance et al., 2002), the 0-100 m gradient captures the zone of most pronounced change while optimizing sampling efficiency and statistical power (Harper et al., 2005). The 0 m plots positioned at the immediate road edge represent maximum disturbance intensity, 50 m plots capture the transition zone, and 100 m plots approach interior forest conditions less influenced by road presence (Forman & Deblinger, 2000).

This design yielded a total of 36 main plots (6 segments × 3 distances × 2 sides), with 144 nested subplots for capturing different vegetation strata following hierarchical sampling protocols (Stohlgren et al., 1995; Kent, 2011). Vegetation surveys were conducted in October 2023 during the dry season to facilitate access and ensure consistent sampling conditions across all plots. In each plot, all woody stems with diameter at breast height (DBH) ≥ 10 cm were measured and identified to species level where possible, with unknown specimens collected for later identification using herbarium reference materials and consultation with botanists familiar with Kibale's flora (Katende et al., 1995; Hamilton, 1991).

For each tree, the following parameters were recorded: species identity and DBH measured at 1.3 m height using diameter tape (Condit, 1998). Smaller size classes were documented in nested subplots: saplings (DBH < 10 cm but height > 1 m) in 5 m × 5 m subplots, shrubs in 2 m × 2 m subplots, and herbaceous vegetation including grasses, forbs and ferns in 1 m × 1 m subplots (Gentry, 1988; Phillips et al., 2003). This comprehensive nested sampling approach across five major growth forms (trees, saplings, shrubs, herbaceous plants, and lianas) enabled assessment of both canopy composition and understory vegetation structure, capturing the full vertical stratification characteristic of tropical forest ecosystems (Richards, 1996; Whitmore, 1998).

Strict precautions were implemented to minimize disturbance to the forest during data collection, including staying within established plot boundaries, avoiding destructive sampling methods, and following park regulations regarding movement and behavior within the protected area. All field activities were conducted under a research permit issued by the Uganda Wildlife Authority, ensuring compliance with ethical and legal requirements for research in protected areas (UWA, 2023).

Analysis of Similarities (ANOSIM) and Non-metric Multidimensional Scaling (NMDS) were computed using the Community Analysis Package (CAP) software to assess differences in vegetation community composition across sampling distances.

3.2.5 Analysis of the application of the mitigation hierarchy

To evaluate the application of the mitigation hierarchy framework, a systematic literature review was conducted following the PRISMA guidelines (Moher et al., 2009). Comprehensive searches were performed using academic databases for peer-reviewed literature and institutional repositories for project documentation, including records from the Uganda National Roads Authority (UNRA), National Environment Management Authority (NEMA), and Uganda Wildlife Authority (UWA). Search terms included "biodiversity mitigation hierarchy," "road construction impacts," and "Fort Portal-Kamwenge road mitigation."

Inclusion criteria focused on documents addressing the implementation of the mitigation hierarchy principle along the Fort Portal-Kamwenge road within Kibale National Park. The quality of selected literature was assessed using the Critical Appraisal Skills Programme (CASP) tools (CASP, 2018). This process identified two key documents: The Environmental and Social Impact Assessment (ESIA) report and the Environmental and Social Management Plan (ESMP) for the road project (NEMA, 2011).

The assessment framework was structured in terms of: avoidance, minimization, restoration, and offset/compensation. For each component, specific indicators were derived through systematic review of international best practice guidelines and national policy frameworks. A total of 16 indicators were established: 4 for avoidance, 4 for minimization, 3 for restoration, and 5 for offset/compensation (Table 1). Specifically, the framework draws from three primary sources: (1) The Business and Biodiversity Offsets Programme (BBOP) Standard on Biodiversity Offsets (BBOP, 2012), which provides internationally recognized principles and criteria for applying the mitigation hierarchy across all four components; (2) Uganda's National Guidelines for Biodiversity and Social Offsets (NEMA, 2022), which adapts international standards to Uganda's regulatory context and ecological conditions; and (3) The International Finance Corporation's Performance Standard 6 on Biodiversity Conservation and Sustainable Management of Living Natural Resources (IFC, 2012), which was directly applicable to this project given its financing by the World Bank. These sources were selected because they represent authoritative guidance that was either legally applicable in Uganda at the time of project implementation or represented international best practice standards that informed project requirements. The specific indicators were formulated by identifying measurable criteria within these guidelines that could be assessed through document review and field verification. For instance, the avoidance component indicators reflect BBOP's emphasis on early consideration of alternatives and identification of critical biodiversity areas, while offset indicators align with both BBOP standards and Uganda's national offset guidelines regarding governance structures, monitoring plans, and no net loss targets (Bull et al., 2013; Gardner et al., 2013). This approach ensures that the evaluation framework is both internationally credible and locally relevant, enabling comparison with mitigation hierarchy implementation in other contexts while accounting for Uganda-specific regulatory requirements and ecological conditions (Phalan et al., 2018).

To address the absence of specific mitigation hierarchy guidelines in Uganda's policy framework, these international standards were adapted to the local context through consultation with environmental assessment experts. The adaptation process involved reviewing similar evaluations in comparable ecological settings to ensure contextual relevance while maintaining alignment with international best practices. A standardized scoring system was developed by the researcher to quantitatively assess implementation levels.

The scoring of mitigation hierarchy implementation was conducted by the researcher based on systematic review of project documentation and field verification. Each indicator in Table 1 was evaluated against evidence from the Environmental and Social Impact Assessment (ESIA) and Environmental and Social Management Plan (ESMP) produced by NEMA (2011), supplemented by field observations conducted in October 2023 to verify actual implementation status. A binary scoring system (0 = not implemented, 1 = implemented) was applied to each indicator, with scores assigned based on documented evidence of measure implementation rather than quality or effectiveness. For example, under "Alternative Site Assessment," a score of 1 was assigned if project documentation demonstrated that multiple route alignments were evaluated and compared, while a score of 0 indicated no evidence of alternatives analysis. This approach follows established methodologies for evaluating environmental impact assessment quality and mitigation measure implementation (Tinker et al., 2005; Phylip-Jones & Fischer, 2015). To ensure objectivity and reduce bias, scoring criteria were predefined before document review commenced, and ambiguous cases were discussed with thesis supervisors who have expertise in environmental assessment.

For each component of the mitigation hierarchy, an implementation score was calculated as the percentage of the total possible score for that component. The overall implementation score was derived by calculating the weighted average across all components, with weighting based on the number of indicators in each component (Table 1).

3.2.6 Land Use and Land Cover analysis

LULC dynamics were analyzed using LANDSAT satellite images acquired from the USGS Earth Explorer platform for 2015, 2017, and 2023. This timeframe was selected to capture conditions before, during, and after the road upgrade project. To minimize seasonal variations, images were consistently collected during the same season for each year (Zhu, 2017).

Image pre-processing involved geometric rectification, atmospheric correction using the FLAASH module in ENVI software, and topographic normalization (Vanonckelen et al., 2013). A supervised classification approach was employed using the maximum likelihood algorithm, which has demonstrated high accuracy in similar tropical forest contexts (Rwanga and Ndambuki, 2017).

Training samples were collected using stratified random sampling, with the number of samples determined by the formula:

N (Training Data samples) = Classes (5) * Pixels (500)

This approach ensures adequate representation of each land cover class while minimizing bias (Congalton and Green, 2009). A 500-meter buffer zone along the road was used for LULC analysis, based on previous studies indicating significant landscape alterations within this distance in forest ecosystems intersected by roads (Laurance et al., 2015). Accuracy assessment was conducted using a confusion matrix and Kappa coefficient, with ground-truthing performed through field visits to validate classification results (Foody, 2002).

3.2.7 Vegetation composition assessment

Vegetation composition was assessed in October 2023 using a stratified random sampling design. The study focused on the 13-kilometer section of the Fort Portal-Kamwenge road that traverses Kibale National Park, as this segment represented the critical interface between infrastructure development and protected forest ecosystems (Laurance et al., 2015). This road section was divided into six approximately equal segments of 2.17 km each to account for potential variations in forest structure and composition along the corridor.

Within each segment, sampling plots were established at three distances (0 m, 50 m, and 100 m) from both sides of the road edge, following established protocols for road ecology studies (Avon et al., 2010; Deljouei et al., 2017). This distance gradient was selected to detect potential edge effects while remaining within the zone of highest expected impact. The sampling design yielded a total of 36 plots (6 segments × 3 distances × 2 sides of road).

A nested plot design was employed to capture different vegetation strata as follows:

- 10 m × 10 m plots for trees ≥10 cm DBH
- 5 m × 5 m subplots for saplings and seedlings
- m × 2 m subplots for shrubs
- 1 m × 1 m subplots for herbs and grasses

This hierarchical approach enables efficient sampling of multiple vegetation layers while maintaining statistical independence (Kent, 2011). In each plot, all plant species were identified and recorded using reference literature on flora of Kibale National Park and verified by a local botanist (Holm and Valtonen, 2009).

Geographical coordinates of sampling locations were recorded to facilitate spatial analysis of vegetation patterns in relation to the road corridor. The spatial data enabled assessment of potential correlation between vegetation characteristics and distance from the road, a key indicator of infrastructure impact on forest ecosystems (Coffin, 2007).

Given that eight years had elapsed since the road upgrade began, special emphasis was placed on tree age to account for potential changes over this period. Trees with a diameter of 10 centimeters or more were classified as mature trees, while those ranging from 9 centimeters to 0.5 centimeters in diameter were categorized as seedlings and saplings. Baseline data on forest composition before road upgrade was also gathered through literature review of the Environmental Impact assessment report that was compiled before project upgrade (UNRA, 2011).

3.3 Data analysis

3.3.1 Analysis of the application of the mitigation hierarchy principle

The information extracted from the literature review was organized in a tabular format using Microsoft Excel, encompassing details such as authors' credentials, publication dates, study designs, implemented mitigation strategies, and adherence to Business and Biodiversity Offsets Programme (BBOP) standards. To evaluate the application of the mitigation hierarchy, key mitigation strategies for each stage were identified by referencing indicators developed through a thorough review of the National Biodiversity and Social offset guidelines by the National Environment Management Authority (NEMA) and the International standards on biodiversity offsetting by BBOP (Bull et al., 2013; Knight et al., 2020).

Lack of a standardized metric selection procedure necessitated development of a robust evaluation framework. Drawing from established methodologies in conservation literature (Namaalwa & Byakagaba, 2019; Segan et al., 2014), a numerical scoring system was devised. This system employed a scale of 0 to 1 for each indicator, allowing for a nuanced assessment of mitigation efforts. The indicators were carefully formulated to capture various aspects of biodiversity conservation, including habitat preservation, species protection, and ecosystem functionality (Fahrig & Rytwinski, 2009). To ensure comprehensive evaluation, the scoring system incorporated both quantitative and qualitative aspects of mitigation strategies, reflecting the multifaceted nature of biodiversity conservation in infrastructure development.

The scoring was conducted by the researcher based on systematic analysis of project documentation (Environmental and Social Impact Assessment, Environmental and Social Management Plan, and approval conditions) supplemented by field verification conducted during the October 2023 survey. Each of the 16 indicators across the four mitigation hierarchy components (Table 1) was evaluated independently using a binary scoring system where 0 = not implemented (no documented evidence or field observation of the measure) and 1 = implemented (documented evidence in project materials and/or confirmed through field observation).

The individual indicator scores were then aggregated to compute Total Scores (TS), providing a holistic measure of the implementation of the mitigation hierarchy Principle across all its components (Carreras Gamarra et al., 2018). This aggregation method allowed for a balanced consideration of different mitigation aspects, avoiding overemphasis on any single factor. The resulting TS not only quantified the overall effectiveness of mitigation efforts but also facilitated comparative analysis across different road development projects and geographical contexts. This approach aligns with recent calls in the field for more standardized and comparable assessment methods in biodiversity offset and mitigation practices (zu Ermgassen et al., 2019; Maron et al., 2018).

The analysis considered the temporal and spatial scales of mitigation strategies, recognizing that biodiversity impacts and conservation efforts often extend beyond the immediate project area and duration. This comprehensive approach captured both short-term mitigation actions and long-term conservation outcomes, addressing a critical gap in many existing assessment frameworks (Bigard et al., 2017; Sonter et al., 2020).

Table 1: Indicators utilized in assessing the application of the Mitigation Hierarchy during Fortportal-Kamwenge road upgrade along Kibale National Park

Components in Mitigation hierarchy principle (BBOP, 2012)	Indicator	Explanation	Max score
Avoidance	Identification of Sensitive Areas	Identification and avoidance of ecologically sensitive zones (e.g., protected areas, critical biodiversity zones).	4
	Alternative Site Assessment	Consideration of alternative project sites to avoid sensitive areas.	
	Avoidance of High-Risk Activities	Prevention of activities with severe environmental impacts (e.g., deforestation, habitat fragmentation).	
	Compliance with Zoning Regulations	Adherence to land-use plans and conservation area restrictions.	
Minimization	Alternative alignments	Consideration of alternative project alignments to reduce ecological impact.	4
	Modifications of the design of the road project	Adjustments to design, materials, and operations to reduce environmental impact.	
	Timeline Operations	Scheduling activities to minimize ecological disturbance.	
	Minimization of construction and operations footprint	Waste reduction, noise control, pollution prevention, resource efficiency, and invasive species control.	
Restoration	Restoration and revegetation of temporary worksites	Replanting and restoring temporary and construction-disturbed areas.	3

	Restoration and revegetation of areas/sites disturbed by road-construction activities.	Restoring areas directly impacted by construction.	
	Habitat Connectivity Improvement	Enhancing ecosystem connectivity (e.g., reforestation corridors, wildlife pathways).	
Offsets/ Compensation	Offset financing mechanism	Funding arrangements for offset activities.	5
	Offset governance/ oversight arrangement	Structures for managing and monitoring offset projects.	
	Offset implementation plan	Detailed plan for executing offset activities.	
	Offset monitoring Plan	Plan for tracking the effectiveness of offsets.	
	Provision for No Net Loss	Ensuring offset benefits balance project impacts.	

3.3.2 Land Use and Land Cover Analysis

Data acquisition and preprocessing formed the initial phase of the land cover assessment. LANDSAT satellite images were acquired from the USGS Earth Explorer platform for 2015, 2017, and 2023, capturing conditions before, during, and after the road upgrade project. To minimize seasonal variations in vegetation phenology and land surface reflectance, images were consistently collected during the same season for each year (Zhu, 2017). Image preprocessing involved geometric rectification to correct distortions caused by sensor geometry and terrain effects and ensure accurate spatial alignment (Lillesand & Keifer, 2015), radiometric calibration to maintain consistent pixel values across the temporal series (Richards & Jia, 2006), and median filtering to reduce noise and enhance image quality (Jain, 1989). Visual interpretability was improved through histogram equalization and contrast stretching (Gonzalez & Woods, 2009), optimizing image contrast to reveal subtle features critical for identifying land cover changes. Various band combinations were explored to enhance specific features, particularly for distinguishing forest vegetation types (Jensen, 2016).

For image classification, a supervised approach using the maximum likelihood algorithm was implemented. Training data were collected by superimposing raster images onto high-resolution base maps, with selection guided by predefined thematic classes (Congalton & Green, 2019). Spectral reflectance values served as numerical descriptors for different land cover categories (Richards & Jia, 2006). The classification process assigned each pixel to the most probable land cover class based on spectral similarity, generating thematic maps that enabled quantitative assessment of land cover changes throughout the study period. Classification accuracy was evaluated using multiple complementary methods, including confusion matrix analysis for overall and class-specific accuracies, Kappa statistics to account for chance agreement (Foody, 2002; Narmada, 2021), and field verification to validate the correspondence between classified data and actual ground conditions (Stehman & Czaplewski, 1998; Olofsson et al., 2014).

The spatial analysis framework established a 500-meter buffer zone extending perpendicularly from the road edge into the forest interior, based on evidence that the most pronounced landscape-level alterations in forests intersected by roads typically occur within this range (Hosonuma et al., 2012; Laurance et al., 2014). Land cover composition was quantified within this buffer for each temporal snapshot to detect potential edge-induced changes in forest structure. Additionally, a detailed assessment of tree cover dynamics utilized the European Commission's Joint Research Centre dataset, which documents tropical moist forest changes at 30-meter spatial resolution using Landsat time series data from 1990-2022 (Hansen et al., 2013; Vancutsem et al., 2021). This analysis incorporated transition mapping and annual change records to identify deforestation, degradation, and forest recovery events, enabling quantification of tree cover changes at various road segments adjacent to Kibale National Park.

3.3.3 Vegetation composition analysis

The vegetation composition assessment was designed to quantify the ecological impacts of the Fort Portal-Kamwenge road presence and its subsequent upgrade on Kibale National Park's forest ecosystem. The selection of sampling distances (0 m, 50 m, 100 m) was informed by previous research on road edge effects in tropical forests, which suggest that significant ecological impacts can occur within the first 100 meters from the road edge (Coffin, 2007; Forman & Deblinger, 2000). While some studies have observed effects up to 500 meters into the forest (Laurance et al., 2014), the focus on the 0-100 m range allowed for detailed examination of the most intense ecological gradients. This approach captured the transition from immediate roadside (0 m) through the transition zone (50 m) to the beginning of the forest interior (100 m), while optimizing statistical power through increased replication within this dynamic edge environment.

Given the absence of comprehensive pre-construction baseline data, a space-for-time substitution approach was employed, using distance gradients from the road as proxies for impact intensity (Pickett, 1989; Blois et al., 2013). This methodology assumes that areas far away from the road exhibit conditions that approximate the pre-disturbance state, while areas in proximity demonstrate greater influence from both the original construction and subsequent upgrade activities (Forman & Deblinger, 2000; Coffin, 2007; Fuentes-Montemayor et al., 2020).

Statistical analysis employed multiple complementary techniques to assess vegetation community structure. Analysis of Similarity (ANOSIM) with Bray-Curtis dissimilarity measures was utilized to test for statistically significant differences in species composition across the distance gradients (Clarke, 1993; Faith et al., 1987). This dissimilarity metric was selected for its effectiveness with ecological abundance data and robustness to zero-inflated datasets common in vegetation surveys. Non-Metric Multidimensional Scaling (NMDS) provided ordination-based visualization of community composition patterns (Kruskal, 1964), with stress values calculated to assess the reliability of the dimensional reduction. Both techniques are non-parametric approaches suitable for complex, often non-normally distributed ecological data (Clarke & Warwick, 2001).

Stem density analysis was performed using R statistical software to quantify potential changes in woody plant abundance along the distance gradient. This approach enabled detection of possible edge-associated thinning or densification effects (Laurance et al., 2009). Analysis of variance (ANOVA) was employed to assess statistical significance of observed variations in woody stem density relative to road proximity, allowing for quantitative evaluation of potential edge effects on forest structure.

Comparative visual analysis employed representative photographs rather than fixed-point photography. Pre-upgrade conditions were documented using archival camera pictures taken in 2011 from the environmental impact assessment (NEMA, 2011), while post-upgrade conditions were systematically photographed at multiple locations along the 13-km section in October 2023. This approach was necessitated by the absence of precise location data for archival photographs. To ensure valid comparison, post-upgrade documentation focused on capturing typical road-forest interface characteristics that were consistently observed throughout the study section, rather than attempting to locate exact pre-upgrade photography points.

3.4 Limitations of the study

While this study employed rigorous methods and triangulated multiple data sources to strengthen validity, several limitations merit acknowledgment as they influence the interpretation and generalizability of findings.

3.4.1 Baseline data limitations

A significant methodological limitation of this study was the absence of systematic vegetation surveys conducted before road upgrade commenced. The Environmental Impact Assessment (NEMA, 2011) contained general descriptions of vegetation types and listed characteristic species but did not include plot-based quantitative surveys that would enable direct statistical comparison of pre- and post-construction vegetation composition and structure. This limitation is common in real-world infrastructure assessments where comprehensive baseline data collection is constrained by time and budget (Morrison-Saunders et al., 2007; Cashmore, 2004).

To address this validity challenge, the study employed space-for-time substitution, a widely accepted approach in ecological research where spatial gradients are used as proxies for temporal change (Pickett, 1989; Blois et al., 2013; Damgaard, 2019). Specifically, the study assumes that vegetation at greater distances from the road (e.g., 100 m) approximates pre-disturbance conditions, while vegetation at intermediate distances (50 m) and road edge (0 m) represents progressively greater road influence. This approach is theoretically justified by edge effect theory, which predicts that impact intensity declines with distance from disturbance source following exponential or power-law decay functions (Harper et al., 2005; Ries et al., 2004).

Several lines of evidence support the validity of this space-for-time substitution in this context. Historical aerial photographs and satellite imagery indicate that the 13-km road corridor existed as an unpaved track prior to upgrade, meaning that all sampled locations (0, 50, and 100 m) were within the same continuous forest matrix before 2016 and experienced similar prior land use history (NEMA, 2011).

Nevertheless, the space-for-time approach carries important assumptions that merit acknowledgment. The primary assumption, that spatial patterns reflect temporal trajectories, holds only if sites at different distances were initially similar before road upgrade (Fuentes-Montemayor et al., 2020). While the evidence cited above supports this assumption, it cannot be definitively verified without true pre-construction data. Additionally, eight years post-construction may be insufficient time for vegetation at road edge to reach equilibrium with new conditions, meaning that observed patterns may represent transitional states rather than stable outcomes (Laurance et al., 2007). Despite these caveats, space-for-time substitution represents the most rigorous approach available for assessing road impacts in the absence of planned pre-construction baselines and is widely employed in road ecology research facing similar constraints (van der Ree et al., 2015; Laurance et al., 2009).

The study's validity was further enhanced through triangulation with remote sensing data, which provided actual temporal comparisons (2015 vs. 2023) of forest cover and landscape configuration. While satellite data cannot resolve species-level compositional changes, the concordance between remote sensing evidence of canopy loss and field observations of reduced canopy connectivity strengthens confidence that observed patterns reflect road-related impacts rather than artifacts of sampling design or pre-existing spatial variation (Jick, 1979; Maxwell, 2012).

3.4.2 Temporal limitations

The eight-year observation period (2015-2023) may capture only initial phases of ecological change rather than long-term equilibrium conditions. Research on tropical forest dynamics following disturbance indicates that vegetation succession, edge effect development, and community reorganization can continue for decades or even centuries after initial disturbance (Chazdon, 2014; Guariguata & Ostertag, 2001). The apparent stability in vegetation composition observed across distance gradients may represent a transitional state preceding more substantial changes that will only manifest over longer timeframes. For example, canopy tree mortality induced by edge effects may not become statistically detectable until 10-20 years post-disturbance when initial survivors begin succumbing to cumulative stress (Laurance et al., 2007; D'Angelo et al., 2004). Similarly, forest regeneration in disturbed areas may initially be dominated by fast-growing pioneer species that create an appearance of recovery but will eventually be succeeded by different species assemblages, meaning that observed patterns at eight years post-construction do not necessarily predict long-term trajectories (Guariguata & Ostertag, 2001; Chazdon et al., 2007). Future studies extending observations beyond 15-20 years would provide more definitive evidence of whether current patterns represent stable outcomes or transitional phases.

3.4.3 Spatial limitations

The spatial extent of vegetation sampling (0-100 m from road edge) may not capture the full extent of road influence. While this distance gradient was selected based on literature indicating that most intense edge effects in tropical forests occur within 100 meters (Broadbent et al., 2008; Harper et al., 2005), some studies have documented impacts extending 300-500 meters for certain species or ecological processes (Laurance et al., 2002; Pfeifer et al., 2017). The study design would not detect such deeper-penetrating effects, potentially underestimating total impact zone area. Extending sampling to 200-300 meters would have provided more complete characterization of edge effects but was constrained by field logistics (increased travel time between plots, greater difficulty accessing interior forest locations, limited personnel and time availability). The 500-meter buffer used for remote sensing analysis partially addresses this limitation by capturing landscape-level changes at greater distances, though satellite data cannot resolve species-level compositional shifts.

Additionally, spatial pseudo-replication may affect statistical power and independence assumptions. The six sampling segments along the 13-km road section are not truly independent because they share the same road, potentially experiencing correlated impacts from factors such as vehicle traffic, road maintenance activities, or enforcement patterns that operate at the corridor scale rather than varying among segments (Hurlbert, 1984; Oksanen, 2001). Ideally, a comparative study would include multiple road corridors with varying characteristics (e.g., different traffic volumes, construction methods, mitigation approaches) to enable more robust generalization. However, such replication was beyond the scope of a single case study and would require multi-site research collaboration across multiple protected areas, which presents logistical and financial challenges for graduate-level research.

3.4.4 Taxonomic and ecological limitations

The study focused exclusively on vegetation, omitting direct assessment of faunal responses to road development and mitigation. While vegetation provides a foundation for assessing habitat quality and indirect evidence of wildlife impacts (e.g., canopy loss affecting arboreal species), many important biodiversity components including mammals, birds, reptiles, amphibians, and invertebrates may respond to roads in ways not fully captured by vegetation surveys (Laurance et al., 2009; Benítez-López et al., 2010). For example, the study documented terrestrial primate crossing behavior opportunistically but did not systematically quantify wildlife mortality rates, crossing frequencies, or population-level impacts such as genetic isolation or demographic decline. Comprehensive biodiversity assessment would integrate faunal surveys including camera trapping, acoustic monitoring, roadkill surveys, and genetic analysis to complement vegetation data (van der Ree et al., 2015). Such multi-taxa approaches were beyond the scope and resources of this study but represent important directions for future research.

Vegetation surveys also captured only a snapshot of plant diversity during a single season field survey (October 2023), potentially missing species with ephemeral or seasonal occurrence patterns. Some herbaceous species, geophytes, or seasonally deciduous plants may be inconspicuous or dormant during the dry season, leading to underestimation of total species richness (Phillips et al., 2003). Multiple surveys across seasons would provide more complete species inventories, though the focus on woody vegetation (which shows less seasonal variation than herbaceous layers) partially mitigates this limitation.

3.4.5 Methodological limitations

Remote sensing analysis using 30-meter resolution Landsat imagery cannot detect fine-scale changes such as individual tree mortality, small canopy gaps, or subtle compositional shifts that occur below the pixel size (Hansen et al., 2013; Nagendra et al., 2013). The supervised classification approach, while suitable for distinguishing major land cover categories (forest, grassland, agriculture), may misclassify mixed pixels at boundaries between categories or fail to distinguish forest types that appear spectrally similar but differ in composition or structure. Higher resolution imagery from sensors such as Sentinel-2 (10-meter resolution) or commercial satellites (sub-meter resolution) would enable detection of finer-scale patterns but was unavailable for all study time periods or exceeded budget constraints. LiDAR data would provide superior information on forest structure and canopy connectivity but is not routinely collected for tropical African regions (Asner et al., 2015).

The mitigation hierarchy evaluation too relied primarily on document analysis supplemented by field verification, rather than comprehensive monitoring data systematically collected throughout project implementation. The assessment of whether mitigation measures were "implemented" versus "not implemented" was necessarily categorical (binary scoring) rather than quantitative assessment of implementation quality, adequacy, or effectiveness. For example, a restoration measure might receive a score of "1" (implemented) based on evidence that revegetation occurred, but the scoring system does not capture whether revegetation used appropriate native species, achieved targeted survival rates, or successfully reestablished ecological function. More nuanced scoring systems exist (e.g., multi-level scales assessing implementation quality) but require detailed monitoring data that was not available in project documentation (Tinker et al., 2005; Morrison-Saunders et al., 2007).

3.4.6 Generalizability limitations

Findings from this single case study of one road upgrade project through one protected area may not generalize to other infrastructure contexts, protected areas, or geographic regions (Yin, 2014). The Fort Portal-Kamwenge road has specific characteristics including its history as an existing unpaved route being upgraded (rather than new construction through pristine forest), location within a well-managed national park with active ranger presence, passage through mid-elevation forest with particular species assemblages, and implementation under specific regulatory frameworks and institutional capacities that may differ from other projects. Patterns observed here, such as substantial forest recovery, limited edge effects within 100 meters, or partial mitigation hierarchy implementation, should be tested across multiple road projects varying in characteristics before drawing general conclusions about mitigation effectiveness in tropical forest contexts (Laurance et al., 2015; van der Ree et al., 2015).

The study cannot definitively establish causation between specific mitigation measures and observed ecological outcomes due to the observational (non-experimental) design too. While the study documents associations (e.g., forest recovery concurrent with implementation of avoidance and minimization measures), alternative explanations such as natural forest resilience, rainfall variability, or management interventions unrelated to the road project (e.g., enhanced anti-poaching patrols) cannot be completely excluded (Ferraro & Pattanayak, 2006; Shadish et al., 2002). Experimental designs with true controls (areas matched in characteristics but without road development) or before-after-control-impact (BACI) designs comparing road corridor areas with distant control areas would strengthen causal inference, but such designs are rarely feasible for real-world infrastructure projects where site selection is determined by transportation needs rather than research considerations (Underwood, 1994).

3.4.7 Data access and verification limitations

Access to comprehensive project implementation records was incomplete. While Environmental Impact Assessment reports and Environmental Management Plans were publicly available through NEMA, detailed implementation records such as construction supervision reports, contractor compliance documentation, monthly progress reports, or monitoring data collected during construction were not accessible through public channels and requests to Uganda National Roads Authority received limited response. This limited the study's ability to verify precisely when and how mitigation measures were implemented, what challenges arose during implementation, or what adaptive management adjustments occurred in response to field conditions differing from planning assumptions. Greater transparency and systematic archiving of project implementation documentation would enhance future evaluations of mitigation effectiveness (Arts et al., 2012; Morrison-Saunders & Pope, 2013).

3.4.8 Acknowledgment of limitations in interpretation

These limitations do not invalidate the study's findings but rather contextualize their interpretation and define boundaries of appropriate generalization. The triangulation of multiple methods (document analysis, remote sensing, field surveys) partially compensates for individual method limitations by providing convergent evidence from independent data sources (Jick, 1979; Maxwell, 2012). The findings provide robust evidence regarding patterns observable at the spatial and temporal scales investigated, while acknowledging uncertainty about longer-term trajectories, fine-scale processes, and transferability to substantially different contexts. Future research addressing these limitations through long-term monitoring, expanded spatial extent, multi-site replication, and integration of faunal assessments would provide more complete understanding of road impacts and mitigation effectiveness in tropical forest protected areas.

CHAPTER FOUR

RESULTS

4.1 Application of the biodiversity mitigation hierarchy

4.1.1 Implementation of mitigation hierarchy components

Evaluation of the Fort Portal-Kamwenge road upgrade project revealed variable implementation across the four components of the biodiversity mitigation hierarchy, with an overall implementation score of 62% (Table 2), indicating that the mitigation hierarchy was implemented incompletely, with substantial gaps particularly in later-stage components.

Table 2: Implementation scores for provision of biodiversity mitigation hierarchy components for the Fort Portal-Kamwenge road upgrade project

Components in Mitigation hierarchy principle	Indicators	Actual Score	Max score	%
Avoidance	Identification of Sensitive Areas Alternative Site Assessment Avoidance of High-Risk Activities Compliance with Zoning Regulations	4	4	100
Minimization	Alternative alignments Modifications of the design of the road project Timeline Operations Minimization of construction and operations footprint	4	4	100
Restoration	Restoration and revegetation of temporary worksites Restoration and revegetation of areas/sites disturbed by road-construction activities. Habitat Connectivity Improvement	2	3	67
Offsets/ Compensation	Offset financing mechanism Offset governance/ oversight arrangement Offset implementation plan Offset monitoring Plan Provision for No Net Loss	0	5	0
Total Score		10	16	62

The restoration component of the mitigation hierarchy was partially provided for, attaining a score of 67%. Two of three indicators were addressed in the ESIA regarding revegetation of temporary worksites and road-disturbed areas. Habitat connectivity improvement measures received no provision, resulting in complete loss of forest canopy connections across the upgraded road corridor. The environmental impact assessment documented locations where the forest canopy extended over the original road, as evidenced by pre-upgrade photographs and ecological surveys referenced in the baseline section (Figure 3). Field assessment in October 2023 revealed complete loss of forest canopy connections across the upgraded road corridor. This pattern was consistent along the entire 13-km road section traversing Kibale National Park, where widened carriageway and cleared vegetation zones eliminated canopy connectivity (Figure 4). While Figure 4 does not depict the identical location as Figure 3, it characterizes the typical post-upgrade forest conditions observed throughout the corridor.

Figure 4: Roadside vegetation along the Fort Portal-Kamwenge road prior to upgrade, showing dense forest vegetation extending to the road edge and forming inter-canopy connections across the road corridor (Photo: NEMA, 2011).

Figure 3: Roadside vegetation on both sides of the Fort Portal-Kamwenge road after the upgrade, showing reduced continuity of the ecological corridor and potential canopy fragmentation. (Photo: Aringaniza, 2023)

The EIA recognized the critical importance of canopy connectivity, stating that the selected alternative would maintain current road dimensions within Kibale National Park because "implementing this option will eliminate impacts associated with land take and destruction of trees and their canopies since current road footprint will be maintained. Road kill resulting from primates crossing at road level is bound to be eliminated since destruction of tree canopies will be greatly reduced." However, field observations in Figures 4 and 5 revealed a disconnect between these mitigation commitments and actual outcomes. Additionally, while the EIA committed to not widening the road corridor within Kibale National Park, field measurements during this study revealed that the road currently maintains a standard 6-meter carriageway width, suggesting that widening may have occurred despite mitigation commitments, which also coincides with the total absence of a connected canopy along the entire road corridor. The overhead canopy absence and terrestrial road crossing by primates suggest increased vulnerability to vehicle collisions (Figure 5), directly contradicting the EIA's prediction that canopy preservation would eliminate ground-level crossings. This behavioral response to habitat fragmentation demonstrates the ecological consequences of inadequate implementation of habitat connectivity measures within the restoration component of the mitigation hierarchy. Monkeys, baboons and chimpanzees were documented crossing directly on the road surface (Figure 5). No dedicated animal crossing structures or reptile crossings were observed along the road section.

Figure 5: Baboon observed crossing the road during the study (Photo: Aringaniza, 2023).

The offset/compensation component received no implementation score (0%) across all five indicators. No offset financing mechanisms, governance/oversight arrangements, implementation plans, monitoring plans, or No Net Loss provisions were documented in project planning or implementation. Consequently, permanent residual impacts of the road upgrade remain unaddressed through compensatory measures.

4.1.2 Allocation of Mitigation Measures Across Hierarchy Components

A total of 102 mitigation measures were identified in the Environmental and Social Impact Assessment and Environmental Management Plan documents for the Fort Portal-Kamwenge road project (Figure 6). The mitigation hierarchy analysis revealed an unbalanced distribution across components, with minimization measures dominating the approach (59 measures). Within the minimization component, the majority of efforts focused on reducing construction and operational footprints (38 measures), encompassing dust suppression systems, waste management protocols, strategic equipment yard placement, noise control measures, and comprehensive worker regulations designed to prevent poaching and wildlife disturbance within Kibale National Park.

The avoidance component comprised 26 measures, demonstrating substantive efforts to identify sensitive ecological areas, assess project alternatives, and eliminate high-risk activities such as canopy destruction and habitat fragmentation. Restoration measures were limited to 10 measures, concentrating on revegetation of temporarily disturbed worksites and borrow pit rehabilitation using native species.

Most significantly, the analysis revealed a complete absence of offset/compensation measures specifically designed to address residual biodiversity impacts. This gap is particularly concerning given that project documentation explicitly acknowledges that certain environmental impacts will persist despite the implementation of avoidance, minimization, and restoration measures. The lack of biodiversity offsets represents a critical shortcoming in achieving no net loss of biodiversity, especially considering the project's location within and adjacent to Kibale National Park's high-value primate habitat. Table 3 presents the detailed allocation of these measures across the mitigation hierarchy components based on their respective indicators.

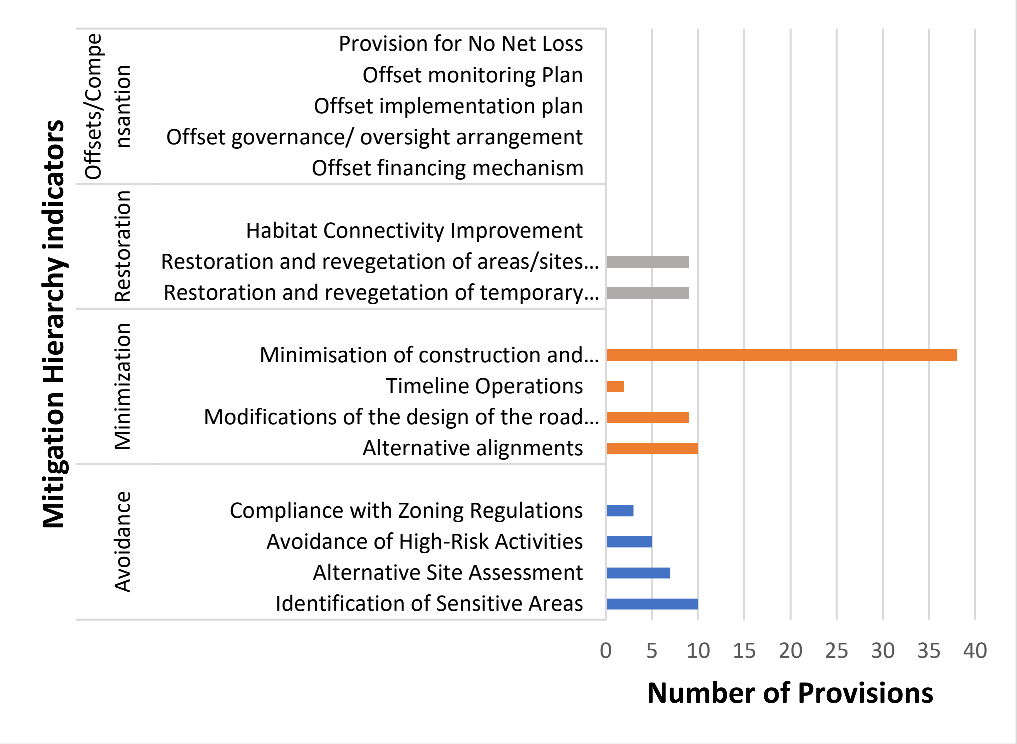


Figure 6: Distribution of mitigation provisions across mitigation hierarchy components

4.2 Land Use and Land Cover (LULC) dynamics

4.2.1 Changes in LULC (2015-2023)

Land use and land cover analysis revealed a distinct pattern of initial forest loss followed by subsequent recovery (Table 3). Forest cover decreased by 3.6% during the construction phase (2015-2017), then increased by 4.9% during the post-construction period (2017-2023), resulting in a net gain of 1.1% over the eight-year period. Grassland expanded continuously, increasing nearly six-fold from 3.87 to 22.14 hectares. Agricultural land peaked during construction (2017) at 19.53 hectares before declining to 3.06 hectares by 2023. Built-up areas similarly increased during construction then returned to pre-construction levels.

Table 3: Land Use & Land Cover Changes along the Fort Portal-Kamwenge Road Corridor (2015-2023)

LULC Type	2015 (ha)	2017 (ha)	2023 (ha)
Forest cover	981.72	946.35	992.88
Grassland	3.87	11.79	22.14
Agricultural land	5.22	19.53	3.06
Structures and facilities	51.3	73.89	51.3

Spatial analysis reveals distinct temporal phases corresponding to construction activities (Figure 7). Pre-construction (2015) shows predominantly intact forest cover. Construction phase (2017) exhibits increased fragmentation with agricultural encroachment and expanded built-up areas at park boundaries. Post-construction (2023) demonstrates substantial forest recovery and reduced agricultural land, though grassland expansion persisted along the road corridor.

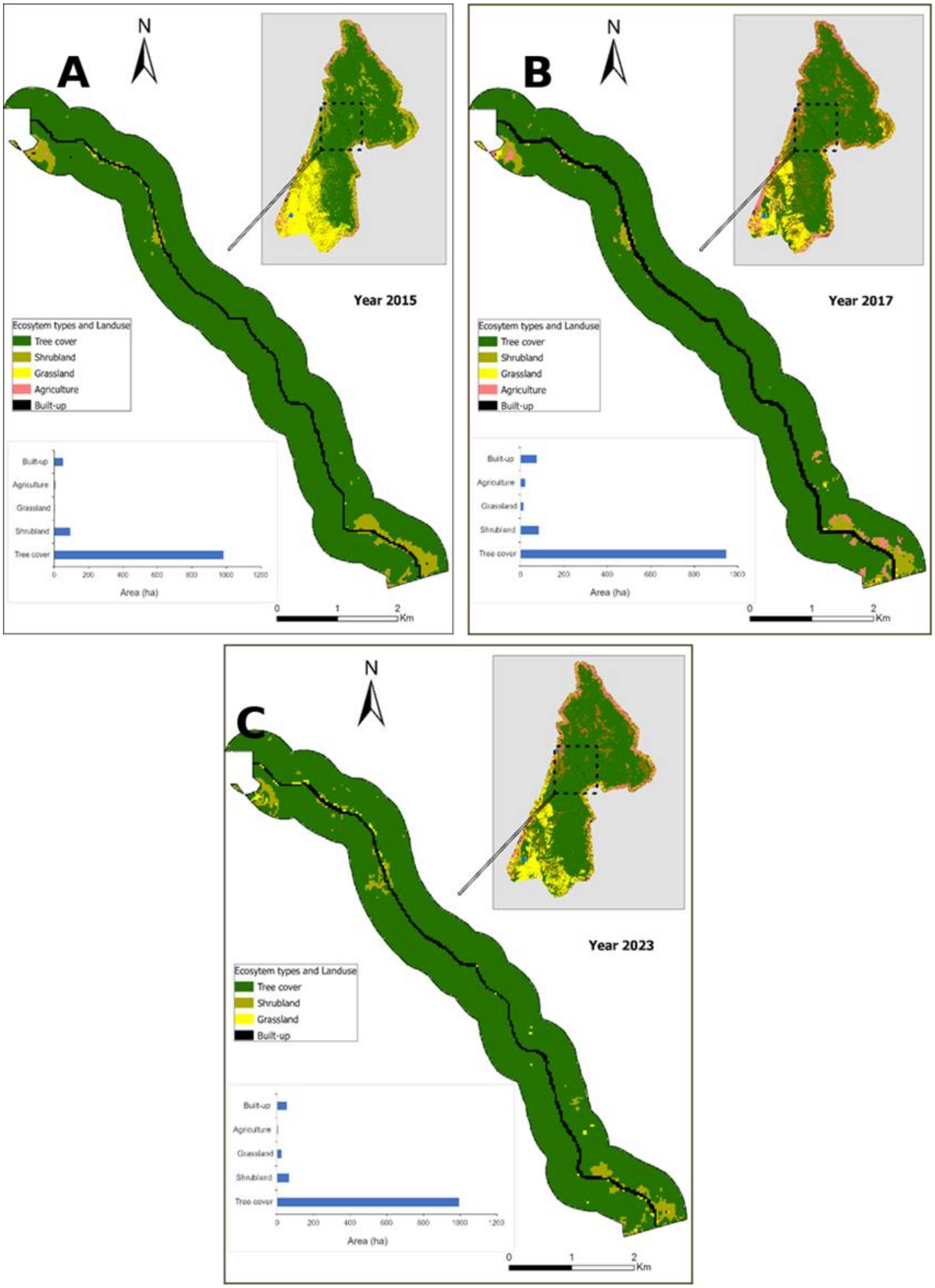
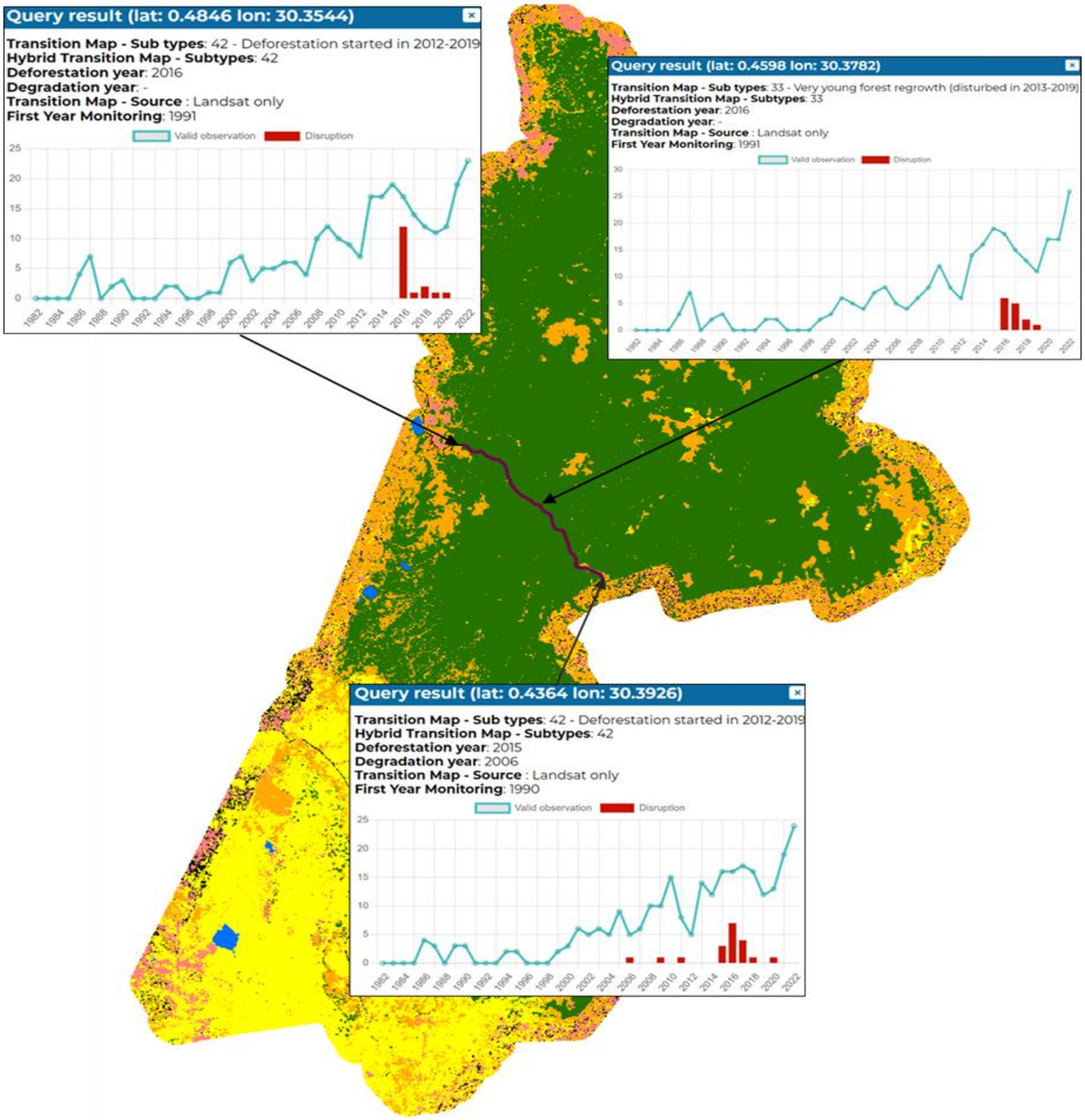


Figure 7: Spatial-temporal changes in land use and land cover within a 500-meter buffer along the Fort Portal-Kamwenge road corridor traversing Kibale National Park: (A) Pre-construction phase (2015) showing predominantly intact forest cover; (B) Construction phase (2017) displaying increased fragmentation, agricultural encroachment, and built-up areas; (C) Post-construction

phase (2023) demonstrating forest recovery and reduction in agricultural land, with persistent expansion of grassland areas.

4.2.2 Forest disturbance trends

The study revealed significant deforestation patterns along the Fortportal-Kamwenge road (Figure 8). In 2016, there was substantial deforestation along all parts of the road, with the edges experiencing the highest rates. In 2017, deforestation escalated within the central segment of the road. Post-2018, deforestation rates stabilized across all sections of the road.



Query result (lat: 0.4846 lon: 30.3544)

Transition Map - Sub types: 42 - Deforestation started in 2012-2019
Hybrid Transition Map - Subtypes: 42
Deforestation year: 2016
Degradation year: -
Transition Map - Source : Landsat only
First Year Monitoring: 1991

Query result (lat: 0.4598 lon: 30.3782)

Transition Map - Sub types: 33 - Very young forest regrowth (disturbed in 2013-2019)
Hybrid Transition Map - Subtypes: 33
Deforestation year: 2016
Degradation year: -
Transition Map - Source : Landsat only
First Year Monitoring: 1991

Query result (lat: 0.4364 lon: 30.3926)

Transition Map - Sub types: 42 - Deforestation started in 2012-2019
Hybrid Transition Map - Subtypes: 42
Deforestation year: 2015
Degradation year: 2006
Transition Map - Source : Landsat only
First Year Monitoring: 1990

Figure 8: Trends in forest disturbance in Kibale national park along Fortportal-Kamwenge road in western Uganda.

4.3 Vegetation composition

4.3.1 Species composition and abundance

The study recorded 94 plant species from 45 families across all sampling plots (Table 4, Appendix A). Fabaceae and Moraceae were the most species-rich families (8 species each), followed by Sapindaceae (6 species) and Apocynaceae (4 species). Regarding conservation status, six species are classified as Vulnerable (*Mondia whitei*, *Warburgia ugandensis*, *Prunus africana*, *Citropsis articulata*, *Fagaropsis angolensis*, and *Chrysophyllum albidum*), one as Endangered (*Cordia millenii*), and three as Near Threatened (*Phoenix reclinata*, *Celtis durandii*, and *Blighia unijugata*) according to the IUCN Uganda National Red List. The vast majority (92 species) were native to the Flora of Tropical East Africa region, while only two (*Senna spectabilis* and *Urtica dioica*) were classified as alien, of which *Senna spectabilis* is widely documented as invasive in tropical forest ecosystems.

Table 4: Summary of plant diversity recorded at varying distances from the Fort Portal-Kamwenge road in Kibale National Park, 2023.

Parameter	Value
Total species	94
Total families	45
Most species-rich families	Fabaceae (8), Moraceae (8), Sapindaceae (6), Apocynaceae (4)
Growth forms	Trees (49), Shrubs (14), Herbs (16), Mixed habit (15)
IUCN Red List status	Vulnerable (6), Endangered (1), Near Threatened (3), Least Concern/Data Deficient (84)
Biogeographic origin	Native (92), Alien (2)

4.3.2 Species similarity across distances

Plant species composition showed no significant variation over a gradient from the road edge into the forest interior (0 m, 50 m, and 100 m). Pairwise comparisons revealed non-significant differences between all distance pairs: road edge and forest interior (0-100 m: ANOSIM R = -0.062, p = 0.762), road edge and intermediate distance (0-50 m: ANOSIM R = -0.147, p = 0.996), and intermediate distance and forest interior (50-100 m: ANOSIM R = -0.109, p = 0.959). These results were further corroborated by NMDS ordination, which demonstrated substantial overlap in plant community composition across the three distances, as evidenced by the proximity of sampling points in ordination space (Figure 9). The substantial overlap of sample points across distance categories (0 m, 50 m, and 100 m) indicates minimal differentiation in community composition.

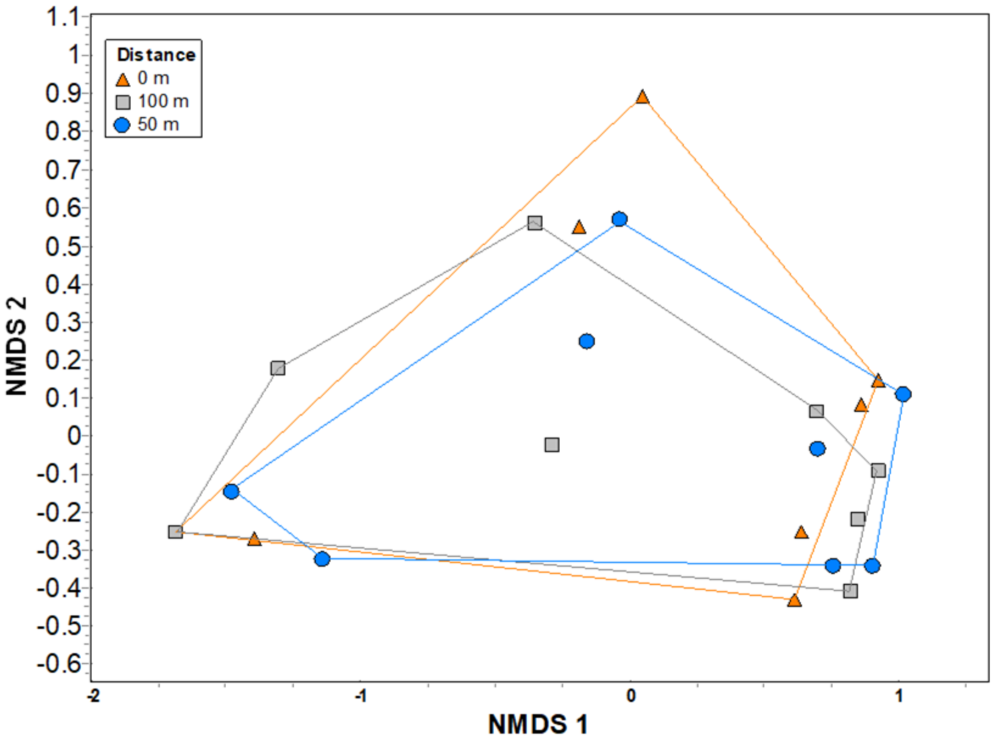


Figure 9: Overlapping distribution of plant species composition showing no significant differences at 0 m, 50 m, and 100 m distances from the road edge.

4.3.3 Stem density

The density of woody vegetation showed no significant variation with increasing distance from the road edge. The mean woody stem density values were 2,480 ± 1,340 stems/ha at the road edge (0 m), 3,630 ± 2,200 stems/ha at intermediate distance (50 m), and 2,270 ± 1,340 stems/ha in the forest interior (100 m). These differences were not statistically significant (ANOVA: F = 0.125, df = 2, P = 0.881; Figure 10). The stem density values include all woody individuals ≥0.5 cm diameter within the 10 m × 10 m sampling plots, encompassing mature trees, saplings, and woody shrubs. The high stem densities reflect the inclusion of multiple strata of woody vegetation typical of tropical forest ecosystems, where numerous understory and mid-story individuals can occupy a single plot alongside fewer large canopy trees. This consistent stem density across the distance gradient further supports the finding that edge effects on forest structure were minimal eight years after road upgrade the land use and land cover analysis conducted in this study.

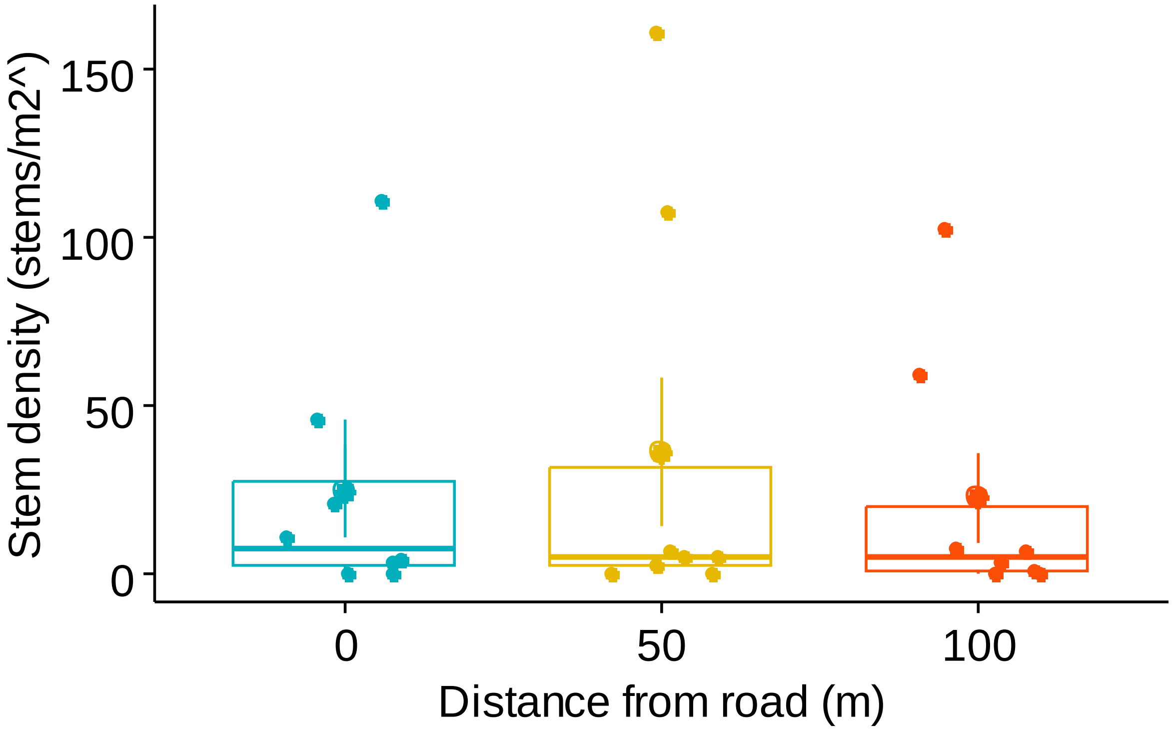


Figure 10: Woody stem density across different distances from the Fort Portal-Kamwenge road in Kibale National Park. Box plots show median (horizontal line), interquartile range (box), and distribution of individual plot values (colored points). No significant differences were detected among the three distances (ANOVA: F = 0.125, df = 2, P = 0.881).

CHAPTER FIVE

DISCUSSION

5.1 Application of the mitigation hierarchy

There was differential provision for the various stages of the mitigation hierarchy in the ESIA report, with complete provision for avoidance and minimization strategies (100%), partial provision for restoration efforts (67%), and a total absence of offset/compensation measures (0%). The successful provision for avoidance and minimization indicates a strong focus on early-stage mitigation, which is essential in reducing immediate environmental impacts. Empirical evidence from infrastructure projects globally supports that avoidance and minimization, when rigorously implemented, produce more certain and cost-effective conservation outcomes than later-stage mitigation components (Phalan et al., 2018; Kiesecker et al., 2010). Studies comparing mitigation effectiveness across the hierarchy consistently demonstrate that preventing impacts through strategic siting and design modifications (avoidance and minimization) achieves better biodiversity outcomes per unit cost than attempting to restore degraded ecosystems or offset impacts elsewhere (Bull et al., 2016; Maron et al., 2016). In this project, the decision to upgrade an existing road rather than construct a new route through intact forest exemplifies effective avoidance, potentially preventing 5-10 times greater forest loss than would have occurred with new construction through the park (Laurance et al., 2015; Kleinschroth et al., 2017). Although restoration achieved a 67% implementation score, a critical gap was identified in habitat connectivity measures. This partial implementation suggests that while early-stage mitigation (avoidance and minimization) received adequate attention, the sequential logic of the mitigation hierarchy, which requires comprehensive application across all four stages to achieve no net loss, was not fully realized (Bull et al., 2013; Phalan et al., 2018).

Pre-construction field observations documented in the Environmental Impact Assessment (NEMA, 2011) revealed forest canopy extending near or over sections of the original road, but these areas of potential canopy connectivity were eliminated during road widening with no compensatory measures implemented by either the contractor or Uganda National Roads Authority as the implementing agency. Consequently, indicating that while some restoration activities were implemented, critical connectivity restoration measures were absent. This partial provision means that degraded areas received some rehabilitation attention, but the most significant ecological impact, loss of canopy connectivity enabling arboreal wildlife movement, remained unaddressed, potentially perpetuating fragmentation effects and wildlife mortality risk over the long term (Laurance et al., 2009; Goosem et al., 2021). This connectivity loss creates a functional barrier for arboreal species including primates, squirrels, and tree-dwelling reptiles that rely on canopy pathways for movement, foraging, and dispersal (Chapman et al., 2016). The ecological consequences include: (1) increased wildlife mortality as species attempt risky ground-level crossings, (2) genetic isolation of populations on either side of the road preventing gene flow necessary for long-term viability, (3) restricted ranging behavior reducing access to food resources and mates, and (4) cascading effects on seed dispersal and forest regeneration that depend on animal movement (Arroyo-Rodríguez et al., 2017; Epps et al., 2005). These impacts are likely to intensify over time as populations become progressively more isolated, creating an extinction debt that may not manifest demographically for decades but is inevitable given the barrier effect (Tilman et al., 1994).

Restoration plays a fundamental role in rehabilitating ecosystems affected by infrastructure projects such as roads, pipelines, transmission lines, and mining operations where construction activities temporarily or permanently degrade habitat quality (Meli et al., 2017; Chazdon & Guariguata, 2016). Successful restoration examples include revegetation of roadside cut slopes to prevent erosion and facilitate natural regeneration, rehabilitation of borrow pits and quarries to restore habitat connectivity, installation of wildlife crossing structures to reconnect fragmented populations, and assisted regeneration in degraded forest edges to reduce edge effect penetration (van der Ree et al., 2015; Crouzeilles et al., 2016), and this specific connectivity gap represents a significant ecological liability.

The Environmental and Social Management Plan (NEMA,2011) acknowledged the importance of preserving forest canopy for wildlife movement but lacked specific measures to maintain or restore this connectivity feature. Similar studies have shown that disruption of arboreal movement pathways can result in prolonged ecological impacts, impeding wildlife movement and increasing road mortality risk (Anderson et al., 2018). The observed behavior of primates crossing at road level provides direct evidence of this impact, as documented in the field observations and consistent with previous findings that inadequate attention to habitat connectivity can impair wildlife movement and survival, exacerbating fragmentation effects (Garcia & Lopez, 2019).

More critically, offset and compensation measures were absent, representing a major gap in the mitigation strategy. Despite the Environmental and Social Management Plan identifying residual impacts that would persist after implementation of avoidance, minimization, and restoration measures, particularly, regarding habitat fragmentation and canopy discontinuity, no offsetting provisions were established. Offsets serve as a crucial safeguard within the hierarchy, addressing residual impacts that cannot be mitigated through earlier measures.

In road construction projects that traverse protected areas, the absence of offset mechanisms has been linked to persistent ecological impacts including progressive biodiversity erosion through cumulative unmitigated impacts, genetic isolation of fragmented populations leading to inbreeding depression and local extinctions, disruption of ecological processes such as seed dispersal and pollination that depend on landscape connectivity, and degradation of ecosystem services including carbon storage, water regulation, and habitat provision (Laurance et al., 2009; Gibson et al., 2011; Barlow et al., 2016). Studies from Brazilian Amazon roads through protected areas documented that projects without offset provisions experienced 30-50% greater long-term forest loss compared to projects where offsets funded enhanced protection of equivalent habitat elsewhere (Soares-Filho et al., 2006; Fearnside, 2015). The observed terrestrial crossing behaviour by primates represents the type of residual impact that offset measures could have addressed through installation of artificial canopy bridges or dedicated wildlife crossing structures.

This practice of prioritizing avoidance and minimization while neglecting restoration and offsetting appears in similar situations in infrastructure projects globally, including road developments in Southeast Asian protected areas (Alamgir et al., 2017), pipeline projects in African savannas (Sloan et al., 2018), transmission line corridors through European forests (Eldegard et al., 2015), and mining operations in Australian biodiversity hotspots (Sonter et al., 2020). zu Ermgassen et al. (2019) found that across 34 countries with biodiversity offset policies, fewer than 30% of infrastructure projects actually implemented offset measures despite regulatory requirements, with implementation rates lowest in developing countries with limited enforcement capacity.

Financial and logistical constraints often result in ecological assessments that focus on the initial mitigation stages, leaving restoration and offset measures inadequately addressed (Jacob et al., 2016). Bigard et al. (2017) also noted that such implementation gaps can have cascading effects on ecosystem services, potentially leading to species loss and long-term environmental imbalance. The implementation assessment conducted in this study revealed limited documentation on restoration monitoring and complete absence of offset planning, suggesting institutional and regulatory frameworks may require strengthening to ensure comprehensive mitigation hierarchy implementation.

Financial, time, and knowledge constraints play a critical role in shaping mitigation hierarchy implementation in infrastructure projects, often leading practitioners to prioritize avoidance and minimization over restoration and compensation (Bull et al., 2016; Kiesecker et al., 2010). This tendency is particularly evident in developing countries, where capacity limitations further exacerbate these challenges. Project developers and implementing agencies frequently focus on familiar avoidance and minimization strategies that can be integrated into engineering specifications, while more complex restoration or offset measures such as wildlife crossing structure installation requiring specialized engineering design, long-term revegetation programs demanding botanical expertise and multi-year monitoring, offset site identification and management requiring spatial planning and conservation biology skills, and adaptive management frameworks necessitating systematic ecological monitoring and statistical analysis capability (van der Ree et al., 2015; Maron et al., 2012).

Gelot and Bigard (2021) highlight that institutional capacity gaps in developing regions further hinder the adoption of comprehensive mitigation strategies through multiple mechanisms including: (1) insufficient numbers of trained professionals capable of designing species-specific mitigation measures, (2) lack of institutional memory and knowledge management systems that would enable learning from past projects, (3) inadequate equipment and technology for ecological monitoring and assessment, (4) weak inter-agency coordination between transportation, environment, and wildlife authorities leading to fragmented implementation, and (5) limited access to international best practices and technical guidance due to language barriers and resource constraints.

Gelot and Bigard (2021) highlight that institutional capacity gaps in developing regions further hinder the adoption of comprehensive mitigation strategies, with financial mechanisms often creating implementation barriers by concentrating available budgets on construction activities while treating environmental mitigation as an unfunded mandate, by structuring payment schedules that incentivize rapid project completion over environmental performance, by failing to allocate dedicated funds for long-term post-construction monitoring and adaptive management, and by requiring competitive bidding processes that favor lowest-cost contractors who may lack environmental expertise or commitment (Balmford & Whitten, 2003; Ferraro & Pattanayak, 2006).

Moreover, economic considerations are key determinants of successful mitigation efforts, with financial mechanisms often creating implementation barriers in resource-limited contexts such as sub-Saharan African countries where infrastructure budgets are constrained by competing development priorities including health, education, and poverty reduction, Southeast Asian nations balancing rapid economic growth with environmental protection, and small island developing states with limited technical expertise and financial resources for sophisticated mitigation approaches (Bidandi & Williams, 2018; Ezzine-de-Blas et al., 2017)

The Fort Portal-Kamwenge road upgrade began in 2015, predating Uganda's formal integration of the mitigation hierarchy into national environmental legislation through the National Environment Act (Government of Uganda, 2019) and the Environmental and Social Assessment Regulations (Government of Uganda, 2020), which likely contributed to the lack of comprehensive mitigation planning. This timing aligns with observations by Maron et al. (2018) in the global review of offset policies that the absence of clear regulatory requirements often results in incomplete implementation of the full mitigation hierarchy. The differential application of mitigation components observed in this project reflects broader institutional and regulatory evolution in Uganda's environmental management framework, suggesting that infrastructure projects initiated under current legislative requirements might demonstrate more balanced application across all mitigation hierarchy components.

While avoidance and minimization measures contributed to observable ecosystem recovery in post-construction, significant gaps in the implementation of restoration and offsetting measures highlight incomplete application of the mitigation hierarchy framework. This research reveals that infrastructure development can coexist with ecological conservation when complete mitigation hierarchy is properly implemented. However, current approaches often prioritize early intervention stages while neglecting comprehensive long-term measures that would be desirable for achieving the mitigation hierarchy's fundamental objective of no net loss of biodiversity, maintaining ecological connectivity and population viability for wide-ranging species, preventing cumulative degradation across multiple infrastructure projects in the same landscape, and reconciling development imperatives with international conservation commitments under the Convention on Biological Diversity and Sustainable Development Goals (Bull et al., 2013; Maron et al., 2012).

5.2 Land Use and Land Cover dynamics

Land Use and Land Cover (LULC) changes observed along the Fort Portal-Kamwenge road corridor between 2015 and 2023 reveal a complex ecological transformation that challenges conventional assumptions about road infrastructure impacts on forest ecosystems. The multi-temporal analysis identified a dynamic pattern of vegetation change, with an initial decline in tree cover from 2015 to 2017, followed by a notable recovery that exceeded 2015 baseline levels by 2023. This trajectory suggests a degree of ecosystem resilience that demands rigorous scientific interpretation because apparent forest recovery observed through satellite imagery may mask important qualitative changes in forest composition and function that remote sensing cannot detect (Meli et al., 2017; Wilson et al., 2020). Forest cover recovery does not necessarily equate to ecological recovery, as regenerating vegetation may differ fundamentally from original forest in species composition (pioneer-dominated vs. old-growth assemblages), structural complexity (simplified vs. multi-layered canopy), functional diversity (weedy generalists vs. specialized taxa), and ecosystem service provision (reduced carbon storage, altered hydrology, diminished wildlife habitat quality) (Chazdon, 2014; Barlow et al., 2016).

While the observed canopy regeneration may initially seem promising, it is important to interpret these findings with caution because satellite-based land cover classifications detect canopy closure and vegetation greenness but cannot distinguish between forest types differing in conservation value. The 30-meter resolution of Landsat imagery used in this study cannot resolve species-level composition, meaning that forests classified identically may be dominated by native old-growth species versus invasive pioneer species, may contain high versus low wildlife habitat quality, or may provide substantially different ecosystem services (Nagendra et al., 2013). Furthermore, the eight-year observation period may capture only initial rapid growth of pioneer species rather than long-term successional trajectories, as tropical forest succession can require 50-100 years to approach old-growth composition and structure (Guariguata & Ostertag, 2001). The caution is warranted because premature interpretation of forest cover recovery as ecological success could lead to complacency about mitigation effectiveness when in reality, significant biodiversity impacts persist beneath an apparently recovered canopy. Consistent with Chazdon et al. (2016), rapid forest regrowth does not necessarily indicate full ecological restoration. Instead, the recovery may be dominated by fast-growing pioneer species, resulting in a simplified forest structure that lacks the complexity of the original forest ecosystem (Rozendaal et al., 2019; Ghazoul & Chazdon, 2017). This distinction is crucial for understanding the long-term ecological trajectory of road-adjacent landscapes.

The systematic expansion of grassland areas by 22.14 ha throughout the study period presents a particularly compelling research observation. Such expansion could be interpreted through multiple ecological lenses: as a potential transitional stage in forest regeneration (Crouzeilles et al., 2016) or as an indicator of more profound ecosystem structural transformations (Malhi et al., 2014). Drawing from Laurance et al. (2009), the observed edge effects in tropical forest ecosystems, which can extend hundreds of meters from linear infrastructure, may create microenvironmental conditions that are more conducive to grass species proliferation. Consequently, the grassland expansion might serve as an early empirical signal of potential long-term ecological restructuring (Catterall et al., 2016).

The dynamics of agricultural land use also exhibited a complex spatial-temporal pattern, peaking in 2017 before declining by 2023. This trajectory reflects the socio-ecological interactions commonly associated with road infrastructure development (Kleinschroth & Healey, 2017). The initial expansion of agriculture aligns with patterns observed by Laurance et al. (2014), where road improvements often facilitate agricultural frontier expansion in tropical regions. The subsequent agricultural land use reduction is attributable to stringent protected area management enforcement following the completion of road construction. Field verification conducted during the study confirmed increased patrol frequency along the road corridor after 2017, effectively limiting agricultural encroachment within the national park boundaries. This pattern of initial resource exploitation followed by institutional response aligns with findings from other protected areas intersected by transportation infrastructure (van der Ree et al., 2015; Brandão Jr et al., 2020).

Developed area dynamics demonstrated a similar complex pattern, with initial expansion followed by a return to 2015 baseline levels. This observation suggests potential effectiveness of spatial planning interventions and infrastructure development management strategies. This outcome aligns with the goals of spatial planning in infrastructure projects and demonstrates the potential effectiveness of well-implemented avoidance and minimization strategies in the mitigation hierarchy (van der Ree et al., 2015).

5.3 Vegetation composition

Analysis of forest composition along the Fort Portal-Kamwenge road revealed patterns that deviate from published information on road impacts in tropical forests. While this study detected no significant compositional changes at distances of 0-100 m from the road edge, previous research has documented substantial edge effects at much greater distances in other tropical forests. For instance, Ahmed et al. (2014) documented compositional changes extending up to 300 m from road edges in Amazonian forests, while Broadbent et al. (2008) identified edge-related impacts propagating to 400 m in other tropical forest ecosystems. This pronounced discrepancy in edge effect spatial dynamics suggests the critical mediating role of localized ecological conditions in modulating infrastructure-induced forest transformations (Harper et al., 2015).

Several mechanisms could explain these unexpected patterns. First, historical disturbance patterns in the study area may have created a more homogeneous forest structure less susceptible to new edge effects (Fletcher et al., 2018). This aligns with Ewers and Didham's (2006) findings that edge effects can be less pronounced in previously disturbed forests where edge-adapted species already dominate. Second, while the vegetation composition assessment was conducted eight years after road construction, the LULC analysis spanning 2015-2023 may still capture only the initial post-disturbance period. Harper et al. (2005) demonstrated that some compositional changes in tropical forests continue developing beyond this timeframe, suggesting that more pronounced edge effects might emerge in subsequent decades as forest succession processes continue.

Temporal resolution limitations represent another critical consideration. Harper et al. (2005) demonstrated that substantive compositional modifications in tropical forest ecosystems can manifest progressively over extended periods, potentially requiring 5-10 years to become fully detectable. The current observational timeframe may thus capture only an initial stage of ecological transformation. Despite apparent compositional stability, two ecologically significant transformations emerged. The invasive plant species *Acanthus pubescens* Engl. demonstrated increasing abundance near forest edges, a pattern consistent with meta-analytical findings by van der Ree et al. (2015) showing that roads frequently facilitate invasive species establishment in tropical forests. Acanthus pubescens' presence particularly threatens understory diversity, as documented by Sharma et al. (2016) in similar forest ecosystems.

A notable structural change was observed in the forest canopy. Field observations documented complete absence of canopy connectivity across the road corridor throughout the 13 km section traversing the national park. This canopy discontinuity represents a significant barrier to arboreal movement, as evidenced by direct observations of terrestrial road crossing by primates that typically travel through the canopy. Arroyo-Rodríguez et al. (2017) demonstrated that canopy discontinuity affects movement patterns of arboreal mammals, while Magnago et al. (2015) found reduced epiphyte diversity in areas with fragmented canopy cover. These findings suggest that while overall forest composition appears stable, the vertical structure has been notably altered. These results highlight the complex nature of forest responses to road development and emphasize the need for multi-dimensional approaches in assessing ecological impacts. The observed patterns suggest that conventional metrics of forest composition may not capture all relevant ecological changes, particularly in the early years following road construction (Jakovac et al., 2016).

CHAPTER SIX

CONCLUSIONS AND RECOMENDATIONS

6.1 Conclusions

This study evaluated the application of the mitigation hierarchy along the Fort Portal-Kamwenge road upgrade through Kibale National Park and its effectiveness in safeguarding biodiversity. The following conclusions are drawn from the findings:

1. Application of the mitigation hierarchy

The mitigation hierarchy was partially provided for, achieving an overall score of 62%. Avoidance and minimization measures were fully provided for (100% each), as evidenced by tree recovery and stable forest composition after construction. Restoration measures achieved partial implementation (67%) with successful revegetation of disturbed areas but failed to address habitat connectivity improvement. Offset/compensation measures were entirely absent (0%), leaving residual impacts unaddressed.

2. Spatial-temporal changes in Land Use and Land Cover

Land cover analysis revealed initial forest loss (-3.6%) during construction (2015-2017), followed by significant recovery (+4.9%) post-construction (2017-2023), resulting in net forest gain of 1.1% over eight years. Changes were most pronounced adjacent to the road corridor, with herbaceous vegetation expanding consistently throughout the study period.

3. Vegetation composition along the road corridor

Forest composition showed no significant differences across the 0-100 m distance gradient from the road edge, suggesting that there is limited edge effects within this range. However, complete canopy discontinuity was documented along the entire road section, eliminating potential arboreal crossing pathways for primates and other mammals.

6.2 Recommendations

Based on the conclusions of this study, the following are recommended:

1. **Application of the mitigation hierarchy**
   - NEMA should strengthen regulatory enforcement to ensure comprehensive implementation of all four mitigation hierarchy stages, in accordance with Section 115 of the National Environment Act, 2019 and the Environmental and Social Assessment Regulations, 2020. Clear guidelines for restoration and offset activities should be developed, supported by adequate funding mechanisms and long-term sustainability plans. Mandatory offset or compensation requirements should be applied to infrastructure projects in protected areas to address residual impacts.
2. **Land use and land cover monitoring**
   - Ministry of Works and Transport, in collaboration with UWA, should implement specific protocols to restore and maintain canopy connectivity across road corridors traversing protected areas. This should include installation of canopy bridges, rope corridors, or other appropriate wildlife crossing structures to facilitate arboreal species movement and reduce wildlife vehicle collision risks and ecological connectivity within protected areas. These authorities should also institutionalize post-construction land use and land cover monitoring within defined road influence zones to track disturbance, recovery, and persistent edge effects.
3. **Vegetation composition management**
   - UWA should extend post-construction ecological monitoring beyond the current eight-year timeframe to capture long-term vegetation recovery patterns and delayed ecological impacts. UWA should implement comprehensive buffer zone management protocols along the road corridor to control herbaceous expansion, manage invasive species, and promote native forest regeneration while maintaining forest integrity adjacent to road corridors. Regular monitoring of wildlife crossing behavior should be undertaken to evaluate the effectiveness of connectivity restoration measures and inform adaptive management in the forest
4. **Future research**

To address remaining knowledge gaps and strengthen evidence for biodiversity-friendly road development in tropical forest protected areas, future research should prioritize:

- Long-term monitoring (20–30 years) to determine whether current vegetation stability is sustained or whether delayed edge effects emerge over time.
- Road mortality assessments across taxa (mammals, birds, reptiles, amphibians), including standardized carcass surveys accounting for scavenger removal and detection probability.
- Connectivity and fragmentation studies using genetic approaches to test whether the road is restricting gene flow for primates and other key species.
- Effectiveness evaluation of canopy bridges/wildlife crossings using before–after or BACI study designs, combining behavioral monitoring, camera trapping, and population indicators.
- Remote sensing innovations (e.g., LiDAR or very high-resolution imagery) to quantify canopy connectivity and subtle structural forest changes that standard LULC classes may miss.
- Comparative spatial analyses across reference forest areas to assess variability in ecological impacts and mitigation outcomes, and to distinguish localized effects from broader, landscape-level patterns.

REFERENCES

Adams, W.M., Aveling, R., Brockington, D., Dickson, B., Elliott, J., Hutton, J., Roe, D., Vira, B. & Wolmer, W. 2004. Biodiversity conservation and the eradication of poverty. Science, 3065699, 1146-1149.

AfDB (African Development Bank) 2012. African Development Bank's Integrated Safeguards System: Policy Statement and Operational Safeguards. African Development Bank Group, Tunis.

AfDB (African Development Bank) 2013. Safeguards and Sustainability Series Volume 1 - Issue 1 (Dec. 2013): African Development Bank Group's Integrated Safeguards System. African Development Bank Group, Tunis.

Ahmed, S.E., Lees, A.C., Moura, N.G., Gardner, T.A., Barlow, J., Ferreira, J. & Ewers, R.M. 2014. Road networks predict human influence on Amazonian bird communities. Proceedings of the Royal Society B, 2851892, 20172497.

Alamgir, M., Campbell, M.J., Sloan, S., Goosem, M., Clements, G.R., Mahmoud, M.I. & Laurance, W.F. 2017. Economic, socio-political and environmental risks of road development in the tropics. Current Biology, 27(20), R1130-R1140.

Alignier, A. & Deconchat, M. 2011. Variability of forest edge effect on vegetation implies reconsideration of its assumed hypothetical pattern. Applied Vegetation Science, 14(1), 67-74.

Anderson, M.J. 2001. A new method for non-parametric multivariate analysis of variance. Austral Ecology, 26(1), 32-46.

Anderson, M.J., Gorley, R.N. & Clarke, K.R. 2008. PERMANOVA+ for PRIMER: Guide to Software and Statistical Methods. PRIMER-E, Plymouth, UK.

Andrén, H. 1994. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: A review. Oikos, 71(3), 355-366.

Apostolopoulou, E. & Adams, W.M. 2017. Biodiversity offsetting and conservation: Reframing nature to save it. Oryx, 51(1), 23-31.

Arima, E.Y., Barreto, P., Araújo, E. & Soares-Filho, B. 2014. Public policies can reduce tropical deforestation: Lessons and challenges from Brazil. Land Use Policy, 41, 465-473.

Arlidge, W.N., Bull, J.W., Addison, P.F., Burgass, M.J., Gianuca, D., Gorham, T.M., Jacob, C., Shumway, N., Sinclair, S.P., Watson, J.E. & Wilcox, C. 2018. A global mitigation hierarchy for nature conservation. BioScience, 68(5), 336-347.

Arroyo-Rodríguez, V., Rös, M., Escobar, F., Melo, F.P., Santos, B.A., Tabarelli, M. & Chazdon, R. 2013. Plant β-diversity in fragmented rain forests: Testing floristic homogenization and differentiation hypotheses. Journal of Ecology, 101(6), 1449-1458.

Arroyo-Rodríguez, V., Saldaña-Vázquez, R.A., Fahrig, L. & Santos, B.A. 2017. Does forest fragmentation cause an increase in forest temperature? Ecological Research, 32(1), 81-88.

Arts, J., Caldwell, P. & Morrison-Saunders, A. 2001. Environmental impact assessment follow-up: Good practice and future directions. Impact Assessment and Project Appraisal, 19(3), 175-185.

Arts, J., Caldwell, P. & Morrison-Saunders, A. 2012. Environmental impact assessment follow-up: Good practice and future directions–findings from a workshop at the IAIA 2000 conference. Impact Assessment and Project Appraisal, 19(3), 175-185.

Asner, G.P. 2001. Cloud cover in Landsat observations of the Brazilian Amazon. International Journal of Remote Sensing, 22(18), 3855-3862.

Asner, G.P., Knapp, D.E., Broadbent, E.N., Oliveira, P.J., Keller, M. & Silva, J.N. 2005. Selective logging in the Brazilian Amazon. Science, 3105747, 480-482.

Asner, G.P., Knapp, D.E., Martin, R.E., Tupayachi, R., Anderson, C.B., Mascaro, J., Sinca, F., Chadwick, K.D., Higgins, M., Farfan, W., Llactayo, W. & Silman, M.R. 2014. Targeted carbon conservation at national scales with high-resolution monitoring. Proceedings of the National Academy of Sciences, 111(47), E5016-E5022.

Asner, G.P., Martin, R.E., Anderson, C.B. & Knapp, D.E. 2015. Quantifying forest canopy traits: Imaging spectroscopy versus field survey. Remote Sensing of Environment, 158, 15-27.

Australian Government Department of Agriculture, Water and the Environment 2021. Environment Protection and Biodiversity Conservation Act 1999. Australian Government, Canberra.

Avon, C., Bergès, L., Dumas, Y. & Dupouey, J.L. 2010. Does the effect of forest roads extend a few meters or more into the adjacent forest? A study on understory plant diversity in managed oak stands. Forest Ecology and Management, 259(8), 1546-1555.

Baguette, M., Blanchet, S., Legrand, D., Stevens, V.M. & Turlure, C. 2013. Individual dispersal, landscape connectivity and ecological networks. Biological Reviews, 88(2), 310-326.

Balkenhol, N. & Waits, L.P. 2009. Molecular road ecology: Exploring the potential of genetics for investigating transportation impacts on wildlife. Molecular Ecology, 18(20), 4151-4164.

Balmford, A. & Whitten, T. 2003. Who should pay for tropical conservation, and how could the costs be met? Oryx, 37(2), 238-250.

Barber, C.P., Cochrane, M.A., Souza Jr, C.M. & Laurance, W.F. 2014. Roads, deforestation, and the mitigating effect of protected areas in the Amazon. Biological Conservation, 177, 203-209.

Barlow, J., Lennox, G.D., Ferreira, J., Berenguer, E., Lees, A.C., Mac Nally, R., Thomson, J.R., Ferraz, S.F.D.B., Louzada, J., Oliveira, V.H.F. & Parry, L. 2016. Anthropogenic disturbance in tropical forests can double biodiversity loss from deforestation. Nature, 5357610, 144-147.

BBOP (Business and Biodiversity Offsets Programme) 2012. Standard on Biodiversity Offsets. BBOP, Washington, D.C.

Beckmann, J.P., Clevenger, A.P., Huijser, M.P. & Hilty, J.A. 2010. Safe Passages: Highways, Wildlife, and Habitat Connectivity. Island Press, Washington, DC.

Beier, P. & Noss, R.F. 1998. Do habitat corridors provide connectivity? Conservation Biology, 12(6), 1241-1252.

Benítez-López, A., Alkemade, R. & Verweij, P.A. 2010. The impacts of roads and other infrastructure on mammal and bird populations: A meta-analysis. Biological Conservation, 143(6), 1307-1316.

Benitez-Malvido, J. & Martinez-Ramos, M. 2013. Impact of forest fragmentation on understory plant species richness in Amazonia. Conservation Biology, 17(2), 389-400.

Bennett, A.F. 2003. Linkages in the Landscape: The Role of Corridors and Connectivity in Wildlife Conservation (2nd ed.). IUCN, Gland, Switzerland.

Bidandi, F. & Williams, J.J. 2018. Understanding the discourse and practice of benefit sharing with a case from the Toro-Semliki Wildlife Reserve, Uganda. Conservation and Society, 16(1), 35-46.

Bigard, C., Pioch, S. & Thompson, J.D. 2017. The inclusion of biodiversity in environmental impact assessment: Policy-related progress limited by gaps and semantic confusion. Journal of Environmental Management, 200, 35-45.

Bigard, C., Pioch, S., & Thompson, J. D. 2021. Challenges to developing mitigation hierarchy policy: Findings from a nationwide database analysis in France. Biological Conservation, 263, 109343.

Blois, J.L., Williams, J.W., Fitzpatrick, M.C., Jackson, S.T. & Ferrier, S. 2013. Space can substitute for time in predicting climate-change effects on biodiversity. Proceedings of the National Academy of Sciences, 110(23), 9374-9379.

Boesch, C. & Boesch-Achermann, H. 2000. The Chimpanzees of the Taï Forest: Behavioural Ecology and Evolution. Oxford University Press, Oxford.

Brandão Jr, A., Souza Jr, C., Ribeiro, J.G., Sales, M. & Hayashi, S. 2020. Deforestation and fires in the Brazilian Amazon from 2001 to 2020: Impacts and drivers. Sustainability in Debate, 11(3), 34-56.

Briant, G., Gond, V. & Laurance, S.G. 2010. Habitat fragmentation and the desiccation of forest canopies: A case study from eastern Amazonia. Biological Conservation, 143(11), 2763-2769.

Broadbent, E.N., Asner, G.P., Keller, M., Knapp, D.E., Oliveira, P.J. & Silva, J.N. 2008. Forest fragmentation and edge effects from deforestation and selective logging in the Brazilian Amazon. Biological Conservation, 141(7), 1745-1757.

Brooks, T.M., Mittermeier, R.A., da Fonseca, G.A., Gerlach, J., Hoffmann, M., Lamoreux, J.F., Mittermeier, C.G., Pilgrim, J.D. & Rodrigues, A.S. 2006. Global biodiversity conservation priorities. Science, 3135783, 58-61.

Bull, J.W., Suttle, K.B., Gordon, A., Singh, N.J. & Milner-Gulland, E.J. 2013. Biodiversity offsets in theory and practice. Oryx, 47(3), 369-380.

Bull, J.W., Hardy, M.J., Moilanen, A. & Gordon, A. 2015. Categories of flexibility in biodiversity offsetting, and their implications for conservation. Biological Conservation, 192, 522-532.

Bull, J.W., Gordon, A., Watson, J.E. & Maron, M. 2016. Seeking convergence on the key concepts in 'no net loss' policy. Journal of Applied Ecology, 53(6), 1686-1693.

Bull, J.W., Brauneder, K., Darbi, M., van Teeffelen, A.J., Quétier, F., Brooks, S.E., Dunnett, S. & Strange, N. 2018. Data transparency regarding the implementation of European 'no net loss' biodiversity policies. Biological Conservation, 218, 64-72.

Bull, J.W., Milner-Gulland, E.J., Suttle, K.B. & Singh, N.J. 2020. Comparing biodiversity offset calculation methods with a case study in Uzbekistan. Biological Conservation, 178, 2-10.

Bull, J.W., Strange, N., Smith, R.J. & Gordon, A. 2020. Reconciling multiple counterfactuals when evaluating biodiversity conservation impact in social-ecological systems. Conservation Biology, 34(4), 895-905.

Bull, J.W., Milner-Gulland, E.J., Addison, P.F., Arlidge, W.N., Baker, J., Brooks, T.M., Taylor, M.L., Weller, K., zu Ermgassen, S.O., Sidemo-Holm, W. & Watson, J.E. 2022. Net positive outcomes for nature. Nature Ecology & Evolution, 6(7), 869-877.

Busetto, L., Meroni, M. & Colombo, R. 2008. Combining medium and coarse spatial resolution satellite data to improve the estimation of sub-pixel NDVI time series. Remote Sensing of Environment, 112(1), 118-131.

Cardelús, C.L., Colwell, R.K. & Watkins Jr, J.E. 2006. Vascular epiphyte distribution patterns: Explaining the mid-elevation richness peak. Journal of Ecology, 94(1), 144-156.

Cares, R. A., & Franco, A. M. A. 2023. Investigating the implementation of the mitigation hierarchy approach in environmental impact assessment in relation to biodiversity impacts. Environmental Impact Assessment Review, 102, 107194.

Cardinale, B.J., Duffy, J.E., Gonzalez, A., Hooper, D.U., Perrings, C., Venail, P., Narwani, A., Mace, G.M., Tilman, D., Wardle, D.A., Kinzig, A.P., Daily, G.C., Loreau, M., Grace, J.B., Larigauderie, A., Srivastava, D.S. & Naeem, S. 2012. Biodiversity loss and its impact on humanity. Nature, 4867401, 59-67.

Caro, T., Dobson, A., Marshall, A.J. & Peres, C.A. 2014. Compromise solutions between conservation and road building in the tropics. Current Biology, 24(16), R722-R725.

Carreras Gamarra, M.J., Lassoie, J.P. & Milder, J. 2018. Accounting for no net loss: A critical assessment of biodiversity offsetting metrics and methods. Journal of Environmental Management, 220, 36-43.

Cashmore, M. 2004. The role of science in environmental impact assessment: Process and procedure versus purpose in the development of theory. Environmental Impact Assessment Review, 24(4), 403-426.

CASP (Critical Appraisal Skills Programme) 2018. CASP Checklists. Available at: https://casp-uk.net/casp-tools-checklists/ [Accessed 15 September 2023].

Catterall, C.P., Freeman, A.N., Kanowski, J. & Freebody, K. 2016. Can active restoration of tropical rainforest rescue biodiversity? A case with bird community indicators. Biological Conservation, 191, 142-149.

Chapman, C.A. & Lambert, J.E. 2000. Habitat alteration and the conservation of African primates: Case study of Kibale National Park, Uganda. American Journal of Primatology, 50(3), 169-185.

Chapman, C.A., Gautier-Hion, A., Oates, J.F. & Onderdonk, D.A. 1999. African primate communities: Determinants of structure and threats to survival. In J.G. Fleagle, C. Janson & K.E. Reed (Eds.), Primate Communities (pp. 1-37). Cambridge University Press.

Chapman, C.A., Struhsaker, T.T. & Lambert, J.E. 2005. Thirty years of research in Kibale National Park, Uganda, reveals a complex picture for conservation. International Journal of Primatology, 26(3), 539-555.

Chapman, C.A., Struhsaker, T.T., Skorupa, J.P., Snaith, T.V. & Rothman, J.M. 2010. Understanding long-term primate community dynamics: Implications of forest change. Ecological Applications, 20(1), 179-191.

Chapman, C.A., Struhsaker, T.T., Skorupa, J.P., Snaith, T.V. & Rothman, J.M. 2016. Understanding long-term primate community dynamics: Implications of forest change. Ecological Applications, 20(1), 179-191.

Chazdon, R.L. 2014. Second Growth: The Promise of Tropical Forest Regeneration in an Age of Deforestation. University of Chicago Press.

Chazdon, R.L. & Guariguata, M.R. 2016. Natural regeneration as a tool for large-scale forest restoration in the tropics: Prospects and challenges. Biotropica, 48(6), 716-730.

Chazdon, R.L., Peres, C.A., Dent, D., Sheil, D., Lugo, A.E., Lamb, D., Stork, N.E. & Miller, S.E. 2009. The potential for species conservation in tropical secondary forests. Conservation Biology, 23(6), 1406-1417.

Chen, J., Franklin, J.F. & Spies, T.A. 1995. Growing-season microclimatic gradients from clearcut edges into old-growth Douglas-fir forests. Ecological Applications, 5(1), 74-86.

Chomitz, K.M. & Gray, D.A. 1996. Roads, land use, and deforestation: A spatial model applied to Belize. The World Bank Economic Review, 10(3), 487-512.

Clare, S., Krogman, N., Foote, L. & Lemphers, N. 2011. Where is the avoidance in the implementation of wetland law and policy? Wetlands Ecology and Management, 19(2), 165-182.

Clarke, K.R. 1993. Non-parametric multivariate analyses of changes in community structure. Australian Journal of Ecology, 18(1), 117-143.

Clarke, K.R. & Warwick, R.M. 2001. Change in Marine Communities: An Approach to Statistical Analysis and Interpretation (2nd ed.). PRIMER-E, Plymouth, UK.

Clevenger, A.P. & Huijser, M.P. 2011. Wildlife Crossing Structure Handbook, Design and Evaluation in North America. Federal Highway Administration, Washington, DC.

Clevenger, A.P. & Waltho, N. 2005. Performance indices to identify attributes of highway crossing structures facilitating movement of large mammals. Biological Conservation, 121(3), 453-464.

Clevenger, A.P., Chruszcz, B. & Gunson, K.E. 2003. Spatial patterns and factors influencing small vertebrate fauna road-kill aggregations. Biological Conservation, 109(1), 15-26.

Coffin, A.W. 2007. From roadkill to road ecology: A review of the ecological effects of roads. Journal of Transport Geography, 15(5), 396-406.

Collinson, W., Davies-Mostert, H., Roxburgh, L. & van der Ree, R. 2019. Status of road ecology research in Africa: Do we understand the impacts of roads, and how to successfully mitigate them? Frontiers in Ecology and Evolution, 7, 479.

Commonwealth of Australia 2021. Environment Protection and Biodiversity Conservation Act 1999. Commonwealth of Australia, Canberra.

Condit, R. 1998. Tropical Forest Census Plots: Methods and Results from Barro Colorado Island, Panama and a Comparison with Other Plots. Springer-Verlag, Berlin.

Congalton, R.G. 1991. A review of assessing the accuracy of classifications of remotely sensed data. Remote Sensing of Environment, 37(1), 35-46.

Congalton, R.G. & Green, K. 2009. Assessing the Accuracy of Remotely Sensed Data: Principles and Practices. CRC Press, Boca Raton.

Congalton, R.G. & Green, K. 2019. Assessing the Accuracy of Remotely Sensed Data: Principles and Practices. CRC Press, Boca Raton.

Coppin, P., Jonckheere, I., Nackaerts, K., Muys, B. & Lambin, E. 2004. Digital change detection methods in ecosystem monitoring: A review. International Journal of Remote Sensing, 25(9), 1565-1596.

Corlett, R.T. 2016. Plant diversity in a changing world: Status, trends, and conservation needs. Plant Diversity, 38(1), 10-16.

Creswell, J.W. & Creswell, J.D. 2018. Research Design: Qualitative, Quantitative, and Mixed Methods Approaches (5th ed.). Sage Publications.

Creswell, J.W. & Plano Clark, V.L. 2017. Designing and Conducting Mixed Methods Research (3rd ed.). Sage Publications.

Critchlow, R., Plumptre, A.J., Driciru, M., Rwetsiba, A., Stokes, E.J., Tumwesigye, C., Wanyama, F. & Beale, C.M. 2015. Spatiotemporal trends of illegal activities from ranger-collected data in a Ugandan national park. Conservation Biology, 29(5), 1458-1470.

Crouzeilles, R., Curran, M., Ferreira, M.S., Lindenmayer, D.B., Grelle, C.E. & Rey Benayas, J.M. 2016. A global meta-analysis on the ecological drivers of forest restoration success. Nature Communications, 7(1), 1-8.

D'Angelo, S.A., Andrade, A.C., Laurance, S.G., Laurance, W.F. & Mesquita, R.C. 2004. Inferred causes of tree mortality in fragmented and intact Amazonian forests. Journal of Tropical Ecology, 20(2), 243-246.

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

Damgaard, C. 2019. A critique of the space-for-time substitution practice in community ecology. Trends in Ecology & Evolution, 34(5), 416-421.

Daniel, T.C., Muhar, A., Arnberger, A., Aznar, O., Boyd, J.W., Chan, K.M., Costanza, R., Elmqvist, T., Flint, C.G., Gobster, P.H., Grêt-Regamey, A., Lave, R., Muhar, S., Penker, M., Ribe, R.G., Schauppenlehner, T., Sikor, T., Soloviy, I., Spierenburg, M., Taczanowska, K., Tam, J. & von der Dunk, A. 2012. Contributions of cultural services to the ecosystem services agenda. Proceedings of the National Academy of Sciences, 109(23), 8812-8819.

Daw, T.M., Coirolo, C., Bueno, P., Rahman, M.A., Djenontin, I.N., Kebede, B.T., Seara, T., Haruna, M., Kassam, L., Gonzalez, A.M., Arce-Ibarra, M., Montero, A., Thomas, J.T. & Leiva, F. 2016. Evaluating taboo trade-offs in ecosystems services and human well-being. Proceedings of the National Academy of Sciences, 113(23), 6406-6411.

Deljouei, A., Abdi, E., Marcantonio, M., Majnounian, B., Amici, V. & Sohrabi, H. 2017. The impact of road disturbance on vegetation and soil properties in a beech stand, Hyrcanian forest. European Journal of Forest Research, 136(4), 565-579.

Díaz, S., Lavorel, S., de Bello, F., Quétier, F., Grigulis, K. & Robson, T.M. 2007. Incorporating plant functional diversity effects in ecosystem service assessments. Proceedings of the National Academy of Sciences, 104(52), 20684-20689.

Didham, R.K. 1998. Altered leaf-litter decomposition rates in tropical forest fragments. Oecologia, 116(3), 397-406.

Didham, R.K. & Lawton, J.H. 1999. Edge structure determines the magnitude of changes in microclimate and vegetation structure in tropical forest fragments. Biotropica, 31(1), 17-30.

Edwards, D.P., Socolar, J.B., Mills, S.C., Burivalova, Z., Koh, L.P. & Wilcove, D.S. 2014. Conservation of tropical forests in the Anthropocene. Current Biology, 24(16), R1008-R1020.

Eigenbrod, F., Hecnar, S.J. & Fahrig, L. 2009. Quantifying the road-effect zone: Threshold effects of a motorway on anuran populations in Ontario, Canada. Ecology and Society, 14(1), 24.

Eldegard, K., Totland, Ø. & Moe, S.R. 2015. Edge effects on plant communities along power line clearings. Journal of Applied Ecology, 52(4), 871-880.

Elzinga, C.L., Salzer, D.W., Willoughby, J.W. & Gibbs, J.P. 2015. Monitoring Plant and Animal Populations: A Handbook for Field Biologists. John Wiley & Sons.

Epps, C.W., Palsbøll, P.J., Wehausen, J.D., Roderick, G.K., Ramey, R.R. & McCullough, D.R. 2005. Highways block gene flow and cause a rapid decline in genetic diversity of desert bighorn sheep. Ecology Letters, 8(10), 1029-1038.

Evans, M.C., Possingham, H.P. & Wilson, K.A. 2011. What to do in the face of multiple threats? Incorporating dependencies within a return on investment framework for conservation. Diversity and Distributions, 17(3), 437-450.

Ewers, R.M. & Didham, R.K. 2006. Confounding factors in the detection of species responses to habitat fragmentation. Biological Reviews, 81(1), 117-142.

Ezzine-de-Blas, D., Wunder, S., Ruiz-Pérez, M. & del Pilar Moreno-Sanchez, R. 2016. Global patterns in the implementation of payments for environmental services. PLoS One, 11(3), e0149847.

Fahrig, L. 2003. Effects of habitat fragmentation on biodiversity. Annual Review of Ecology, Evolution, and Systematics, 34(1), 487-515.

Fahrig, L. & Rytwinski, T. 2009. Effects of roads on animal abundance: An empirical review and synthesis. Ecology and Society, 14(1), 21.

Faith, D.P., Minchin, P.R. & Belbin, L. 1987. Compositional dissimilarity as a robust measure of ecological distance. Vegetatio, 69(1-3), 57-68.

Fearnside, P.M. 2015. Highway construction as a force in destruction of the Amazon forest. In Handbook of Road Ecology (pp. 414-424). John Wiley & Sons, Chichester.

Fearnside, P.M. & Graça, P.M.L.A. 2006. BR-319: Brazil's Manaus-Porto Velho highway and the potential impact of linking the arc of deforestation to central Amazonia. Environmental Management, 38(5), 705-716.

Ferraro, P.J. & Pattanayak, S.K. 2006. Money for nothing? A call for empirical evaluation of biodiversity conservation investments. PLoS Biology, 4(4), e105.

Fischer, J. & Lindenmayer, D.B. 2007. Landscape modification and habitat fragmentation: A synthesis. Global Ecology and Biogeography, 16(3), 265-280.

Fisher, B., Turner, R.K. & Morling, P. 2009. Defining and classifying ecosystem services for decision making. Ecological Economics, 68(3), 643-653.

Fletcher, R.J., Didham, R.K., Banks-Leite, C., Barlow, J., Ewers, R.M., Rosindell, J., Holt, R.D., Gonzalez, A., Pardini, R., Damschen, E.I., Melo, F.P., Ries, L., Prevedello, J.A., Tscharntke, T., Laurance, W.F., Lovejoy, T. & Haddad, N.M. 2018. Is habitat fragmentation good for biodiversity? Biological Conservation, 226, 9-15.

Flory, S.L. & Clay, K. 2009. Invasive plant removal method determines native plant community responses. Journal of Applied Ecology, 46(2), 434-442.

Foody, G.M. 2002. Status of land cover classification accuracy assessment. Remote Sensing of Environment, 80(1), 185-201.

Forman, R.T. 1995. Land Mosaics: The Ecology of Landscapes and Regions. Cambridge University Press, Cambridge.

Forman, R.T. & Alexander, L.E. 1998. Roads and their major ecological effects. Annual Review of Ecology and Systematics, 29(1), 207-231.

Forman, R.T. & Deblinger, R.D. 2000. The ecological road-effect zone of a Massachusetts (USA) suburban highway. Conservation Biology, 14(1), 36-46.

Forman, R.T. & Godron, M. 1986. Landscape Ecology. John Wiley & Sons, New York.

Forman, R.T., Sperling, D., Bissonette, J.A., Clevenger, A.P., Cutshall, C.D., Dale, V.H., Fahrig, L., France, R.L., Heanue, K., Goldman, C.R. & Jones, J. 2003. Road Ecology: Science and Solutions. Island Press, Washington, D.C.

Frankham, R. 2005. Genetics and extinction. Biological Conservation, 126(2), 131-140.

Fuentes-Montemayor, E., Goulson, D., Cavin, L., Wallace, J.M. & Park, K.J. 2013. Fragmented woodlands in agricultural landscapes: The influence of woodland character and landscape context on bats and their insect prey. Agriculture, Ecosystems & Environment, 172, 6-15.

Fuentes-Montemayor, E., Ferryman, M., Watts, K., Macgregor, N.A., Lopez-Gallego, Z., Pahlen, J., Procter, D.S., Irvine, K.N. & Park, K.J. 2020. Small mammal responses to long-term large-scale woodland creation: The influence of local and landscape-level attributes. EcologicalApplications, 30(2), e02028.

Gardner, T.A., Barlow, J., Chazdon, R., Ewers, R.M., Harvey, C.A., Peres, C.A. & Sodhi, N.S. 2009. Prospects for tropical forest biodiversity in a human-modified world. Ecology Letters, 12(6), 561-582.

Gardner, T.A., von Hase, A., Brownlie, S., Ekstrom, J.M., Pilgrim, J.D., Savy, C.E., Stephens, R.T., Treweek, J., Ussher, G.T., Ward, G. & Ten Kate, K. 2013. Biodiversity offsets and the challenge of achieving no net loss. Conservation Biology, 27(6), 1254-1264.

Gelbard, J.L. & Belnap, J. 2003. Roads as conduits for exotic plant invasions in a semiarid landscape. Conservation Biology, 17(2), 420-432.

Geldmann, J., Barnes, M., Coad, L., Craigie, I.D., Hockings, M. & Burgess, N.D. 2013. Effectiveness of terrestrial protected areas in reducing habitat loss and population declines. Biological Conservation, 161, 230-238.

Gelot, F. & Bigard, C. 2021. Biodiversity offsetting and payments for environmental services: Clarifications and implications for conservation policy in France. Ecosystem Services, 52, 101359.

Gelot, M. & Bigard, C. 2021. Ecological mitigation hierarchy and biodiversity offsets revisited through habitat connectivity modelling. Journal of Environmental Management, 277, 111437.

Gentry, A.H. 1988. Changes in plant community diversity and floristic composition on environmental and geographical gradients. Annals of the Missouri Botanical Garden, 75(1), 1-34.

Gerwing, J.J., Schnitzer, S.A., Burnham, R.J., Bongers, F., Chave, J., DeWalt, S.J., Ewango, C.E., Foster, R., Kenfack, D., Martínez-Ramos, M., Parren, M., Parthasarathy, N., Pérez-Salicrup, D.R., Putz, F.E. & Thomas, D.W. 2006. A standard protocol for liana censuses. Biotropica, 38(2), 256-261.

Gibbs, H.K., Brown, S., Niles, J.O. & Foley, J.A. 2007. Monitoring and estimating tropical forest carbon stocks: Making REDD a reality. Environmental Research Letters, 2(4), 045023.

Goldberg, T.L. & Wrangham, R.W. 1997. Genetic correlates of social behaviour in wild chimpanzees: Evidence from mitochondrial DNA. Animal Behaviour, 54(3), 559-570.

Goldingay, R.L. & Taylor, B.D. 2009. Gliding performance and its relevance to gap crossing by the squirrel glider (Petaurus norfolcensis). Australian Journal of Zoology, 57(2), 99-104.

Goldstein, J.H., Caldarone, G., Duarte, T.K., Ennaanay, D., Hannahs, N., Mendoza, G., Polasky, S., Wolny, S. & Daily, G.C. 2012. Integrating ecosystem-service tradeoffs into land-use decisions. Proceedings of the National Academy of Sciences, 109(19), 7565-7570.

Goosem, M. 2007. Fragmentation impacts caused by roads through rainforests. Current Science, 93(11), 1587-1595.

Gordon, A., Langford, W.T., Todd, J.A., White, M.D., Mullerworth, D.W. & Bekessy, S.A. 2011. Assessing the impacts of biodiversity offset policies. Environmental Modelling & Software, 26(12), 1481-1488.

Government of Canada 2019. Impact Assessment Act. S.C. 2019, c. 28, s. 1.

Government of Uganda 2015. Uganda's Intended Nationally Determined Contribution (INDC). Ministry of Water and Environment, Kampala.

Government of Uganda 2019. The National Environment Act, 2019. Uganda Gazette No. 21, Volume CXII.

Government of Uganda 2020. The National Environment (Environmental and Social Assessment) Regulations, 2020. Statutory Instrument 2020 No. 22.

Greene, J.C., Caracelli, V.J. & Graham, W.F. 1989. Toward a conceptual framework for mixed-method evaluation designs. Educational Evaluation and Policy Analysis, 11(3), 255-274.

Guariguata, M.R. & Ostertag, R. 2001. Neotropical secondary forest succession: Changes in structural and functional characteristics. Forest Ecology and Management, 148(1-3), 185-206.

Gullison, R.E., Frumhoff, P.C., Canadell, J.G., Field, C.B., Nepstad, D.C., Hayhoe, K., Avissar, R., Curran, L.M., Friedlingstein, P., Jones, C.D. & Nobre, C. 2007. Tropical forests and climate policy. Science, 3165827, 985-986.

Gustafson, E.J. 1998. Quantifying landscape spatial pattern: What is the state of the art? Ecosystems, 1(2), 143-156.

Hamilton, A. 1991. A Field Guide to Ugandan Forest Trees. Makerere University, Kampala.

Hanski, I. & Ovaskainen, O. 2000. The metapopulation capacity of a fragmented landscape. Nature, 4046779, 755-758.

Harper, K.A., MacDonald, S.E., Burton, P.J., Chen, J., Brosofske, K.D., Saunders, S.C., Euskirchen, E.S., Roberts, D., Jaiteh, M.S. & Esseen, P.A. 2005. Edge influence on forest structure and composition in fragmented landscapes. Conservation Biology, 19(3), 768-782.

Hartter, J. & Goldman, A. 2011. Local responses to a forest park in western Uganda: Alternate narratives on fortress conservation. Oryx, 45(1), 60-68.

Hartter, J., Ryan, S.J., MacKenzie, C.A., Parker, J.N. & Strasser, C.A. 2015. Spatially explicit data: Stewardship and ethical challenges in science. PLoS Biology, 13(9), e1002275.

Hickey, J.R., Basabose, A., Gilardi, K.V., Greer, D., Nampindo, S., Robbins, M.M. & Stoinski, T.S. 2019. Gorilla census 2015-2016: Virunga Massif chimpanzee population assessment. In A. Plumptre, J.R. Hickey, A. Kirkby & E. Williamson (Eds.), Surveys of the Chimpanzee Population in Uganda (pp. 45-67). Wildlife Conservation Society, Kampala.

Hickey, S. & Mohan, G. 2005. Relocating participation within a radical politics of development. Development and Change, 36(2), 237-262.

Hilty, J.A., Lidicker Jr, W.Z. & Merenlender, A.M. 2006. Corridor Ecology: The Science and Practice of Linking Landscapes for Biodiversity Conservation. Island Press, Washington, DC.

Holl, K.D. & Aide, T.M. 2011. When and where to actively restore ecosystems? Forest Ecology and Management, 261(10), 1558-1563.

Howe, C., Suich, H., Vira, B. & Mace, G.M. 2014. Creating win-wins from trade-offs? Ecosystem services for human well-being: A meta-analysis of ecosystem service trade-offs and synergies in the real world. Global Environmental Change, 28, 263-275.

Humle, T., Maisels, F., Oates, J.F., Plumptre, A. & Williamson, E.A. 2016. Pan troglodytes (errata version published in 2018). The IUCN Red List of Threatened Species 2016: e.T15933A129038584. https://dx.doi.org/10.2305/IUCN.UK.2016-2.RLTS.T15933A17964454.en

Hurlbert, S.H. 1984. Pseudoreplication and the design of ecological field experiments. Ecological Monographs, 54(2), 187-211.

Ibisch, P.L., Hoffmann, M.T., Kreft, S., Pe'er, G., Kati, V., Biber-Freudenberger, L., DellaSala, D.A., Vale, M.M., Hobson, P.R. & Selva, N. 2016. A global map of roadless areas and their conservation status. Science, 3546318, 1423-1427.

Jaeger, J.A., Bowman, J., Brennan, J., Fahrig, L., Bert, D., Bouchard, J., Charbonneau, N., Frank, K., Gruber, B. & von Toschanowitz, K.T. 2005. Predicting when animal populations are at risk from roads: An interactive model of road avoidance behavior. Ecological Modelling, 185(2-4), 329-348.

Jensen, J.R. 2016. Introductory Digital Image Processing: A Remote Sensing Perspective (4th ed.). Pearson Education, Glendale, CA.

Jick, T.D. 1979. Mixing qualitative and quantitative methods: Triangulation in action. Administrative Science Quarterly, 24(4), 602-611.

Johnson, R.B. & Onwuegbuzie, A.J. 2004. Mixed methods research: A research paradigm whose time has come. Educational Researcher, 33(7), 14-26.

Joly, M., Bertrand, P., Gbangou, R.Y., White, M.C., Dubé, J. & Lavoie, C. 2011. Paving the way for invasive species: Road type and the spread of common ragweed (Ambrosia artemisiifolia). Environmental Management, 48(3), 514-522.

Kapos, V. 1989. Effects of isolation on the water status of forest patches in the Brazilian Amazon. Journal of Tropical Ecology, 5(2), 173-185.

Katende, A.B., Birnie, A. & Tengnäs, B. 1995. Useful Trees and Shrubs for Uganda: Identification, Propagation and Management for Agricultural and Pastoral Communities. Regional Soil Conservation Unit, Nairobi.

Kennedy, R.E., Yang, Z. & Cohen, W.B. 2010. Detecting trends in forest disturbance and recovery using yearly Landsat time series: 1. LandTrendr—Temporal segmentation algorithms. Remote Sensing of Environment, 114(12), 2897-2910.

Kent, M. 2011. Vegetation Description and Data Analysis: A Practical Approach (2nd ed.). Wiley-Blackwell, Chichester, UK.

Kirkby, A., Plumptre, A.J., Wilson, E. & Mugume, S. 2018. Kibale National Park Biodiversity Assessment Survey: Methods and Preliminary Results. Wildlife Conservation Society, Albertine Rift Programme, Kampala, Uganda.

Kisangala, K., Majule, A. & Ngaga, Y. 2019. Effectiveness of environmental impact assessment follow-up in Tanzania: A critical review. Journal of Environmental Assessment Policy and Management, 21(2), 1950007.

Kleinschroth, F., Healey, J.R., Sist, P., Mortier, F. & Gourlet-Fleury, S. 2016. How persistent are the impacts of logging roads on Central African forest vegetation? Journal of Applied Ecology, 53(4), 1127-1137.

Kroll, C., Warchold, A. & Pradhan, P. 2019. Sustainable Development Goals (SDGs): Are we successful in turning trade-offs into synergies? Palgrave Communications, 5(1), 1-11.

Kruskal, J.B. 1964. Multidimensional scaling by optimizing goodness of fit to a nonmetric hypothesis. Psychometrika, 29(1), 1-27.

Kuussaari, M., Bommarco, R., Heikkinen, R.K., Helm, A., Krauss, J., Lindborg, R., Öckinger, E., Pärtel, M., Pino, J., Rodà, F., Stefanescu, C., Teder, T., Zobel, M. & Steffan-Dewenter, I. 2009. Extinction debt: A challenge for biodiversity conservation. Trends in Ecology & Evolution, 24(10), 564-571.

Laliberté, E., Wells, J.A., DeClerck, F., Metcalfe, D.J., Catterall, C.P., Queiroz, C., Aubin, I., Bonser, S.P., Ding, Y., Fraterrigo, J.M., McNamara, S., Morgan, J.W., Merlos, D.S., Vesk, P.A. & Mayfield, M.M. 2010. Land-use intensification reduces functional redundancy and response diversity in plant communities. Ecology Letters, 13(1), 76-86.

Langner, A. & Siegert, F. 2009. Spatiotemporal fire occurrence in Borneo over a period of 10 years. Global Change Biology, 15(1), 48-62.

Laurance, W.F. & Goosem, M. 2008. Impacts of habitat fragmentation and linear clearings on Australian rainforest. In Living in a Dynamic Tropical Forest Landscape (pp. 295-311). Blackwell Publishing.

Laurance, W.F. & Williamson, G.B. 2001. Positive feedbacks among forest fragmentation, drought, and climate change in the Amazon. Conservation Biology, 15(6), 1529-1535.

Laurance, W.F., Lovejoy, T.E., Vasconcelos, H.L., Bruna, E.M., Didham, R.K., Stouffer, P.C., Gascon, C., Bierregaard, R.O., Laurance, S.G. & Sampaio, E. 2002. Ecosystem decay of Amazonian forest fragments: A 22-year investigation. Conservation Biology, 16(3), 605-618.

Laurance, W.F., Nascimento, H.E., Laurance, S.G., Andrade, A.C., Fearnside, P.M., Ribeiro, J.E. & Capretz, R.L. 2006. Rain forest fragmentation and the proliferation of successional trees. Ecology, 87(2), 469-482.

Laurance, W.F., Nascimento, H.E., Laurance, S.G., Andrade, A., Ewers, R.M., Harms, K.E., Luizão, R.C. & Ribeiro, J.E. 2007. Habitat fragmentation, variable edge effects, and the landscape-divergence hypothesis. PLoS One, 2(10), e1017.

Laurance, W.F., Nascimento, H.E.M., Laurance, S.G., Andrade, A., Ribeiro, J.E.L.S., Giraldo, J.P., Lovejoy, T.E., Condit, R., Chave, J., Harms, K.E. & D'Angelo, S. 2006. Rapid decay of tree-community composition in Amazonian forest fragments. Proceedings of the National Academy of Sciences, 103(50), 19010-19014.

Laurance, W.F., Camargo, J.L., Luizão, R.C., Laurance, S.G., Pimm, S.L., Bruna, E.M., Stouffer, P.C., Williamson, G.B., Benítez-Malvido, J., Vasconcelos, H.L., Van Houtan, K.S., Zartman, C.E., Boyle, S.A., Didham, R.K., Andrade, A. & Lovejoy, T.E. 2011. The fate of Amazonian forest fragments: A 32-year investigation. Biological Conservation, 144(1), 56-67.

Legendre, P. & Legendre, L. 2012. Numerical Ecology (3rd ed.). Elsevier, Amsterdam.

Lillesand, T., Kiefer, R.W. & Chipman, J. 2015. Remote Sensing and Image Interpretation (7th ed.). John Wiley & Sons, Hoboken, NJ.

Lindenmayer, D.B. & Likens, G.E. 2010. The science and application of ecological monitoring. Biological Conservation, 143(6), 1317-1328.

Lindsey, P.A., Romañach, S.S., Matema, S., Matema, C., Mupamhadzi, I. & Muvengwi, J. 2011. Dynamics and underlying causes of illegal bushmeat trade in Zimbabwe. Oryx, 45(1), 84-95.

Lu, D., Mausel, P., Brondízio, E. & Moran, E. 2004. Change detection techniques. International Journal of Remote Sensing, 25(12), 2365-2401.

Malcolm, J.R. 1994. Edge effects in central Amazonian forest fragments. Ecology, 75(8), 2438-2445.

Mandle, L. & Tallis, H. 2016. Spatial service flow: An application to the ecosystem service value of tropical forest protection. Biological Conservation, 196, 133-141.

Maron, M., Hobbs, R.J., Moilanen, A., Matthews, J.W., Christie, K., Gardner, T.A., Keith, D.A., Lindenmayer, D.B. & McAlpine, C.A. 2012. Faustian bargains? Restoration realities in the context of biodiversity offset policies. Biological Conservation, 155, 141-148.

Maron, M., Gordon, A., Mackey, B.G., Possingham, H.P. & Watson, J.E. 2016. Stop misuse of biodiversity offsets. Nature, 5237561, 401-403.

Marvin, D.C., Asner, G.P., Knapp, D.E., Anderson, C.B., Martin, R.E., Sinca, F. & Tupayachi, R. 2014. Amazonian landscapes and the bias in field studies of forest structure and biomass. Proceedings of the National Academy of Sciences, 111(48), E5224-E5232.

Mas, J.F. 1999. Monitoring land-cover changes: A comparison of change detection techniques. International Journal of Remote Sensing, 20(1), 139-152.

Maxwell, J.A. 2012. Qualitative Research Design: An Interactive Approach (3rd ed.). Sage Publications.

McCune, B. & Grace, J.B. 2002. Analysis of Ecological Communities. MjM Software Design, Gleneden Beach, Oregon.

McGarigal, K. & Marks, B.J. 1995. FRAGSTATS: Spatial Pattern Analysis Program for Quantifying Landscape Structure. USDA Forest Service General Technical Report PNW-GTR-351.

McRae, B.H., Dickson, B.G., Keitt, T.H. & Shah, V.B. 2008. Using circuit theory to model connectivity in ecology, evolution, and conservation. Ecology, 89(10), 2712-2724.

Ministry of Works and Transport 2018. National Transport Master Plan 2018-2040. Republic of Uganda, Kampala.

Morgan, R.K. 2012. Environmental impact assessment: The state of the art. Impact Assessment and Project Appraisal, 30(1), 5-14.

Morrison-Saunders, A. & Pope, J. 2013. Conceptualising and managing consistency expectations in impact assessment. Environmental Impact Assessment Review, 38, 74-81.

Morrison-Saunders, A., Baker, J. & Arts, J. 2007. Lessons from practice: Towards successful follow-up. Impact Assessment and Project Appraisal, 21(1), 43-56.

Mortensen, D.A., Rauschert, E.S., Nord, A.N. & Jones, B.P. 2009. Forest roads facilitate the spread of invasive plants. Invasive Plant Science and Management, 2(3), 191-199.

Murcia, C. 1995. Edge effects in fragmented forests: Implications for conservation. Trends in Ecology & Evolution, 10(2), 58-62.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., Da Fonseca, G.A. & Kent, J. 2000. Biodiversity hotspots for conservation priorities. Nature, 4036772, 853-858.

Nagendra, H., Lucas, R., Honrado, J.P., Jongman, R.H., Tarantino, C., Adamo, M. & Mairota, P. 2013. Remote sensing for conservation monitoring: Assessing protected areas, habitat extent, habitat condition, species diversity, and threats. Ecological Indicators, 33, 45-59.

Naidoo, R., Balmford, A., Ferraro, P.J., Polasky, S., Ricketts, T.H. & Rouget, M. 2006. Integrating economic costs into conservation planning. Trends in Ecology & Evolution, 21(12), 681-687.

Naidoo, R., Balmford, A., Costanza, R., Fisher, B., Green, R.E., Lehner, B., Malcolm, T.R. & Ricketts, T.H. 2008. Global mapping of ecosystem services and conservation priorities. Proceedings of the National Academy of Sciences, 105(28), 9495-9500.

Naughton-Treves, L. & Treves, A. 2005. Socio-ecological factors shaping local support for wildlife: Crop-raiding by elephants and other wildlife in Africa. In R. Woodroffe, S. Thirgood & A. Rabinowitz (Eds.), People and Wildlife, Conflict or Co-existence? (pp. 252-277). Cambridge University Press.

NEMA (National Environment Management Authority) 1998. The National Environment (Environmental Impact Assessment) Regulations, 1998. Kampala, Uganda.

NEMA (National Environment Management Authority) 2016. State of Environment Report for Uganda 2014/2015. National Environment Management Authority, Kampala.

NEMA (National Environment Management Authority) 2022. National Guidelines for Biodiversity and Social Offsets. National Environment Management Authority, Kampala, Uganda.

Oksanen, L. 2001. Logic of experiments in ecology: Is pseudoreplication a pseudoissue? Oikos, 94(1), 27-38.

Olofsson, P., Foody, G.M., Herold, M., Stehman, S.V., Woodcock, C.E. & Wulder, M.A. 2014. Good practices for estimating area and assessing accuracy of land change. Remote Sensing of Environment, 148, 42-57.

Pal, M. 2005. Random forest classifier for remote sensing classification. International Journal of Remote Sensing, 26(1), 217-222.

Patton, M.Q. 2002. Qualitative Research and Evaluation Methods (3rd ed.). Sage Publications.

Pereira, H.M., Ferrier, S., Walters, M., Geller, G.N., Jongman, R.H., Scholes, R.J., Bruford, M.W., Brummitt, N., Butchart, S.H., Cardoso, A.C., Coops, N.C., Dulloo, E., Faith, D.P., Freyhof, J., Gregory, R.D., Heip, C., Höft, R., Hurtt, G., Jetz, W., Karp, D.S., McGeoch, M.A., Obura, D., Onoda, Y., Pettorelli, N., Reyers, B., Sayre, R., Scharlemann, J.P., Stuart, S.N., Turak, E., Walpole, M. & Wegmann, M. 2013. Essential biodiversity variables. Science, 3396117, 277-278.

Peres, C.A., Barlow, J. & Laurance, W.F. 2006. Detecting anthropogenic disturbance in tropical forests. Trends in Ecology & Evolution, 21(5), 227-229.

Pfaff, A.S. 1999. What drives deforestation in the Brazilian Amazon? Evidence from satellite and socioeconomic data. Journal of Environmental Economics and Management, 37(1), 26-43.

Pfeifer, M., Lefebvre, V., Peres, C.A., Banks-Leite, C., Wearn, O.R., Marsh, C.J., Butchart, S.H., Arroyo-Rodríguez, V., Barlow, J., Cerezo, A., Cisneros, L., D'Cruze, N., Faria, D., Hadley, A., Harris, S.M., Klingbeil, B.T., Kormann, U., Lens, L., Medina-Rangel, G.F., Morante-Filho, J.C., Olivier, P., Peters, S.L., Pidgeon, A., Ribeiro, D.B., Scherber, C., Schneider-Maunoury, L., Struebig, M., Urbina-Cardona, N., Watling, J.I., Willig, M.R., Wood, E.M. & Ewers, R.M. 2017. Creation of forest edges has a global impact on forest vertebrates. Nature, 5517679, 187-191.

Phillips, O.L., Martínez, R.V., Arroyo, L., Baker, T.R., Killeen, T., Lewis, S.L., Malhi, Y., Mendoza, A.M., Neill, D., Vargas, P.N., Alexiades, M., Cerón, C., Di Fiore, A., Erwin, T., Jardim, A., Palacios, W., Saldias, M. & Vinceti, B. 2003. Efficient plot-based floristic assessment of tropical forests. Journal of Tropical Ecology, 19(6), 629-645.

Phylip-Jones, J. & Fischer, T.B. 2015. Strategic environmental assessment (SEA) for wind energy planning: Lessons from the United Kingdom and Germany. Environmental Impact Assessment Review, 50, 203-212.

Plumptre, A.J., Cox, D. & Mugume, S. 2003. The Status of Chimpanzees in Uganda. Wildlife Conservation Society, Albertine Rift Programme, Kampala.

Plumptre, A.J., Kayitare, A., Rainer, H., Gray, M., Munanura, I., Barakabuye, N., Asuma, S., Sivha, M. & Namara, A. 2004. The Socio-economic Status of People Living Near Protected Areas in the Central Albertine Rift. Wildlife Conservation Society, Albertine Rift Programme.

Plumptre, A.J., Behangana, M., Ndomba, E., Davenport, T., Kahindo, C., Kityo, R., Eilu, G., Ssegawa, P., Ewango, C., Meirte, D., Kahindo, C., Herremans, M., Peterhans, J.K., Pilgrima, J.D., Wilson, M., Languy, M. & Moyer, D. 2007. The biodiversity of the Albertine Rift. Biological Conservation, 134(2), 178-194.

Plumptre, A.J., Sterling, E.J. & Buckland, S.T. 2013. Primate census and survey techniques. In E.J. Sterling, N. Bynum & M.E. Blair (Eds.), Primate Ecology and Conservation: A Handbook of Techniques (pp. 10-26). Oxford University Press, Oxford.

Plumptre, A.J., Fuller, R.A., Rwetsiba, A., Wanyama, F., Kujirakwinja, D., Driciru, M., Nangendo, G., Watson, J.E. & Possingham, H.P. 2014. Efficiently targeting resources to deter illegal activities in protected areas. Journal of Applied Ecology, 51(3), 714-725.

Plumptre, A.J., Nixon, S., Caillaud, D., Hall, J.S., Hart, J.A., Nishuli, R. & Williamson, E.A. 2016. Status of Grauer's gorilla and chimpanzees in eastern Democratic Republic of Congo: Historical and current distribution and abundance. Primate Conservation, 30, 1-11.

Plumptre, A.J., Kirkby, A., Tushabe, H., Forrest, T., Davenport, T., Chapman, H. & Kasoma, P. 2019. Assessment of the Forest Biodiversity of Kasyoha-Kitomi, Bugoma, Kalinzu, and Kibale National Park. Uganda National Parks.

Plumptre, A.J., Kirkby, A., Dowhaniuk, N., Seguya, A., Critchlow, R., Nampindo, S., Mpeirwe, R., Aryamanya, I., Wanyama, F., Baranga, D., Akampulira, E., Mugerwa, B., Kayijamahe, C., Kigoolo, S., Omondi, P., Limo, S., Ayebare, S., Mugabe, H., Majaliwa, M. & McNeilage, A. 2019. Conservation Action Plan for Uganda's Chimpanzees 2018-2028. Wildlife Conservation Society, Kampala, Uganda.

Pomeroy, D. & Dranzoa, C. 1997. Methods of studying the distribution, diversity and abundance of birds in East Africa—Some quantitative approaches. African Journal of Ecology, 35(2), 110-123.

Pontius Jr, R.G., Shusas, E. & McEachern, M. 2004. Detecting important categorical land changes while accounting for persistence. Agriculture, Ecosystems & Environment, 101(2-3), 251-268.

Pontzer, H. & Wrangham, R.W. 2004. Climbing and the daily energy cost of locomotion in wild chimpanzees: Implications for hominoid locomotor evolution. Journal of Human Evolution, 46(3), 317-335.

Pope, J., Bond, A., Morrison-Saunders, A. & Retief, F. 2013. Advancing the theory and practice of impact assessment: Setting the research agenda. Environmental Impact Assessment Review, 41, 1-9.

Pütz, S., Groeneveld, J., Henle, K., Knogge, C., Martensen, A.C., Metz, M., Metzger, J.P., Ribeiro, M.C., de Paula, M.D. & Huth, A. 2014. Long-term carbon loss in fragmented Neotropical forests. Nature Communications, 5(1), 5037.

Rainey, H.J., Pollard, E.H., Dutson, G., Ekstrom, J.M., Livingstone, S.R., Temple, H.J. & Pilgrim, J.D. 2015. A review of corporate goals of No Net Loss and Net Positive Impact on biodiversity. Oryx, 49(2), 232-238.

Richards, P.W. 1996. The Tropical Rain Forest (2nd ed.). Cambridge University Press.

Ries, L., Fletcher Jr, R.J., Battin, J. & Sisk, T.D. 2004. Ecological responses to habitat edges: Mechanisms, models, and variability explained. Annual Review of Ecology, Evolution, and Systematics, 35, 491-522.

Rodriguez-Galiano, V.F., Ghimire, B., Rogan, J., Chica-Olmo, M. & Rigol-Sanchez, J.P. 2012. An assessment of the effectiveness of a random forest classifier for land-cover classification. ISPRS Journal of Photogrammetry and Remote Sensing, 67, 93-104.

Rosenzweig, M.L. 1995. Species Diversity in Space and Time. Cambridge University Press.

Roy, D.P., Wulder, M.A., Loveland, T.R., Woodcock, C.E., Allen, R.G., Anderson, M.C., Helder, D., Irons, J.R., Johnson, D.M., Kennedy, R., Scambos, T.A., Schaaf, C.B., Schott, J.R., Sheng, Y., Vermote, E.F., Belward, A.S., Bindschadler, R., Cohen, W.B., Gao, F., Hipple, J.D., Hostert, P., Huntington, J., Justice, C.O., Kilic, A., Kovalskyy, V., Lee, Z.P., Lymburner, L., Masek, J.G., McCorkel, J., Shuai, Y., Trezza, R., Vogelmann, J., Wynne, R.H. & Zhu, Z. 2014. Landsat-8: Science and product vision for terrestrial global change research. Remote Sensing of Environment, 145, 154-172.

Rudel, T.K., Coomes, O.T., Moran, E., Achard, F., Angelsen, A., Xu, J. & Lambin, E. 2005. Forest transitions: Towards a global understanding of land use change. Global Environmental Change, 15(1), 23-31.

Rytwinski, T. & Fahrig, L. 2012. Do species life history traits explain population responses to roads? A meta-analysis. Biological Conservation, 147(1), 87-98.

Santos, S.M., Carvalho, F. & Mira, A. 2011. How long do the dead survive on the road? Carcass persistence probability and implications for road-kill monitoring surveys. PLoS One, 6(9), e25383.

Sawaya, M.A., Clevenger, A.P. & Kalinowski, S.T. 2014. Demographic connectivity for ursid populations at wildlife crossing structures in Banff National Park. Conservation Biology, 28(4), 1089-1100.

Schnitzer, S.A., DeWalt, S.J. & Chave, J. 2006. Censusing and measuring lianas: A quantitative comparison of the common methods. Biotropica, 38(5), 581-591.

Schott, J.R., Salvaggio, C. & Volchok, W.J. 1988. Radiometric scene normalization using pseudoinvariant features. Remote Sensing of Environment, 26(1), 1-16.

Secretariat of the Convention on Biological Diversity 2020. Global Biodiversity Outlook 5. Montreal.

Secretariat of the Convention on Biological Diversity 2022. Kunming-Montreal Global Biodiversity Framework. CBD/COP/DEC/15/4.

Shadish, W.R., Cook, T.D. & Campbell, D.T. 2002. Experimental and Quasi-Experimental Designs for Generalized Causal Inference. Houghton Mifflin, Boston.

Singh, A. 1989. Review article digital change detection techniques using remotely-sensed data. International Journal of Remote Sensing, 10(6), 989-1003.

Sloan, S., Campbell, M.J., Alamgir, M., Collier-Baker, E., Nowak, M.G., Usher, G. & Laurance, W.F. 2018. Infrastructure development and contested forest governance threaten the Leuser Ecosystem, Indonesia. Land Use Policy, 77, 298-309.

Soanes, K., Lobo, M.C., Vesk, P.A., McCarthy, M.A., Moore, J.L. & van der Ree, R. 2013. Movement re-established but not restored: Inferring the effectiveness of road-crossing mitigation for a gliding mammal by monitoring use. Biological Conservation, 159, 434-441.

Soares-Filho, B.S., Nepstad, D.C., Curran, L.M., Cerqueira, G.C., Garcia, R.A., Ramos, C.A., Voll, E., McDonald, A., Lefebvre, P. & Schlesinger, P. 2006. Modelling conservation in the Amazon basin. Nature, 4407083, 520-523.

Srivastava, D.S., Cadotte, M.W., MacDonald, A.A.M., Marushia, R.G. & Mirotchnick, N. 2012. Phylogenetic diversity and the functioning of ecosystems. Ecology Letters, 15(7), 637-648.

Stafford-Smith, M., Griggs, D., Gaffney, O., Ullah, F., Reyers, B., Kanie, N., Stigson, B., Shrivastava, P., Leach, M. & O'Connell, D. 2017. Integration: The key to implementing the Sustainable Development Goals. Sustainability Science, 12(6), 911-919.

Stehman, S.V. & Czaplewski, R.L. 1998. Design and analysis for thematic map accuracy assessment: Fundamental principles. Remote Sensing of Environment, 64(3), 331-344.

Stohlgren, T.J., Falkner, M.B. & Schell, L.D. 1995. A modified-Whittaker nested vegetation sampling method. Vegetatio, 117(2), 113-121.

Struhsaker, T.T. 1997. Ecology of an African Rain Forest: Logging in Kibale and the Conflict between Conservation and Exploitation. University Press of Florida.

Taubert, F., Fischer, R., Groeneveld, J., Lehmann, S., Müller, M.S., Rödig, E., Wiegand, T. & Huth, A. 2018. Global patterns of tropical forest fragmentation. Nature, 5547693, 519-522.

Teixeira, F.Z., Coelho, A.V., Esperandio, I.B. & Kindel, A. 2013. Vertebrate road mortality estimates: Effects of sampling methods and carcass removal. Biological Conservation, 157, 317-323.

Tilman, D., May, R.M., Lehman, C.L. & Nowak, M.A. 1994. Habitat destruction and the extinction debt. Nature, 3716492, 65-66.

Tillman, F.D., Callegary, J.B., Nagler, P.L. & Glenn, E.P. 2012. A simple method for estimating basin-scale groundwater discharge by vegetation in the basin and range province of Arizona using remote sensing information and geographic information systems. Journal of Arid Environments, 82, 44-52.

Tinker, L., Cobb, D., Bond, A., Cashmore, M. & Mason-Apps, J. 2005. Impact mitigation in environmental impact assessment: Paper promises or the basis of consent conditions? Impact Assessment and Project Appraisal, 23(4), 265-280.

Turner, M.G., Gardner, R.H. & O'Neill, R.V. 2001. Landscape Ecology in Theory and Practice: Pattern and Process. Springer-Verlag, New York.

Turner, W., Spector, S., Gardiner, N., Fladeland, M., Sterling, E. & Steininger, M. 2003. Remote sensing for biodiversity science and conservation. Trends in Ecology & Evolution, 18(6), 306-314.

Turpie, J., Marais, C. & Blignaut, J. 2008. The working for water programme: Evolution of a payments for ecosystem services mechanism that addresses both poverty and ecosystem service delivery in South Africa. Ecological Economics, 65(4), 788-798.

Uganda National Roads Authority 2020. Annual Roads Sector Performance Report FY 2019/2020. Ministry of Works and Transport, Kampala.

Uganda National Roads Authority 2020. Environmental and Social Management Framework for Road Development Projects. UNRA, Kampala.

Underwood, A.J. 1994. On beyond BACI: Sampling designs that might reliably detect environmental disturbances. Ecological Applications, 4(1), 3-15.

United Nations 2015. Transforming Our World: The 2030 Agenda for Sustainable Development. A/RES/70/1.

UNRA (Uganda National Roads Authority) 2015. Environmental and Social Management Framework for the Fort Portal-Kamwenge Road Project. Uganda National Roads Authority, Kampala.

UNRA (Uganda National Roads Authority) 2018. Fort Portal-Kamwenge Road Completion Report. Uganda National Roads Authority, Kampala.

UWA (Uganda Wildlife Authority) 2015. Kibale National Park General Management Plan 2015-2025. Uganda Wildlife Authority, Kampala.

UWA (Uganda Wildlife Authority) 2019. Strategic Plan 2019-2024: Conserving for Generations. Uganda Wildlife Authority, Kampala.

UWA (Uganda Wildlife Authority) 2023. Research Guidelines and Permit Requirements for Protected Areas. Uganda Wildlife Authority, Kampala.

Vedeld, P., Angelsen, A., Bojö, J., Sjaastad, E. & Berg, G.K. 2007. Forest environmental incomes and the rural poor. Forest Policy and Economics, 9(7), 869-879.

Villarroya, A., Barros, A.C. & Kiesecker, J. 2014. Policy development for environmental licensing and biodiversity offsets in Latin America. PLoS One, 9(9), e107144.

WCS (Wildlife Conservation Society) 2018. Kibale Chimpanzee Population Survey 2018: Technical Report. Wildlife Conservation Society, Albertine Rift Programme, Kampala, Uganda.

Whitmore, T.C. 1998. An Introduction to Tropical Rain Forests (2nd ed.). Oxford University Press.

Wilkie, D., Shaw, E., Rotberg, F., Morelli, G. & Auzel, P. 2000. Roads, development, and conservation in the Congo Basin. Conservation Biology, 14(6), 1614-1622.

Wilson, K.A., Auerbach, N.A., Sam, K., Magini, A.G., Moss, A.S., Langhans, S.D., Budiharta, S., Terzano, D. & Meijaard, E. 2016. Conservation research is not happening where it is most needed. PLoS Biology, 14(3), e1002413.

Winkler, K., Fuchs, R., Rounsevell, M. & Herold, M. 2021. Global land use changes are four times greater than previously estimated. Nature Communications, 12(1), 2501.

World Bank 2018. Environmental and Social Framework. Washington, DC.

Yin, R.K. 2014. Case Study Research: Design and Methods (5th ed.). Sage Publications.

Zeller, K.A., McGarigal, K. & Whiteley, A.R. 2012. Estimating landscape resistance to movement: A review. Landscape Ecology, 27(6), 777-797.

Zhu, Z. & Woodcock, C.E. 2014. Continuous change detection and classification of land cover using all available Landsat data. Remote Sensing of Environment, 144, 152-171.

Adams, W.M., Aveling, R., Brockington, D., Dickson, B., Elliott, J., Hutton, J., Roe, D., Vira, B. and Wolmer, W., 2004. Biodiversity conservation and the eradication of poverty. Science, 3065699, pp.1146-1149.

AfDB (African Development Bank), 2012. African Development Bank's Integrated Safeguards System: Policy Statement and Operational Safeguards. African Development Bank Group, Tunis.

AfDB (African Development Bank), 2013. Safeguards and Sustainability Series Volume 1 - Issue 1 (Dec. 2013): African Development Bank Group's Integrated Safeguards System. African Development Bank Group, Tunis.

Ahmed, S.E., Lees, A.C., Moura, N.G., Gardner, T.A., Barlow, J., Ferreira, j., Ewers, R.M. 2014. Road networks predict human influence on Amazonian bird communities. Proceedings of the Royal Society B, 2851892, 20172497.

Alamgir, M., Campbell, M.J., Sloan, S., Goosem, M., Clements, G.R., Mahmoud, M.I. and Laurance, W.F., 2017. Economic, socio-political and environmental risks of road development in the tropics. Current Biology, 27(20), pp.R1130-R1140.

Alignier, A. and Deconchat, M., 2011. Variability of forest edge effect on vegetation implies reconsideration of its assumed hypothetical pattern. Applied Vegetation Science, 14(1), pp.67-74.

Apostolopoulou, E. and Adams, W.M., 2017. Biodiversity offsetting and conservation: reframing nature to save it. Oryx, 51(1), pp.23-31.

Arlidge, W.N., Bull, J.W., Addison, P.F., Burgass, M.J., Gianuca, D., Gorham, T.M., Jacob, C., Shumway, N., Sinclair, S.P., Watson, J.E. and Wilcox, C., 2018. A global mitigation hierarchy for nature conservation. BioScience, 68(5), pp.336-347.

Arroyo-Rodríguez, V., Rös, M., Escobar, F., Melo, F.P., Santos, B.A., Tabarelli, M. and Chazdon, R., 2013. Plant β-diversity in fragmented rain forests: testing floristic homogenization and differentiation hypotheses. Journal of Ecology, 101(6), pp.1449-1458.

Arroyo-Rodríguez, V., Saldaña-Vázquez, R.A., Fahrig, L. and Santos, B.A., 2017. Does forest fragmentation cause an increase in forest temperature? Ecological Research, 32(1), pp.81-88.

Australian Government Department of Agriculture, Water and the Environment. 2021. Environment Protection and Biodiversity Conservation Act 1999. Canberra: Australian Government.

Avon, C., Bergès, L., Dumas, Y. and Dupouey, J.L., 2010. Does the effect of forest roads extend a few meters or more into the adjacent forest? A study on understory plant diversity in managed oak stands. Forest Ecology and Management, 259(8), pp.1546-1555.

Baguette, M., Blanchet, S., Legrand, D., Stevens, V.M. and Turlure, C., 2013. Individual dispersal, landscape connectivity and ecological networks. Biological Reviews, 88(2), pp.310-326.

Balmford, A. and Whitten, T., 2003. Who should pay for tropical conservation, and how could the costs be met? Oryx, 37(2), pp.238-250.

Barber, C.P., Cochrane, M.A., Souza Jr, C.M. and Laurance, W.F., 2014. Roads, deforestation, and the mitigating effect of protected areas in the Amazon. Biological Conservation, 177, pp.203-209.

Barlow, J., Lennox, G.D., Ferreira, J., Berenguer, E., Lees, A.C., Mac Nally, R., Thomson, J.R., Ferraz, S.F.D.B., Louzada, J., Oliveira, V.H.F. and Parry, L., 2016. Anthropogenic disturbance in tropical forests can double biodiversity loss from deforestation. Nature, 5357610, pp.144-147.

BBOP (Business and Biodiversity Offsets Programme), 2012. Standard on Biodiversity Offsets. BBOP, Washington, D.C.

Benitez-Malvido, J., & Martinez-Ramos, M. 2013. Impact of forest fragmentation on understory plant species richness in Amazonia. Conservation Biology, 17(2), 389-400.

Bigard, C., Pioch, S. and Thompson, J.D., 2017. The inclusion of biodiversity in environmental impact assessment: Policy-related progress limited by gaps and semantic confusion. Journal of Environmental Management, 200, pp.35-45.

Blois, J.L., Williams, J.W., Fitzpatrick, M.C., Jackson, S.T. and Ferrier, S., 2013. Space can substitute for time in predicting climate-change effects on biodiversity. Proceedings of the National Academy of Sciences, 110(23), pp.9374-9379.

Brandão Jr, A., Rausch, L., Durán, A.P., et al. 2020. Estimating the potential for conservation and farming in the Amazon and Cerrado under four policy scenarios. Sustainability, 12(3), 1277.

Broadbent, E.N., Asner, G.P., Keller, M., Knapp, D.E., Oliveira, P.J. and Silva, J.N., 2008. Forest fragmentation and edge effects from deforestation and selective logging in the Brazilian Amazon. Biological Conservation, 141(7), pp.1745-1757.

Bull, J.W., Gordon, A., Watson, J.E. and Maron, M., 2016. Seeking convergence on the key concepts in 'no net loss' policy. Journal of Applied Ecology, 53(6), pp.1686-1693.

Bull, J.W., Suttle, K.B., Gordon, A., Singh, N.J. and Milner-Gulland, E.J., 2013. Biodiversity offsets in theory and practice. Oryx, 47(3), pp.369-380.

Bull, J.W., Strange, N., Smith, R.J. and Gordon, A., 2020. Reconciling multiple counterfactuals when evaluating biodiversity conservation impact in social-ecological systems. Conservation Biology, 34(4), pp.895-905.

Bull, J.W., Milner-Gulland, E.J., Addison, P.F., Arlidge, W.N., Baker, J., Brooks, T.M., Taylor, M.L., Weller, K., zu Ermgassen, S.O., Sidemo-Holm, W. and Watson, J.E., 2022. Net positive outcomes for nature. Nature Ecology & Evolution, 6(7), pp.869-877.

Government of Canada, 2019, Impact Assessment Act, Government of Canada, Ottawa.

Caro, T., Dobson, A., Marshall, A.J. and Peres, C.A., 2014. Compromise solutions between conservation and road building in the tropics. Current Biology, 24(16), pp.R722-R725.

Carreras Gamarra, M.J., Lassoie, J.P. and Milder, J., 2018. Accounting for no net loss: A critical assessment of biodiversity offsetting metrics and methods. Journal of Environmental Management, 220, pp.36-43.

CASP (Critical Appraisal Skills Programme), 2018. CASP Checklists. Available at: https://casp-uk.net/casp-tools-checklists/ [Accessed 15 September 2023].

Catterall, C.P., Freeman, A.N., Kanowski, J., & Freebody, K. 2016. Can active restoration of tropical rainforest rescue biodiversity? A case with bird community indicators. Biological Conservation, 191, 142-149.

Chapman, C.A., Struhsaker, T.T. and Lambert, J.E., 2005. Thirty years of research in Kibale National Park, Uganda, reveals a complex picture for conservation. International Journal of Primatology, 26(3), pp.539-555.

Chazdon, R.L. and Guariguata, M.R., 2016. Natural regeneration as a tool for large-scale forest restoration in the tropics: prospects and challenges. Biotropica, 48(6), pp.716-730.

Clare, S., Krogman, N., Foote, L. and Lemphers, N., 2011. Where is the avoidance in the implementation of wetland law and policy? Wetlands Ecology and Management, 19(2), pp.165-182.

Clarke, K.R., 1993. Non-parametric multivariate analyses of changes in community structure. Australian Journal of Ecology, 18(1), pp.117-143.

Clarke, K.R. and Warwick, R.M., 2001. Change in marine communities: an approach to statistical analysis and interpretation. 2nd Edition. PRIMER-E Ltd, Plymouth.

Coffin, A.W., 2007. From roadkill to road ecology: a review of the ecological effects of roads. Journal of Transport Geography, 15(5), pp.396-406.

Collinson, W., Davies-Mostert, H., Roxburgh, L. and van der Ree, R., 2019. Status of road ecology research in Africa: do we understand the impacts of roads, and how to successfully mitigate them? Frontiers in Ecology and Evolution, 7, p.479.

Congalton, R.G. and Green, K., 2009. Assessing the accuracy of remotely sensed data: principles and practices. CRC press, Boca Raton.

Congalton, R.G. and Green, K., 2019. Assessing the accuracy of remotely sensed data: principles and practices. CRC press, Boca Raton.

Corlett, R.T., 2016. Plant diversity in a changing world: status, trends, and conservation needs. Plant Diversity, 38(1), pp.10-16.

Creswell, J. W., & Creswell, J. D. 2018. Research design: Qualitative, quantitative, and mixed methods approaches (5th ed.). Sage Publications.

Crouzeilles, R., Curran, M., Ferreira, M.S., Lindenmayer, D.B., Grelle, C.E. and Rey Benayas, J.M., 2016. A global meta-analysis on the ecological drivers of forest restoration success. Nature Communications, 7(1), pp.1-8.

Deljouei, A., Abdi, E., Marcantonio, M., Majnounian, B., Amici, V. and Sohrabi, H., 2017. The impact of road disturbance on vegetation and soil properties in a beech stand, Hyrcanian forest. European Journal of Forest Research, 136(4), pp.565-579.

Eigenbrod, F., Hecnar, S.J. and Fahrig, L., 2009. Quantifying the road-effect zone: threshold effects of a motorway on anuran populations in Ontario, Canada. Ecology and Society, 14(1), p.24.

Elzinga, C. L., Salzer, D. W., Willoughby, J. W., & Gibbs, J. P. 2015. Monitoring plant and animal populations: a handbook for field biologists. John Wiley & Sons.

Ewers, R.M. and Didham, R.K., 2006. Confounding factors in the detection of species responses to habitat fragmentation. Biological Reviews, 81(1), pp.117-142.

Ezzine-de-Blas, D., Wunder, S., Ruiz-Pérez, M. and del Pilar Moreno-Sanchez, R., 2016. Global patterns in the implementation of payments for environmental services. PloS one, 11(3), p.e0149847.

Fahrig, L., 2003. Effects of habitat fragmentation on biodiversity. Annual Review of Ecology, Evolution, and Systematics, 34(1), pp.487-515.

Fahrig, L. and Rytwinski, T., 2009. Effects of roads on animal abundance: an empirical review and synthesis. Ecology and Society, 14(1), p.21.

Fearnside, P.M., 2015. Highway construction as a force in destruction of the Amazon forest. In Handbook of Road Ecology (pp. 414-424). John Wiley & Sons, Chichester.

Ferraro, P.J. and Pattanayak, S.K., 2006. Money for nothing? A call for empirical evaluation of biodiversity conservation investments. PLoS Biology, 4(4), p.e105.

Fletcher, R.J., Didham, R.K., Banks-Leite, C., et al. 2018. Is habitat fragmentation good for biodiversity? Biological Conservation, 226, 9-15.

Foody, G.M., 2002. Status of land cover classification accuracy assessment. Remote Sensing of Environment, 80(1), pp.185-201.

Forman, R.T., 1995. Land mosaics: the ecology of landscapes and regions. Cambridge University Press, Cambridge.

Forman, R.T. and Deblinger, R.D., 2000. The ecological road-effect zone of a Massachusetts (USA) suburban highway. Conservation Biology, 14(1), pp.36-46.

Forman, R.T., Sperling, D., Bissonette, J.A., Clevenger, A.P., Cutshall, C.D., Dale, V.H., Fahrig, L., France, R.L., Heanue, K., Goldman, C.R. and Jones, J., 2003. Road ecology: science and solutions. Island Press, Washington, D.C.

Fuentes-Montemayor, E., Ferryman, M., Watts, K., Macgregor, N.A., Lopez-Gallego, Z., Pahlen, J., Procter, D.S., Irvine, K.N. and Park, K.J., 2020. Small mammal responses to long-term large-scale woodland creation: the influence of local and landscape-level attributes. Ecological Applications, 30(2), p.e02028.

Gardner, T.A., von Hase, A., Brownlie, S., Ekstrom, J.M., Pilgrim, J.D., Savy, C.E., Stephens, R.T., Treweek, J., Ussher, G.T., Ward, G. and Ten Kate, K., 2013. Biodiversity offsets and the challenge of achieving no net loss. Conservation Biology, 27(6), pp.1254-1264.

Gardner, T.A., Barlow, J., Chazdon, R., Ewers, R.M., Harvey, C.A., Peres, C.A. and Sodhi, N.S., 2009. Prospects for tropical forest biodiversity in a human-modified world. Ecology Letters, 12(6), pp.561-582.

Gelot, M. and Bigard, C., 2021. Ecological mitigation hierarchy and biodiversity offsets revisited through habitat connectivity modelling. Journal of Environmental Management, 277, p.111437.

Ghazoul, J., & Chazdon, R. 2017. Degradation and recovery in changing forest landscapes: A multiscale conceptual framework. Annual Review of Environment and Resources, 42, 161-188.

Gibson, L., Lee, T.M., Koh, L.P., Brook, B.W., Gardner, T.A., Barlow, J., Peres, C.A., Bradshaw, C.J., Laurance, W.F., Lovejoy, T.E. and Sodhi, N.S., 2011. Primary forests are irreplaceable for sustaining tropical biodiversity. Nature, 4787369, pp.378-381.

Gonzalez, R.C. and Woods, R.E., 2009. Digital Image Processing. 3rd Edition. Pearson, Upper Saddle River.

Goosem, M., Chazdon, R.L., Suntana, A.S., Laurance, W.F., Wilson, R.F., Baxter, G., Asega, S., Abe, E.M., Agbo, B. and Pinard, M., 2021. The effectiveness of canopy bridges for arboreal mammals: A systematic review and meta-analysis. PLoS One, 16(6), p.e0252088.

GoU (Government of Uganda), 2019. The National Environment Act, 2019. Uganda Printing and Publishing Corporation, Entebbe.

GoU (Government of Uganda), 2020. The National Environment (Environmental and Social Assessment) Regulations, 2020. Uganda Printing and Publishing Corporation, Entebbe.

Hansen, M.C., Potapov, P.V., Moore, R., Hancher, M., Turubanova, S.A., Tyukavina, A., Thau, D., Stehman, S.V., Goetz, S.J., Loveland, T.R. and Kommareddy, A., 2013. High-resolution global maps of 21st-century forest cover change. Science, 3426160, pp.850-853.

Hanski, I., 1998. Metapopulation dynamics. Nature, 3966706, pp.41-49.

Harper, K.A., Macdonald, S.E., Burton, P.J., Chen, J., Brosofske, K.D., Saunders, S.C., Euskirchen, E.S., Roberts, D., Jaiteh, M.S. and Esseen, P.A., 2005. Edge influence on forest structure and composition in fragmented landscapes. Conservation Biology, 19(3), pp.768-782.

Hartter, J., 2010. Resource use and ecosystem services in a forest park landscape. Society and Natural Resources, 23(3), pp.207-223.

Holm, L.G. and Valtonen, M., 2009. Herbs and Shrubs of Kibale National Park: A Field Guide. Wildlife Conservation Society, New York.

Hosonuma, N., Herold, M., De Sy, V., De Fries, R.S., Brockhaus, M., Verchot, L., Angelsen, A. and Romijn, E., 2012. An assessment of deforestation and forest degradation drivers in developing countries. Environmental Research Letters, 7(4), p.044009.

IFC (International Finance Corporation), 2012. Performance Standard 6: Biodiversity Conservation and Sustainable Management of Living Natural Resources. World Bank Group, Washington, D.C.

Ives, C.D. and Bekessy, S.A., 2015. The ethics of offsetting nature. Frontiers in Ecology and the Environment, 13(10), pp.568-573.

Jacob, C., Vaissiere, A.C., Bas, A. and Calvet, C., 2016. Investigating the inclusion of ecosystem services in biodiversity offsetting. Ecosystem Services, 21, pp.92-102.

Jain, A.K., 1989. Fundamentals of digital image processing. Prentice-Hall, Inc., Englewood Cliffs.

Jakovac, C.C., Peña-Claros, M., Kuyper, T.W., & Bongers, F. 2016. Loss of secondary-forest resilience by land-use intensification in the Amazon. Journal of Ecology, 104(1), 67-77.

Jensen, J.R., 2016. Introductory digital image processing: a remote sensing perspective. Pearson Education, Inc., Glenview.

Kent, M., 2011. Vegetation description and data analysis: a practical approach. John Wiley & Sons, Chichester.

Kiesecker, J.M., Copeland, H., Pocewicz, A. and McKenney, B., 2010. Development by design: blending landscape-level planning with the mitigation hierarchy. Frontiers in Ecology and the Environment, 8(5), pp.261-266.

Kleinschroth, F. and Healey, J.R., 2017. Impacts of logging roads on tropical forests. Biotropica, 49(5), pp.620-635.

Knight, K.B., Seddon, E.S. and Toombs, T.P., 2020. A framework for evaluating biodiversity mitigation metrics. Ambio, 49(6), pp.1232-1240.

Kruskal, J.B., 1964. Nonmetric multidimensional scaling: a numerical method. Psychometrika, 29(2), pp.115-129.

Laurance, W.F., Goosem, M. and Laurance, S.G., 2009. Impacts of roads and linear clearings on tropical forests. Trends in Ecology & Evolution, 24(12), pp.659-669.

Laurance, W.F., Clements, G.R., Sloan, S., O'connell, C.S., Mueller, N.D., Goosem, M., Venter, O., Edwards, D.P., Phalan, B., Balmford, A. and Van Der Ree, R., 2014. A global strategy for road building. Nature, 5137517, pp.229-232.

Laurance, W.F., Peletier-Jellema, A., Geenen, B., Koster, H., Verweij, P., Van Dijck, P., Lovejoy, T.E., Schleicher, J. and Van Kuijk, M., 2015. Reducing the global environmental impacts of rapid infrastructure expansion. Current Biology, 25(7), pp.R259-R262.

Laurance, W.F., Camargo, J.L., Luizão, R.C., Laurance, S.G., Pimm, S.L., Bruna, E.M., Stouffer, P.C., Williamson, G.B., Benítez-Malvido, J., Vasconcelos, H.L. and Van Houtan, K.S., 2011. The fate of Amazonian forest fragments: a 32-year investigation. Biological Conservation, 144(1), pp.56-67.

Levins, R., 1969. Some demographic and genetic consequences of environmental heterogeneity for biological control. American Entomologist, 15(3), pp.237-240.

Lillesand, T.M. and Kiefer, R.W., 2015. Remote sensing and image interpretation. John Wiley & Sons, New York.

Lindenmayer, D.B. and Fischer, J., 2006. Habitat fragmentation and landscape change: an ecological and conservation synthesis. Island Press, Washington, D.C.

Lu, D., Li, G., & Moran, E. 2014. Current situation and needs of change detection techniques. International Journal of Image and Data Fusion, 5(1), 13-38.

MacArthur, R.H. and Wilson, E.O., 1967. The theory of island biogeography. Princeton University Press, Princeton.

MacKenzie, C.A., Salerno, J., Hartter, J., Chapman, C.A., Reyna, R., Tumusiime, D.M. and Drake, M., 2017. Changing perceptions of protected area benefits and problems around Kibale National Park, Uganda. Journal of Environmental Management, 200, pp.217-228.

Magnago, L. F. S., Rocha, M. F., Meyer, L., Martins, S. V., & Meira-Neto, J. A. A. 2015. Microclimatic Conditions at Forest Edges Have Significant Impacts on Vegetation

Structure in Large Atlantic Forest Fragments. Biodiversity and Conservation, 24, 2305-2318.

Malhi, Y., Roberts, J.T., Betts, R.A., Killeen, T.J., Li, W. and Nobre, C.A., 2008. Climate change, deforestation, and the fate of the Amazon. Science, 3195860, pp.169-172.

Maron, M., Hobbs, R.J., Moilanen, A., Matthews, J.W., Christie, K., Gardner, T.A., Keith, D.A., Lindenmayer, D.B. and McAlpine, C.A., 2012. Faustian bargains? Restoration realities in the context of biodiversity offset policies. Biological Conservation, 155, pp.141-148.

Maron, M., Brownlie, S., Bull, J.W., Evans, M.C., von Hase, A., Quétier, F., Watson, J.E. and Gordon, A., 2018. The many meanings of no net loss in environmental policy. Nature Sustainability, 1(1), pp.19-27.

McCune, B. and Grace, J.B., 2002. Analysis of ecological communities. MjM Software Design, Gleneden Beach.

Meli, P., Holl, K.D., Benayas, J.M.R., Jones, H.P., Jones, P.C., Montoya, D. and Mateos, D.M., 2017. A global review of past land use, climate, and active vs. passive restoration effects on forest recovery. PloS one, 12(2), p.e0171368.

Mendenhall, C.D., Karp, D.S., Meyer, C.F., Hadly, E.A. and Daily, G.C., 2014. Predicting biodiversity change and averting collapse in agricultural landscapes. Nature, 5097499, pp.213-217.

Menezes Silva, M. A., Araújo Mendes Alencar, P. G., Novacosque Feitosa Guerra, T., Laurênio Melo, A., Borges Lins-e-Silva, A. C., & Nogueira Rodal, M. J. 2015. Edge effects on the structure and dynamics of an Atlantic Forest fragment in northeastern Brazil. Revista Brasileira de Ciências Agrárias, 10(4), 538-543.

Millennium Ecosystem Assessment, 2005. Ecosystems and human well-being: synthesis. Island Press, Washington, D.C.

Moher, D., Liberati, A., Tetzlaff, J. and Altman, D.G., 2009. Preferred reporting items for systematic reviews and meta-analyses: the PRISMA statement. PLoS Medicine, 6(7), p.e1000097.

Morrison-Saunders, A. and Pope, J., 2013. Conceptualising and managing trade-offs in sustainability assessment. Environmental Impact Assessment Review, 38, pp.54-63.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., Da Fonseca, G.A. and Kent, J., 2000. Biodiversity hotspots for conservation priorities. Nature, 4036772, pp.853-858.

Namaalwa, S. and Byakagaba, P., 2019. Examining the mitigation hierarchy in environmental impact assessment: The case of selected road projects in Uganda. Journal of Environmental Planning and Management, 62(12), pp.2095-2115.

Namatovu, S., 2022. Assessing the effectiveness of biodiversity offsets in Uganda: A case study of oil and gas development in the Albertine Graben. Environmental Impact Assessment Review, 92, p.106697.

Nangendo, G., Namukisa, L., Pomeroy, D., Plumptre, A.J. and Ndemere, P., 2022. Oil and gas development in Murchison Falls National Park: Effects of the exploration and appraisal phase on biodiversity. Environmental Impact Assessment Review, 92, p.106690.

Narmada, K., Srinivasulu, C. and Srinivasulu, B., 2021. Accuracy Assessment in Remote Sensing Classification: A Review. In Proceedings of the International Conference on Innovative Computing & Communications (ICICC).

Naughton-Treves, L., Treves, A., Chapman, C. and Wrangham, R., 1998. Temporal patterns of crop-raiding by primates: linking food availability in croplands and adjacent forest. Journal of Applied Ecology, 35(4), pp.596-606.

NEMA (National Environment Management Authority), 2011. Environmental and Social Impact Assessment for the Proposed Upgrading of Fort Portal-Kamwenge Road (66.2km) to Bitumen Standard. National Environment Management Authority, Kampala.

NEMA (National Environment Management Authority), 2022. National Guidelines for Biodiversity and Social Offsets. National Environment Management Authority, Kampala.

Olofsson, P., Foody, G. M., Herold, M., Stehman, S. V., Woodcock, C. E., & Wulder, M. A. 2014. Good practices for estimating area and assessing accuracy of land change. Remote Sensing of Environment, 148, 42-57.

Patton, M. Q. 2002. Qualitative research and evaluation methods (3rd ed.). Sage Publications.

Phalan, B., Hayes, G., Brooks, S., Marsh, D., Howard, P., Costelloe, B., Vira, B., Kowalska, A. and Whitaker, S., 2018. Avoiding impacts on biodiversity through strengthening the first stage of the mitigation hierarchy. Oryx, 52(2), pp.316-324.

Pickett, S.T., 1989. Space-for-time substitution as an alternative to long-term studies. In Long-term studies in ecology (pp. 110-135). Springer, New York.

Reed, M.S., 2008. Stakeholder participation for environmental management: a literature review. Biological Conservation, 141(10), pp.2417-2431.

Richards, J.A. and Jia, X., 2006. Remote sensing digital image analysis: an introduction. Springer, Berlin.

Rozendaal, D.M.A., Bongers, F., Aide, T.M., et al. 2019. Biodiversity recovery of Neotropical secondary forests. Science Advances, 5(3), eaau3114.

Rwanga, S.S. and Ndambuki, J.M., 2017. Accuracy assessment of land use/land cover classification using remote sensing and GIS. International Journal of Geosciences, 8(4), pp.611-622.

Rytwinski, T., Soanes, K., Jaeger, J.A., Fahrig, L., Findlay, C.S., Houlahan, J., Van Der Ree, R. and van der Grift, E.A., 2016. How effective is road mitigation at reducing road-kill? A meta-analysis. PLoS One, 11(11), p.e0166941.

Schröter, M., van der Zanden, E.H., van Oudenhoven, A.P., Remme, R.P., Serna-Chavez, H.M., De Groot, R.S. and Opdam, P., 2014. Ecosystem services as a contested concept: a synthesis of critique and counter-arguments. Conservation Letters, 7(6), pp.514-523.

Segan, D.B., Murray, K.A. and Watson, J.E., 2016. A global assessment of current and future biodiversity vulnerability to habitat loss–climate change interactions. Global Ecology and Conservation, 5, pp.12-21.

Sharma, L.N., Grytnes, J.A., Måren, I.E., & Vetaas, O.R. 2016. Do composition and richness of woody plants vary between gaps and closed canopy patches in subtropical forests? Journal of Vegetation Science, 27(6), 1129-1139.

Sodhi, N.S. and Ehrlich, P.R. eds., 2010. Conservation biology for all. Oxford University Press, Oxford.

Sonter, L.J., Simmonds, J.S., Watson, J.E., Jones, J.P., Kiesecker, J.M., Costa, H.M., Bennun, L., Edwards, S., Grantham, H.S., Griffiths, V.F. and Jones, K., 2020. Local conditions and policy design determine whether ecological compensation can achieve no net loss goals. Nature Communications, 11(1), pp.1-11.

Spash, C.L., 2015. Bulldozing biodiversity: The economics of offsets and trading-in Nature. Biological Conservation, 192, pp.541-551.

Spellerberg, I.F., 1998. Ecological effects of roads and traffic: a literature review. Global Ecology and Biogeography, 7(5), pp.317-333.

Stampone, M.D., Hartter, J., Chapman, C.A. and Ryan, S.J., 2011. Trends and variability in localized precipitation around Kibale National Park, Uganda, Africa. Research Journal of Environmental and Earth Sciences, 3(1), pp.14-23.

Stehman, S. V., & Foody, G. M. 2019. Key issues in rigorous accuracy assessment of land cover products. Remote Sensing of Environment, 231, 111199.

Struhsaker, T.T., 1997. Ecology of an African rain forest: logging in Kibale and the conflict between conservation and exploitation. University Press of Florida, Gainesville.

Tallis, H., Kennedy, C.M., Ruckelshaus, M., Goldstein, J. and Kiesecker, J.M., 2015. Mitigation for one & all: An integrated framework for mitigation of development impacts on biodiversity and ecosystem services. Environmental Impact Assessment Review, 55, pp.21-34.

Teixeira, F.Z., Printes, R.C., Fagundes, J.C.G., Alonso, A.C. and Kindel, A., 2013. Canopy bridges as road overpasses for wildlife in urban fragmented landscapes. Biota Neotropica, 13(1), pp.117-123.

Tinker, L., Cobb, D., Bond, A. and Cashmore, M., 2005. Impact mitigation in environmental impact assessment: paper promises or the basis of consent conditions? Impact Assessment and Project Appraisal, 23(4), pp.265-280.

Trombulak, S.C. and Frissell, C.A., 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conservation Biology, 14(1), pp.18-30.

UNDP (United Nations Development Programme), 2016. Biodiversity Offsets: A User Guide. UNDP, New York.

UNRA (Uganda National Roads Authority), 2019. Guidelines for Minimizing Impacts on Wildlife in Road Projects. Uganda National Roads Authority, Kampala.

UWA (Uganda Wildlife Authority), 2015. Kibale National Park General Management Plan 2015-2025. Uganda Wildlife Authority, Kampala.

van der Ree, R., Smith, D.J. and Grilo, C. eds., 2015. Handbook of road ecology. John Wiley & Sons, Chichester.

Vancutsem, C., Achard, F., Pekel, J.F., Vieilledent, G., Carboni, S., Simonetti, D., Gallego, J., Aragão, L.E. and Nasi, R., 2021. Long-term (1990–2019) monitoring of forest cover changes in the humid tropics. Science Advances, 7(10), p.eabe1603.

Vanonckelen, S., Lhermitte, S. and Van Rompaey, A., 2013. The effect of atmospheric and topographic correction methods on land cover classification accuracy. International Journal of Applied Earth Observation and Geoinformation, 24, pp.9-21.

Wilson, E.O., 1988. Biodiversity. National Academy Press, Washington, D.C.

Wilson, S.J., Schelhas, J., Grau, R., et al. 2020. Forest ecosystem-service transitions: The ecological dimensions of the forest transition. Ecology and Society, 25(1), 27.

Wintle, B.A., Kujala, H., Whitehead, A., Cameron, A., Veloz, S., Kukkala, A., Moilanen, A., Gordon, A., Lentini, P.E., Cadenhead, N.C. and Bekessy, S.A., 2019. Global synthesis of conservation studies reveals the importance of small habitat patches for biodiversity. Proceedings of the National Academy of Sciences, 116(3), pp.909-914.

World Bank, 2018. Environmental and Social Framework. World Bank, Washington, D.C.

Zhu, Z., 2017. Change detection using landsat time series: A review of frequencies, preprocessing, algorithms, and applications. ISPRS Journal of Photogrammetry and Remote Sensing, 130, pp.370-384.

zu Ermgassen, S.O., Baker, J., Griffiths, R.A., Strange, N., Struebig, M.J. and Bull, J.W., 2019. The ecological outcomes of biodiversity offsets under "no net loss" policies: A global review. Conservation Letters, 12(6), p.e12664.

APPENDIX

Family	**Species**	**Plant form**	**0 m**	**50 m**	**100 m**	**Status**	**Origin**
Acanthaceae	*Acanthus pubescens* Engl.	Herb	1	4	57	LC	Native
Acanthaceae	*Mimulopsis speciosa* Baker	Herb	28	17	71	LC	Native
Achariaceae	*Dasylepis eggelingii* J.B. Gillett	Tree	2	2	2	LC	Native
Annonaceae	*Monodora myristica* (Gaertn.) Dunal	Tree	12	11	15	LC	Native
Annonaceae	*Uvariopsis congensis* Robyns & Ghesq.	Tree	41	88	38	LC	Native
Apocynaceae	*Funtumia africana* (Benth.) Stapf	Tree	51	26	25	LC	Native
Apocynaceae	*Mondia whitei* (Hook. f.) Skeels	Climber	1	0	0	VU	Native
Apocynaceae	*Rauvolfia vomitoria* Afzel.	Shrub/Tree	0	3	2	LC	Native
Apocynaceae	*Tabernaemontana odoratissima* (Stapf) Leeuwenb.	Shrub/Tree	9	20	16	LC	Native

Arecaceae	*Phoenix reclinata* Jacq.	Tree	0	15	20	NT	Native
Asparagaceae	*Dracaena laxissima* Engl.	Shrub	0	0	1	LC	Native
Asteraceae	*Aspilia africana* (Pers.) C.D. Adams	Herb	0	0	3	LC	Native
Asteraceae	*Vernonia auriculifera* Hiern	Shrub	0	0	2	LC	Native
Bignoniaceae	*Markhamia lutea* (Benth.) K. Schum.	Tree	0	3	2	LC	Native
Bignoniaceae	*Spathodea campanulata* P. Beauv.	Tree	0	1	0	LC	Native
Canellaceae	*Warburgia ugandensis* Sprague	Tree	3	0	0	VU	Native
Cannabaceae	*Celtis africana* Burm. f.	Tree	6	3	1	LC	Native
Cannabaceae	*Celtis gomphophylla Baker*	Tree	12	14	9	NT	Native
Commelinaceae	*Commelina benghalensis* L.	Herb	3	0	2	LC	Native
Commelinaceae	*Palisota schweinfurthii* C.B. Clarke	Herb	1	0	2	LC	Native
Convolvulaceae	*Ipomoea spathulata* Hallier f.	Herb	19	0	0	LC	Native

Cordiaceae	*Cordia millenii* Baker	Tree	0	1	0	EN	Native
Costaceae	*Costus spectabilis* (Fenzl) K. Schum.	Herb	60	55	15	LC	Native
Ebenaceae	*Diospyros abyssinica* (Hiern) F. White	Tree	14	29	4	LC	Native
Euphorbiaceae	*Acalypha ornata Hochst.* ex A. Rich.	Shrub	10	25	50	LC	Native
Euphorbiaceae	*Croton sylvaticus* Hochst.	Tree	1	0	1	LC	Native
Euphorbiaceae	*Macaranga schweinfurthii* Pax	Tree	0	1	2	LC	Native
Euphorbiaceae	*Shirakiopsis elliptica (Hochst.) Esser*	Tree	6	7	5	LC	Native
Fabaceae	*Vachellia sieberiana (DC.) Kyal. & Boatwr.*	Tree	0	1	2	LC	Native
Fabaceae	*Albizia coriaria* Welw. ex Oliv.	Tree	1	0	0	LC	Native
Fabaceae	*Albizia grandibracteata* Taub.	Tree	4	1	0	LC	Native
Fabaceae	*Cynometra alexandri* C.H.Wright	Tree	107	13	3	LC	Native
Fabaceae	*Newtonia buchananii* (Baker f.)	Tree	0	0	1	LC	Native

Fabaceae	*Senna spectabilis* (DC.) H.S. Irwin & Barneby	Tree	0	0	1	LC	Alien
Francoaceae	*Bersama abyssinica* Fresen.	Tree/Shrub	1	0	2	LC	Native
Hypericaceae	*Harungana madagascariensis* Lam. ex Poir.	Tree	0	2	0	LC	Native
Lamiaceae	*Clerodendrum buchholzii* Gürke	Climber	1	1	0	LC	Native
Lamiaceae	*Hoslundia opposita* Vahl	Shrub	0	1	0	LC	Native
Lamiaceae	*Premna angolensis* Gürke	Tree/Shrub	0	2	0	LC	Native
Malvaceae	*Dombeya mukole* Sprague	Tree	2	0	3	DD	Native
Malvaceae	*Triumfetta tomentosa* Bojer	Shrub	0	0	7	LC	Native
Malvaceae	*Leptonychia usambarensis* K. Schum.	Tree	4	6	4	LC	Native
Malvaceae	*Pterygota mildbraedii* Engl.	Tree	25	23	19	LC	Native
Marantaceae	*Marantochloa leucantha* (K. Schum.) Milne-Redh.	Herb	1	4	0	LC	Native
Meliaceae	*Trichilia splendida* A. Chev.	Tree	2	7	3	LC	Native

Metteniusaceae	*Apodytes dimidiata* var. acutifolia Hochst. ex A. Rich.	Tree	3	40	0	LC	Native
Monimiaceae	*Xymalos monospora* (Harv.) Baill. ex Warb.	Tree	0	1	1	LC	Native
Moraceae	*Antiaris toxicaria* subsp. africana (Engl.) C.C. Berg	Tree	4	0	2	LC	Native
Moraceae	*Trilepisium madagascariense* DC	Tree	1	1	5	LC	Native
Moraceae	*Ficus sansibarica Warb. subsp. macrosperma* (Mildbr. & Burret) C.C.Berg	Tree	0	1	0	LC	Native
Moraceae	*Ficus congensis* Engl.	Tree	1	0	1	LC	Native
Moraceae	*Ficus dawei* Hutch.	Tree	0	1	0	LC	Native
Moraceae	*Ficus urceolaris* Welw. ex Hiern	Tree	50	4	0	LC	Native
Moraceae	*Ficus vallis-choudae* Delile	Tree	1	0	0	LC	Native
Moraceae	*Treculia africana* Decne.	Tree	2	0	5	LC	Native
Nephrolepidaceae	*Nephrolepis biserrata* (Sw.) Schott	Herb	0	0	30	LC	Native
Olacaceae	*Strombosia scheffleri* Engl	Tree	2	11	13	LC	Native

Oleaceae	*Linociera johnsonii* Baker	Tree/Shrub	1	0	2	LC	Native
Phyllanthaceae	*Bridelia micrantha* (Hochst.) Baill.	Tree	4	1	1	LC	Native
Phyllanthaceae	*Phyllanthus maderaspatensis* L.	Climber	1	1	0	LC	Native
Piperaceae	*Piper capense* L. f.	Shrub	0	14	10	LC	Native
Piperaceae	*Piper guineense* Schumach. & Thonn.	Shrub	12	13	3	LC	Native
Piperaceae	*Piper umbellatum* L.	Shrub	0	14	6	LC	Native
Poaceae	*Pharus latifolius* L.	Herb	103	43	81	LC	Native
Poaceae	*Setaria* P. Beauv.	Herb	0	9	14	LC	Native
Primulaceae	*Maesa lanceolata* Exell	Tree/Shrub	2	1	0	LC	Native
Rosaceae	*Prunus africana* (Hook. f.) Kalkman	Tree	2	0	0	VU	Native
Rosaceae	*Rubus apetalus* Poir.	Shrub	0	0	3	LC	Native

Rubiaceae	*Coffea eugenioides* S. Moore	Shrub	1	4	2	LC	Native
Rubiaceae	*Psychotria capensis* (Eckl.) Vatke	Shrub	90	6	3	LC	Native
Rubiaceae	*Rothmannia urcelliformis* (Schweinf. ex Hiern)	Tree/Shrub	4	2	2	LC	Native
Rubiaceae	*Vangueria apiculata* K. Schum.	Tree/Shrub	1	0	0	LC	Native
Rutaceae	*Citropsis articulata* Willd. ex Spreng.	Tree/Shrub	2	0	3	VU	Native
Rutaceae	*Fagaropsis angolensis* (Engl.) Dale	Tree	1	1	3	VU	Native
Rutaceae	*Vepris nobilis* (Delile) Mziray	Tree/Shrub	18	24	66	LC	Native
Salicaceae	*Dovyalis macrocalyx* (Oliv.) Warb.	Tree/Shrub	1	3	3	LC	Native
Salicaceae	*Scolopia rhamniphylla* Gilg	Tree/Shrub	0	3	0	LC	Native
Sapindaceae	*Allophylus abyssinicus* (Hochst.) Radlk.	Tree	0	4	1	LC	Native

Sapindaceae	*Allophylus macrocarpus* Danguy & Choux	Tree/Shrub	4	4	2	LC	Native
Sapindaceae	*Aphania senegalensis* (Juss. ex Poir.) Radlk.	Tree/Shrub	9	8	6	LC	Native
Sapindaceae	*Blighia unijugata* Baker	Tree	4	5	4	NT	Native
Sapindaceae	*Cardiospermum halicacabum* L.	Climber	0	0	50	LC	Native
Sapindaceae	*Pancovia turbinata* Radlk.	Tree	2	3	4	LC	Native
Sapotaceae	*Chrysophyllum gorungosanum* Engl.	Tree	17	7	28	LC	Native
Sapotaceae	*Chrysophyllum albidum* G. Don	Tree	11	11	13	VU	Native
Sapotaceae	*Mimusops bagshawei* S. Moore	Tree	8	13	7	LC	Native
Stilbaceae	*Nuxia congesta* R. Br. ex Fresen.	Tree	4	2	0	LC	Native
Ulmaceae	*Chaetacme aristata* E.Mey. ex Planch.	Tree/Shrub	1	2	0	LC	Native
Urticaceae	*Laportea ovalifolia (Schumach.) Chew.*	Herb	0	0	4	LC	Native

Urticaceae	*Urtica dioica* L.	Herb	0	1	50	LC	Alien
Violaceae	*Rinorea* Aubl.	Tree/Shrub	3	5	6	LC	Native
Zingiberaceae	*Aframomum spiroligulatum* A.D. Poulsen & Lock	Herb	87	34	13	LC	Native
Zygophyllaceae	*Balanites wilsoniana* Dawe & Sprague	Tree	12	4	10	LC	Native